# The Challenge of the Urine Patch for Managing Nitrogen in Grazed Pasture Systems

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## Abstract

Ruminants excrete as much as 70–95% of the nitrogen (N) they consume. The urine patch is the conduit through which much of this N is recycled in grazed pasture systems. This chapter focuses on three key areas: urine patch characteristics and N cycling processes; implications for N cycling at the farm and paddock scale; and strategies available to mitigate N losses from the urine patch. The urine patch N loading rate is a key metric for quantifying and modeling fate of N; yet it is a derived value, relying on estimates of urine volume and N concentration, and the urine patch surface area, all of which are variable. Much is known about N cycling processes in the urine patch but further understanding of N loss, leaching of dissolved organic N, and mineralization–immobilization turnover is needed. Typical values (as a percentage of the deposited urinary N) were estimated as: 13% ammonia volatilization; 2% nitrous oxide emission; 20% nitrate leaching; 41% pasture uptake; 26% gross immobilization. The relative importance of each process is influenced by urine patch characteristics and environmental factors. Models are an important tool for scaling from the individual urine patch to the paddock and farm scale, though accounting for variability in urine patch characteristics, and spatial and temporal distribution, remains a challenge. Many potential management strategies to decrease N loss from the urine patch are still at the proof of concept stage with few actually deployed on the farm. Further research is required to integrate these into farm management systems.

## 1. INTRODUCTION

Nitrogen (N) is essential for the growth and reproduction of plants and animals. Manufactured N fertilizers are critical to support food production, and this demand is growing (Gruber and Galloway, 2008). However, the agricultural N cycle is "leaky." Losses of "reactive" N (N forms other than inert $N_2$ gas) from agricultural systems include: emissions of nitrous oxide ($N_2O$), contributing to global warming (IPCC, 2007); emissions of ammonia ($NH_3$) which can be redeposited, causing N enrichment of sensitive ecosystems; and leaching of soluble N forms, causing adverse effects in waterways (Sutton et al., 2011). Nitrogen lost from agricultural systems also represents an economic loss and a reduction in N-use efficiency (Rotz et al., 2005). Of the approximately 100 Tg of fertilizer N used annually in agriculture, only 17 Tg is consumed in dairy, meat, and crops (Rotz et al., 2005).

Grazed pasture systems, where meat, milk, and fiber are produced from ruminant animals, vary in intensity from extensive sheep and cattle systems where animals graze outdoors all year round, to intensive animal production systems involving housing and diets with 100% brought-in feeds. The systems differ in their relative reliance on N inputs from N fertilizer, N fixation (atmospheric/biological), and brought-in feeds. A common feature, however, is that ruminants are inefficient users of dietary N. Only 5–30% of consumed N is converted into products, meaning that 70–95% of ingested N is excreted in urine and dung (Oenema et al., 2005). The majority of excreted N is returned as urine, which contains N in more readily available forms than does dung (Jarvis et al., 1995). As N intake increases, the proportion of N excreted in dung remains relatively constant while the proportion excreted in urine increases (Barrow and Lambourne, 1962; Jarvis et al., 1995). Furthermore, the N loading rate in urine patches often exceeds the N requirements of the pasture and the excess is vulnerable to loss. The longer animals spend grazing, the greater is the contribution of directly voided urine to N losses. Therefore, the focus of this chapter is N deposited in the urine patch.

A number of reviews have considered N cycling through animal production systems (Jarvis et al., 1995; Ledgard et al., 1999; Rotz et al., 2005), yet few have focused specifically on the urine patch as the "engine room" of N cycling in grazed pasture systems. We hypothesize that understanding more about urine patch characteristics and the underlying N cycling will help to identify management interventions that will improve N-use efficiency. Haynes and Williams (1993b) undertook an analysis of information available at the time. The objectives of this chapter were to revisit this state of knowledge to:

1. Enable better understanding of urine patch N dynamics including processes and driving factors;
2. Determine the implications of the concentrated returns of N in urine patches for N cycling and losses at the paddock and farm scale;
3. Define management strategies to reduce urinary N losses in grazed pasture systems.

## 2. THE URINE PATCH IN GRAZED PASTURE SYSTEMS

### 2.1 Urine Composition

The concentration of N in ruminant urine varies from less than 1 to over 20 g N $L^{-1}$ (Table 1; Whitehead, 1995; Dijkstra et al., 2013). Our *meta*-analysis of published trials produced average urine N concentrations of 6.9 ($n = 51$), 7.2 ($n = 8$), 8.7 ($n = 10$), 4.1 ($n = 1$), and 12 g N $L^{-1}$ ($n = 2$) for dairy cattle, beef cattle, sheep, deer, and goats, respectively, grazing a predominantly grass diet. For Table 1, experiments were selected on the basis that they included replication, of either >1 measurement from the same animal, or a number of animals. In the only published study of grazing deer identified, the average urine N concentration was 4.1 g N $L^{-1}$ (Hoogendoorn et al., 2010). Ruminant animals grazing the same pasture have been assumed to produce approximately the same urine N concentration (10 g N $L^{-1}$ for sheep and cattle; Haynes and Williams, 1993b), whereas in our *meta*-analysis, the average N concentration for sheep and cattle ranged from 2.0 to 12.0 g N $L^{-1}$. However on average, it seems that the N concentrations *within* species are greater than *between* species (Hoogendoorn et al., 2010).

The most important factors influencing urine N concentration are the N intake (driving the amount of N surplus to metabolic requirement) and the water intake (affecting both volume and frequency of urination). As described previously, increased N intake increases the amount of N excreted in urine. However, a high N diet does not necessarily correspond to high urine N concentration: water intake tends to increase in higher N diets, which helps to dilute the N in the urine. Van Vuuren and Smits (1997) reported a 74% increase in urination volume following a change from a low to a high N diet, while the N concentration increased only marginally, from 7.4 to 7.6 g N $L^{-1}$. This dilution effect has also been highlighted in studies where animal diets were supplemented with salt (Van Vuuren and Smits, 1997; Spek et al., 2012).

Urine N concentration also varies with animal reproductive status, season, and time of day (Petersen et al., 2004; Betteridge et al., 1986; Hoogendoorn et al., 2010; Bryant et al., 2013). Methods for estimating urine N concentration are "spot" sampling (Hoogendoorn et al., 2010), urine sensors (Betteridge et al., 2010b), and modeling based on intake of N; sodium and potassium (Bannink et al., 1999). Bryant et al. (2013) recognized the need for caution when interpreting urine N concentration from spot sampling,

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**Table 1** Characteristics of urine patches deposited by dairy and beef cattle, sheep, and deer grazing predominantly pasture-based diets. Of the range of studies reported in Haynes and Williams (1993b), we have presented the authors average only

| Study | Country | Species (class) | Urine N concentration ($g\ N\ L^{-1}$) | | Urination volume (L) | | Urination frequency (# per day) | | Urine patch area (wetted) ($m^2$) | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Average | Range | Average | Range | Average | Range | Average | Range |
| Betteridge et al. (1986) | New Zealand | Cattle (beef) | 6.7* | 3.9–9.4 | | | | | | |
| Haynes and Williams (1993b) | New Zealand | Cattle (beef/dairy) | 10 | 8–15§ | 2 | 1.6–2.2 | | | 0.20 | 0.16–0.49 |
| Hoogendoorn et al. (2010) | New Zealand | Cattle (beef) | 4.4 | 0.9–13.2 | | | | | | |
| Jarvis et al. (1995) | The United Kingdom | Cattle (beef/dairy) | | | | | 9 | | | |
| Weeth and Lesperance (1965) | The Netherlands | Cattle (beef) | 5.7 | | 8.4 | | | | | |
| Whitehead (1995) | The United Kingdom | Cattle (beef/dairy) | 9.0 | 2–20 | 2 | | | | | |
| Aland et al. (2002) | Sweden | Cattle (dairy) | | | | | 9 | 5–18 | | |
| Bristow et al. (1992) | New Zealand | Cattle (dairy) | 11 | 6.8–20.5 | | | | | | |
| Bryant et al. (2013) | New Zealand | Cattle (dairy) | 4.5 | 0.9–10.8 | | | | | | |

(*Continued*)

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**Table 1** Characteristics of urine patches deposited by dairy and beef cattle, sheep, and deer grazing predominantly pasture-based diets. Of the range of studies reported in Haynes and Williams (1993b), we have presented the authors average only—cont'd

| Study | Country | Species (class) | Urine N concentration (g N $L^{-1}$) | | Urination volume (L) | | Urination frequency (# per day) | | Urine patch area (wetted) ($m^2$) | |
|---|---|---|---|---|---|---|---|---|---|---|
| Clark et al. (2010) | New Zealand | Cattle (dairy) | | | | | 14 | 11–16 | | |
| Gonda and Lindberg (1994) | The Netherlands | Cattle (dairy) | 8.7 | 5.8–10.7 | | | | | | |
| Kool et al. (2006a) | The Netherlands | Cattle (dairy) | 9.7 | 9.0–10.3 | | | | | | |
| Lantinga et al. (1987) | The Netherlands | Cattle (dairy) | 8.0 | 6.1–9.7 | | | | 10–12 | | |
| Pakrou and Dillon (1995) | Australia | Cattle (dairy) | 9.0 | 5.9–12.3 | | | | | | |
| Petersen et al. (1998) | Denmark | Cattle (dairy) | 9.1 | 4.8–13.3 | | | | | | |
| Saarijärvi and Virkajrvi (2009) | Finland | Cattle (dairy) | 7.5 | | 2.4 | | | | 0.35 | |
| Spek et al. (2012) | The Netherlands | Cattle (dairy) | 6.0 | 3.0–10.4 | | | | | | |
| van Vuuren and Smits (1997) | The Netherlands | Cattle (dairy) | 6.0 | 3.9–7.6 | | | | | | |
| Welten et al. (2013b) | New Zealand | Cattle (dairy) | | | | | | | 0.14* | 0.05–0.31 |

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| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Bristow et al. (1992) | New Zealand | Sheep | 8.8 | 3.0–13.7 | | | | | | |
| Haynes and Williams (1993b) | New Zealand | Sheep | 10 | | 0.15 | 0.10–0.18 | | | 0.03 | 0.03–0.05 |
| Hoogendoorn et al. (2010) | New Zealand | Sheep | 7.9 | 1.4–17.8 | | | | | | |
| Ledgard et al. (2008) | New Zealand | Sheep | | | 2 | 0.5–3.0 | | | | |
| Shand et al. (2002) | Scotland | Sheep | 5.1 | | | | | | | |
| Sherlock and Goh (1984) | New Zealand | Sheep/goat | 10 | 5–15 | | | | | | |
| Bristow et al. (1992) | New Zealand | Goat | 14 | 12.0–16.9 | | | | | | |
| Hoogendoorn et al. (2010) | New Zealand | Deer | 4.1 | 0.5–16.6 | | | | | | |

* median.
§8–15 g N $L^{-1}$ range quoted by Whitehead (1995).

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**Table 2** Nitrogenous constituents of dairy cattle urine and their relationship to urine patch N transformation pathways

| Urinary constituent | Concentration* ($g\ L^{-1}$) | % of total N† | Pathway affected (response to concentration increase) |
|---|---|---|---|
| Total N | 8.2 | — | Volatilization, denitrification, leaching, uptake, N transformations |
| Urea | 6.0 | 73 | Volatilization (increase) |
| Allantoin | 0.86 | 10 | |
| Hippuric acid | 0.51 | 6.2 | Volatilization, denitrification ($N_2O$; both increase and decrease) |
| Creatinine | 0.26 | 3.2 | |
| Creatine | 0.26 | 3.2 | |
| Ammonia | 0.20 | 2.4 | Volatilization |
| Amino acids | 0.15 | 1.8 | Volatilization |
| Uric acid | 0.08 | 0.98 | |
| (Hypo)xanthine | 0.05 | 0.61 | |

*Adapted from Dijkstra et al. (2013); data are average of the average reported concentrations (Bristow et al., 1992; Gonda and Lindberg, 1994; Kool et al., 2006a; Lantinga et al., 1987; Spek et al., 2012; Van Vuuren and Smits, 1997).
†Calculated using average concentration in adjacent column.

e.g., during milking or feeding indoors or from a single animal during one season, because they do not take into account the major sources of variability mentioned above.

Urine contains a number of N compounds. Urea is the largest proportion of urinary N but also includes allantoin, hippuric acid, creatine, creatinine, and ammonia (Table 2). As N intake increases, the proportion of urinary N present as urea increases (Topps and Elliott, 1967; Petersen et al., 1998), from as low as 25% in a sheep diet with low protein, to 90% for cows grazing heavily fertilized, high N-containing grass (Jarvis et al., 1995). The proportion of total N in urine as urea has been shown to be higher in the morning than evening (Petersen et al., 1998; Bryant et al., 2013).

Urine composition has also been shown to influence N transformation processes (Clough et al., 2003b; Decau et al., 2004; Petersen et al., 2004; van Groenigen et al., 2005a). "Artificial" urine has produced differences in N dynamics to that of "real" urine (Kool et al., 2006a), and real urine is recommended for experimental use (de Klein et al., 2003); or at least, artificial urine containing urea and hippuric acid as the main N sources (Kool et al., 2006a).

## 2.2 Urination Volume and Frequency

Daily urination volume is influenced mainly by water intake, which is related to the mineral load ingested and excreted by the animal. Large variation in urination volume and frequency is evident, both between individual animals and with time of day (Betteridge et al., 1986). Based on Table 1, the average urination volume is 2.1 L for dairy cattle ($n = 8$), 1.2 L for beef cattle ($n = 3$), and 0.5 L for sheep ($n = 6$). No published studies for deer or goats were found. Urine volume is rarely measured, especially the volume of individual urinations, which is possibly due to the cost and labor requirement for urine capture in the paddock or in specially designed housing (Clark et al., 2010). Another reason for the lack of data is the research gap between N partitioning-type studies and experiments that focus on urine patch N dynamics; few studies have measured urine volume, frequency of urination, or urine patch area. The frequency of urination is more stable, at 10–12 events per day, on average, for cattle (range 9–14), or 18–20 per day for sheep (Lantinga et al., 1987; Haynes and Williams, 1993b; Clark et al., 2010).

## 2.3 Urine Patch Area

The area of a urine patch can be defined by (1) the wetted area, where urine is directly voided and (2) the area immediately outside the wetted area where plants can access urinary N through root extension and N diffusion through the soil (Lantinga et al., 1987; Tinker and Nye, 2000). These two areas, combined, can be termed the "effective area" of a urine patch. Accounting for the effective area (not only the wetted area) is important for accurately estimating N-removal processes.

The wetted area covered by a single urination for cattle ranged from 0.14 to 0.49 $m^2$ with an average area of 0.24 $m^2$ ($n = 6$) from our *meta*-analysis of published studies; this is similar to the wetted area of 0.2 $m^2$ assumed by Haynes and Williams (1993b) for a dairy cow urine patch. For sheep, the average wetted area has been reported as 0.03 $m^2$ by Haynes and Williams (1993b). The effective area is estimated to range from 0.03 to 1.1 $m^2$ for cattle, with an average area of 0.68 $m^2$ (Lotero et al., 1966; Lantinga et al., 1987; Moir et al., 2011).

The variation in urine patch area contributes to the large-scale spatial heterogeneity observed in soil N concentrations (Bertram et al., 2009).

The urine patch area is determined by the volumes of urine deposited, wind, slope, antecedent soil moisture, and soil physical conditions. Measurements of urine patch area (wetted or effective) in the literature are rare, and methods vary. Saarijärvi and Virkajrvi (2009) measured the wetted area by placing paper over a freshly deposited patch and measuring the area of the paper wet by the urine. Welten et al. (2013b) visually identified the urine patch in the paddock using a chain to mark the edge of the wetted area.

The volume of soil wetted by a urine patch varies with surface area, soil moisture, surface water repellence, surface compaction, micro-topography, vegetation cover, slope, and wind (Williams and Haynes, 1994). Tracer studies have shown that preferential flow of urine occurs in some soils, with the wetting front penetrating as deep as 400 mm (Williams and Haynes, 1994) and with up to 46% of the urine moving below the top 150 mm of soil (Williams et al., 1990). Monaghan et al. (1999) showed that up to 68% (average 17%) of the urine moved below 200 mm depth within 6 h of application to different soil types. This same study reported that up to 73% of urinary N remained in the top 100 mm of soil 6 h after urine application; Williams and Haynes (1994) measured up to 50% of deposited urinary N in the top 50 mm of soil, thus demonstrating the highly variable penetrative depth of urine depositions. This variation in the surface area and soil volume affected by a urine patch, along with soil heterogeneity and climatic variation, creates large variation in the potential for urinary N uptake by plants, as well as the potential for N losses.

## 2.4 Urine Patch N Loading Rate

The amount of N deposited or the "N loading rate" in a urine patch is a function of the N concentration of the urine, the urine volume excreted, and the surface area receiving urine:

$$\left(\text{Urine N rate}\left(\text{kg N ha}^{-1}\right) = \text{Conc}\left(\text{g N L}^{-1}\right) \times \frac{\text{Vol (L)}}{\text{Surface area }\left(\text{m}^2\right)} \times 10\right).$$

The most commonly cited reference for urine N loading rate is Haynes and Williams (1993b), who assumed an average N load of 1000 kg N ha$^{-1}$ for a dairy cow, which consisted of: urine N concentration of 10 g N L$^{-1}$, urination volume of 2 L, and urine patch surface area (wetted) of 0.2 m$^2$. For sheep, a similar loading rate of 500 kg N ha$^{-1}$ was assumed by Haynes and Williams (1993b), based on 10 g N L$^{-1}$ urine, 0.5 L, and 0.1 m$^2$. Using the updated averages from our *meta*-analysis, we calculate the average urine N loading rate to be: 613 kg N $ha^{-1}$ for dairy cattle, 345 kg N $ha^{-1}$ for beef cattle, and 1089 kg N $ha^{-1}$ for sheep. This calculated value for sheep is considerably higher than the 500 kg N $ha^{-1}$ rate reported by Haynes and Williams (1993a) and is a likely to be a function of the 0.5-L average volume ($n = 6$) from the *meta*-analysis. A large uncertainty around this value is suggested.

In recognition of the variable urine N concentration and urination volume from grazing ruminants, a range of urine N loading rates is reported: in summary, 200–2000 kg N $ha^{-1}$ for cattle (Lantinga et al., 1987; Jarvis et al., 1995; Oenema et al., 1997; Di and Cameron, 2002b; Bolan et al., 2004). Pakrou and Dillon (1995) recognized the effect of season on urine N loading rate, when they reported 653 kg N $ha^{-1}$ for an autumn urine patch and 1366 kg N $ha^{-1}$ for a spring urine patch deposited by grazing dairy cattle. We found only one study, by Saarijärvi and Virkajrvi (2009), which measured urine concentration (7.5 g N $L^{-1}$), volume (2.4 L), and wetted area (0.35 $m^2$) for a dairy cow, and they therefore calculated an N loading rate of 514 kg N $ha^{-1}$. This was slightly less than our literature average of 613 kg N $ha^{-1}$ for a dairy cow, but within the 200–2000 kg N $ha^{-1}$ range reported above. The message is that, for improved data on urine patch N loading rate, experiments need to measure volume, N concentration, and surface area.

## 2.5 Conditions in the Urine Patch

Following urine deposition, there is a high soil N concentration beneath the urine patch. Urea hydrolysis is rapid, with 80–90% of urea being hydrolyzed within 48 h (Williams and Haynes, 1994). During the conversion from urea to $NH_4HCO_3$, hydroxide ions ($OH^-$) are produced, which raises soil pH to as much as pH 8 in the first 5 days after urine deposition. The subsequent conversion of $NH_4^+$ to $NO_3^-$ during nitrification leads to a decrease in soil pH over a 2- to 3-week period (Haynes and Williams, 1992).

The process of nitrification is central to urine patch N dynamics because it controls the amounts of $NH_4^+$ and $NO_3^-$ as substrates for the various N pathways. Nitrification is the two-step biological oxidation of $NH_4^+$ consisting of: (Step 1) $NH_4^+$ to $NO_2^-$ and (Step 2) $NO_2^-$ to $NO_3^-$ (Cameron et al., 2013). Because urea hydrolysis is usually rapid, the resulting concentration of $NH_4^+$ in urine-affected soil may be initially high (>200 mg N kg $soil^{-1}$) (Haynes and Williams, 1992). In the urine patch, the conversion of $NH_4^+$ to $NO_3^-$ via nitrification is usually complete within 30 days (Williams and

Haynes, 2000; Moir et al., 2011), although longer periods have been reported.

Typically, the high N substrate availability and soil aeration promotes nitrification. However the rate of nitrification may be slowed by the rapid increase in pH and high concentration of $NH_4^+$, leading to inhibition of activity of $NO_2^-$ oxidizing bacteria (Monaghan and Barraclough, 1992; Smith et al., 1997). Nitrite oxidizers, e.g., *Nitrobacter,* are more sensitive to elevated $NH_3$ than $NH_4^+$ oxidizing *Nitrosomonas* (Smith et al., 1997), therefore $NO_2^-$ may accumulate where large amounts of urine N are applied (Clough et al., 2003b). A urine N concentration of 16 g N $L^{-1}$ was suggested as a threshold above which nitrification was affected (Monaghan and Barraclough, 1992), whereas Smith et al. (1997) suggested that the rate of $NH_4^+$ oxidation was only required to be slightly higher than $NO_2^-$ oxidation in order to generate $NO_2^-$ accumulation.

The ionic strength of the soil solution in the surface 25 mm can also increase following urine application; this combination of fluctuations in soil solution pH and ionic strength might indirectly affect nutrient availability in the urine patch (Haynes and Williams, 1992).

## 3. NITROGEN-REMOVAL PROCESSES IN THE URINE PATCH

### 3.1 Ammonia Volatilization

Most of the $NH_3$ volatilization from excreta deposited during grazing comes from the urine patch. Laubach et al. (2013) measured 89% of $NH_3$ volatilization from urine and 11% from dung in a grazed paddock, with $NH_3$ volatilization from dung noticeable by its delayed peak in emissions (c.3 days) compared with urine. Others suggest that the $NH_3$ loss from dung/feces is negligible (Mulvaney et al., 2008; Petersen et al., 1998).

#### *3.1.1 The Process and Factors Affecting $NH_3$ Volatilization*

The local soil pH rise after urine deposition causes $NH_4^+$ to disassociate to $NH_3$ and be volatilized. Because urea hydrolysis is rapid, volatilization starts to occur shortly after urine deposition and continues until the soil $NH_4^+$ pool is depleted by volatilization and other competing processes. Consequently, rates tend to peak 1–2 days after urine application (Whitehead et al., 1989). A diurnal pattern of $NH_3$ loss has also been noted due to the diurnal pattern of soil temperatures (Saarijärvi et al., 2006). The chemistry of urea hydrolysis and volatilization is well documented, as are the key physical (soil pH, cation exchange capacity (CEC)) and environmental (temperature, wind speed, soil moisture) drivers of volatilization (Bolan et al., 2004). Below, we consider factors specific to losses from the urine patch.

Soil CEC is an important soil property influencing volatilization from urine because urine infiltrates into the soil, with greater contact between the resultant ammonium-N and soil matrix than occurs with solid fertilizers (Whitehead and Raistrick, 1993). Soils with a higher CEC will volatilize less.

Petersen et al. (1998) suggested that some of the variation in $NH_3$ volatilization that could not be explained by physical and environmental factors might be due to differences in urine composition. For example, N intake by grazing animals affects the total amount of N excreted and the proportion excreted as urine, as described earlier. Urine urea hydrolysis is more rapid than hydrolysis of pure urea (Bolan et al., 2004). The high pH of urine favors hydrolysis. The presence of hippuric acid in the urine also increases the pH effect, thus increasing volatilization rate, particularly in the first 1–2 days (Whitehead et al., 1989). Although hippuric acid is a minor constituent of ruminant urine, it has important implications for the methodology used to quantify $NH_3$ emissions from synthetic urine.

### *3.1.2 Typical N Losses and Management of $NH_3$ Volatilization*

Obtaining reliable estimates of $NH_3$ losses is confounded by methodological considerations. Methods range from whole-paddock assessments using micrometeorological methods through to enclosure-based methods that measure losses from individual dung or urine patches. Given that the primary source of $NH_3$ from grazed pasture is from urine, it might be expected that micrometeorological measurements across a grazed paddock will give a good indication of $NH_3$ losses from individual urine patches. However, urinary N that has been volatilized can be recycled by pasture because shoots act as a sink for the volatilized $NH_3$ (Frank et al., 2004). Ross and Jarvis (2001) demonstrated that 20–60% of $NH_3$ emitted is redeposited within 2 m, even though absorbed $NH_3$ can then be further reemitted. The switch between absorption and desorption across a sward means that the sum of the $NH_3$ emitted from individual urine patches could exceed the net flux of $NH_3$ leaving the paddock. This potential difference in measured size of loss depending on measurement scale is an important consideration when estimating $NH_3$ loss from an individual urine patch. Table 3 summarizes reported measurements of $NH_3$ volatilization at the urine patch scale using enclosure-based methods. The data were notable for their variability, even when conditions between experiments were considered to be similar.

**Table 3** A summary of ammonia volatilization measurements made at the urine patch scale

| Authors | Location | Species | No. measurements | Season | Soil type[‖] | Urine N rate (kg N ha$^{-1}$) | Days measured | N loss (%) |
|---|---|---|---|---|---|---|---|---|
| ***Chamber method*** | | | | | | | | |
| Di and Cameron (2004) | New Zealand | Dairy | 1 | Autumn | FSL | 1000 | 18 | 3.5 |
| Leterme et al. (2003) | France | Dairy | 3 | Spring, summer, autumn | ZL | c.800[¶] | 5 | 1–3 |
| Menneer et al. (2008a) | New Zealand | Dairy | 1 | Autumn | LS | 775 | 20 | 14.0 |
| Saarijärvi et al. (2006) | Finland | Dairy | 8 | Summer | SL | 1130 | 5–8 | 2.2–18.4 |
| Sherlock and Goh (1984) | New Zealand | Sheep | 5 | Summer, autumn, winter | ZL | 500 | 6–10 | 12–38 |
| Vallis et al. (1982) | Australia | Cattle | 3 | Winter, spring, summer | SL | 364–476 | 14 | 18.8–28.4 |
| Carran et al. (1982) | New Zealand | Dairy | 2 | Summer | ZL | 300 | 10 | 20–40[*] |
| Ball et al. (1979) | New Zealand | Beef | 2 | Summer | ZL | 300–600 | 18 | 15–17 |
| Ball and Ryden (1984) | New Zealand | Dairy | 6 | Not specified | | 300–600 | | 6–66[*] |
| Zaman and Blennerhassett (2010) | New Zealand | Dairy | 2[§] | Autumn, spring | ZL | 600 | 14 | 5–7 |

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|---|---|---|---|---|---|---|---|---|
| Zaman et al. (2013) | New Zealand | Dairy | 4[§] | Autumn, spring | ZL | 600 | 14 | 4–23 |
| Zaman et al. (2009) | New Zealand | Dairy | 3[§] | Spring, summer, autumn | ZL | 600 | 14 | 4–17 |
| Zaman and Nguyen (2012) | New Zealand | Dairy | 2[§] | Autumn, spring | ZL | 600 | 12 | 1.5–7 |
| Mulvaney et al. (2008) | The United States | Dairy/ Beef | 4 | Spring, summer, winter | SL | nr | 14 | 1.8–20.9 |
| ***Wind tunnel method*** | | | | | | | | |
| Lockyer and Whitehead (1990) | The United Kingdom | Dairy | 12 | All seasons | SL | 151–714 | 6–15 | 3.7–26.9 |
| Petersen et al. (1998) | Denmark | Dairy | 4 | Summer | LS | 208–466 | 10 | 8–24 |
| Ryden et al. (1987) | The United Kingdom | Dairy | 4 | Summer, autumn | CL | 420–437 | 14 | 8.8–24.7 |

*High values are where urine was applied to dry soil.
[§]Excludes inhibitor treatments.
[¶]Estimated from experiment description.
[||]Soil types: FSL (fine sandy loam); ZL (silt loam); LS (loamy sand); SL (sandy loam); CL (clay loam).

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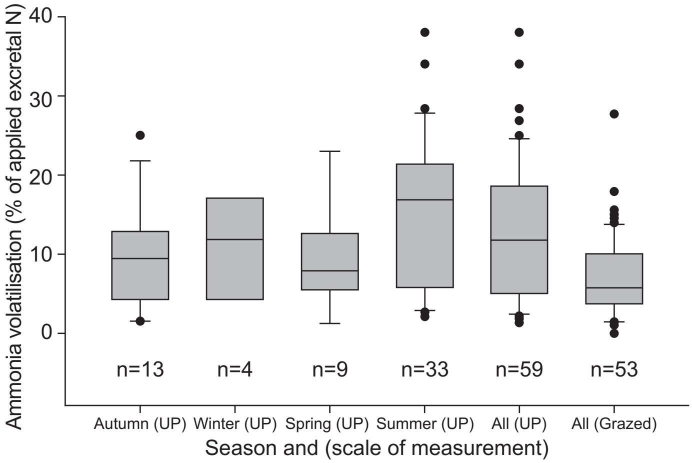


**Figure 1** Effect of season and scale of measurement on $NH_3$ volatilization losses from urine-treated pastures. "UP" means urine patch, measured by wind tunnel or chamber methods; "Grazed" means whole-grazed area measured by micrometeorology methods. The horizontal line is the median, and upper and lower limits of the box represent the 75% and 25% percentiles. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, respectively. Outlying points are plotted as closed circles.

The average loss was 12.9% of urinary N applied (median 11.9%, $n = 59$) with a range of 1–38%. Three values were considered atypical and omitted because the urine had been applied to dry soil and these resulted in larger losses than in any other experiment (40–66%). Most measurements were made in the summer (average volatilization 15%, Figure 1), but the seasonal effect was not statistically significant, possibly because few measurements were made outside of summer. As would be expected, there was a weak but significant relationship ($P < 0.01$) between average air temperature and the proportion of urinary N that was volatilized.

Based on experiments where $NH_3$ volatilization has been measured at the "whole-system" grazing level using micrometeorological methods, reported losses tend to be at the lower end of reported losses for individual urine patches (Figure 1): mean = 8%, range 1–28%, $n = 53$ (Ryden et al., 1987; Bussink, 1992, 1994). However, these paddock-scale measurements come from fewer experiments.

Urease inhibitor decreases $NH_3$ volatilization from urine by slowing urea breakdown, thus decreasing the amount of $NH_4^+/NH_3$ in the soil. The efficacy of the inhibitor depends on many factors, but a recent analysis by Saggar et al. (2012) indicated that a c.50% reduction in $NH_3$ volatilization from an individual urine patch could be achieved by applying N-(*n*-butyl) thiophosphoric triamide to the urine patch. The application of "double inhibitor" combining a urease inhibitor with a nitrification inhibitor has also been trialled. Use of a nitrification inhibitor holds more N in an ammonium form and can therefore increase $NH_3$ volatilization (Kim et al., 2012). There is evidence that inclusion of a urease inhibitor negates the extra risk of volatilization loss from urine treated with nitrification inhibitor alone (Zaman and Blennerhassett, 2010); however, there is no evidence of a synergistic effect (Saggar et al., 2012).

## 3.2 Denitrification and Associated Processes

The chemistry and key drivers of denitrification and other associated N gas-producing processes in agricultural soils have been well documented (Haynes and Sherlock, 1986; Firestone and Davidson, 1989; Mosier, 2001; Saggar et al., 2004). Other than $NH_3$, the main forms of gaseous N lost from urine patches are dinitrogen ($N_2$), nitrous oxide ($N_2O$), and, to a lesser extent, nitric oxide (NO) (Clough et al., 1998, 2003b). Current knowledge indicates that the majority of $N_2O$, $N_2$, and NO gas lost from the urine patch is produced via the biologically mediated processes of denitrification, nitrification, and nitrifier-denitrification (de Klein and van Logtestijn, 1994; Lovell et al., 1995; Carter, 2007) (Figure 2). Here, we focus on these gases and processes in conditions specific to urine patches.

### 3.2.1 The Processes and Factors Affecting Denitrification

The nitrification process results in an accumulation of $NO_3^-$ which is the main source of N for $N_2O$ and $N_2$ emissions. Nitrification therefore supplies substrate for both denitrification and $N_2O$ emissions. As measurement methods have improved, research suggests that other minor pathways contribute more to $N_2O$ and $N_2$ emissions than previously thought: nitrifier-denitrification (Wrage et al., 2001), dissimilatory nitrate reduction

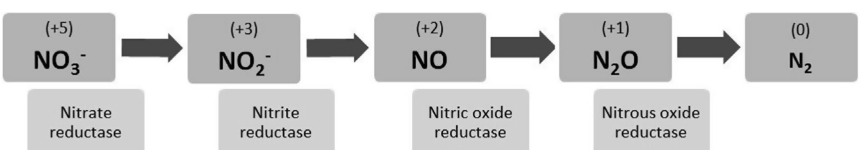


**Figure 2** General sequence of the reduction of $NO_3^-$ in denitrification. *(This was based on a figure published in Mineral Nitrogen in the Plant-Soil System (Ed. Haynes, R.J.), Gaseous losses of nitrogen, Haynes, R.J., Sherlock, R.R, page 260, Copyright Elsevier (1986)).* Values in brackets denote the oxidation state of N.

to ammonium (Mueller, 1995), chemodenitrification (Chalk and Smith, 1983), anaerobic ammonium oxidation, and co-denitrification (Long et al., 2013). However, evidence of emissions from these processes has not yet been reported for urine patches.

The main drivers for loss of $N_2O$ and $N_2$ include: N availability (primarily $NO_3^-$), organic C availability (labile source), pH, and associated environmental factors (temperature, moisture, and $O_2$ availability) (Tiedje, 1988; Firestone and Davidson, 1989; de Klein et al., 2001). Urine contains a large source of labile N and C which, combined with a large water input, generally results in conditions conducive to high denitrification rates.

#### 3.2.1.1 N and C Availability

Rates of denitrification and $N_2O$ fluxes generally increase as N availability increases (Haynes and Sherlock, 1986). Cardenas et al. (2010) reported an exponential increase in cumulative $N_2O$ emissions when the urine N rate increased from 0 to 300 kg N ha$^{-1}$. In contrast, increasing the N loading rate in a urine patch from 0 up to 1000 kg N ha$^{-1}$ did not significantly change the proportion of N applied emitted as $N_2O$, which was consistently less than 0.5% (Selbie et al., 2014). van Groenigen et al. (2005a) found no effect of urine N concentration on $N_2O$ emissions. The addition of urine to soil has been shown to mineralize soil C in an apparent priming effect which may supply more C for denitrification than contained in the urine itself (Monaghan and Barraclough, 1993; Lambie et al., 2013).

#### 3.2.1.2 pH, $O_2$ Availability, and Moisture

Anaerobic conditions in the urine patch increase the potential for $N_2O$ and $N_2$ losses. Increasing pH usually stimulates denitrification, although Clough et al. (2004) found a greater effect on $N_2O$ and $N_2$ emissions in urine-affected soil from an increase in moisture content, than from pH increase. Water-filled pore space (WFPS%) is commonly used to predict the process responsible (Linn and Doran, 1984) and the magnitude of $N_2O$ loss (van der Weerden et al., 2012). van Groenigen et al. (2005b) reported peak $N_2O$ emissions from urine-affected soil occurring at a WFPS% of 60–70%. Pugging and compaction reduce $O_2$ availability and decrease gas diffusivity. When combined with an increase in WFPS, denitrification losses from urine can therefore be large (Anger et al., 2003; van Groenigen et al., 2005b). The relative contribution of nitrification and denitrification to $N_2O$ emissions from urine patches is influenced by $O_2$ availability. de Klein and van Logtestijn (1994) found that denitrification was the main $N_2O$ source process soon after urine application; when the soil moisture became less than 15% v/v, nitrification became the main process responsible.

#### 3.2.1.3 Urine Composition

Hippuric acid and its derivative, benzoic acid, influence $N_2O$ emissions from urine applied to soil (Kool et al., 2006b; Bertram et al., 2009; Clough et al., 2009). Increasing the hippuric acid concentration from 3% to 9% of total urinary N reduced $N_2O$ emissions from 8.4% to 4.4% of the applied N (Kool et al., 2006b). The function is thought to be related to benzoic acid, although the mode of inhibition is not yet clear.

### *3.2.2 Typical N Losses and Management of Denitrification*

As with $NH_3$ volatilization, there are two main approaches for measuring $N_2O$ emissions from field soils: chambers and micrometeorological methods. Chamber techniques are less costly and provide reliable measurements in time and space, but do not reliably represent farm or ecosystem scale $N_2O$ fluxes; here, integrative meteorological techniques become advantageous (Henault et al., 2012). Nitrous oxide emission from grazed pastures have been the focus of much work due to its importance as a greenhouse gas (de Klein et al., 2003; Rees and Ball, 2010), whereas NO and benign $N_2$ losses have been less well explored (Monaghan and Barraclough, 1993; Skiba et al., 1993; Lovell and Jarvis, 1996). A summary of the literature shows an average $N_2O$ loss of 2.1% (range 0–14%) of applied urine N ($n = 40$), from a range of soil types, urine N application rates and seasons (Table 4). In Table 4 there was a tendency toward higher $N_2O$ emission factors in autumn and winter, relative to summer and spring, although there were exceptions. This value of 2.1% is slightly higher than the average of 1.7% reported by van Groenigen et al. (2005a) in a literature review of urine emission factors for $N_2O$ ($n = 31$ studies). Importantly, 2.1% is similar to the default emission factor of 2% which the Intergovernmental Panel on Climate Change (IPCC) assumes for urine N deposited during grazing (IPCC, 2007). de Klein et al. (2003) separated urine emission factors according to rainfall and drainage class, ranging from 0.3% of N applied in a well-drained stony soil to 2.5% of the N applied to a poorly drained soil.

There are few reported values of $N_2$ emissions from urine patches, which is most likely due to the large atmospheric background concentration of $N_2$ (78% of air) causing even a large treatment-induced $N_2$ flux to be below detection limits using current analysis techniques. Monaghan and Barraclough (1993) reported $N_2$ emissions of 30–65% of the urine N applied

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**Table 4** A summary of reported literature values for nitrous oxide, nitric oxide, and dinitrogen losses from urine patches (as % of N applied)

| Study | Location | Soil type | N form | Method[§] | Urine N rate[*] (kg N $ha^{-1}$) | Season[¶] | Soil C (%) | N loss | Duration (days) |
|---|---|---|---|---|---|---|---|---|---|
| Allen et al. (1996) | The United Kingdom | Clay loam | $N_2O$ | FP | 700[r] | Spr. | | 0 | 80 |
| | | | | | 700 | Aut. | | 1.5 | |
| | | Loam | | | 700 | Spr. | | 0 | |
| | | | | | 700 | Aut. | | 2 | |
| Anger et al. (2003) | Germany | Sandy silt loam | $N_2O$ | FP, SC | 1010[a] | Spr. | 3.9 | 1.4–4.2 | 357 |
| | | | | | | Aut. | | 0.3–0.9 | |
| Bol et al. (2004) | Denmark | Sandy loam | $N_2O$ | FP | 233–398[r] | Aut. | 3.6 | 0.015–0.021 | 14 |
| Bronson et al. (1999) | Australia | Sand | $N_2O$, NO | L | 205[r] | Sum. | | 0.0 | 28 |
| Clough et al. (1996) | New Zealand | Peat | $N_2O$ | L, SC | 500[a] | Win. | | <1 | 100 |
| | | Silt loam | $N_2O$ | | | | | 1.5–3 | |
| Clough et al. (1998) | New Zealand | Silt loam | $N_2O$, $N_2$ | L, SC | 1000[a] | Win. | 8.3 | 1 | 112 |
| | | Sandy loam | | | | | 4 | 0.8 | |
| | | Clay | | | | | 7.9 | 1.9 | |
| | | Peat | | | | | 28 | 1.9 | |
| Clough et al. (2003a) | New Zealand | Silt loam | $N_2O$, NO, $NO_2$ | JI | 0–1000[a] | C Cond. | 3.1 | 6.4–0.5 | 21 |
| de Klein et al. (2003) | New Zealand | Silt loam | $N_2O$ | L, SC | 592[r] | Aut. | 10 | 0.3–2.5 | 120 |
| | | Peat | | | 592 | | 45 | 0.3 | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| de Klein et al. (2011) | New Zealand | Sandy loam | $N_2O$ | FP | 1000[r] | Aut. | | 1.3 | ~180 |
| | | Silt loam | $N_2O$ | | 610 | | | 0.9–1.4 | |
| de Klein and van Logtestijn (1994) | The Netherlands | Sandy loam | $N_2O$ | JI | 400[a] | C Cond. | | 8–16 | 14 |
| Di et al. (2007) | New Zealand | Silt loam | $N_2O$ | L | 1000[r] | Aut. | 6.2 | 0.6 | 69–137 |
| | | Pumice sand | | | 700 | | 7.3 | 0.1 | |
| | | Sandy loam | | | 1000 | | 2.4 | 2 | |
| | | Stony silt loam | | | 1000 | | 3.7 | 0.8 | |
| Di and Cameron (2003) | New Zealand | Stony silt loam | $N_2O$ | L, SC | 1000[r] | Aut. | 3.7 | 2.2 | ~180 |
| Di and Cameron (2006) | New Zealand | Sandy loam | $N_2O$ | L, SC | 1000[r] | Aut. | 2.4 | 3.6 | ~90 |
| | | Stony silt loam | | | | | 3.7 | 2.3–2.7 | |
| Di and Cameron (2008) | New Zealand | Sandy loam | $N_2O$ | L, SC | 1000[r] | Aut. | 2.4 | 0.4 | ~90 |
| Flessa et al. (1996) | Germany | Loam | $N_2O$ | FP, SC | 1270[r] | Sum. | 2.5 | 3.8 | 78 |
| Hoogendoorn et al. (2008) | New Zealand | Silt loam | $N_2O$ | FP, SC | 360[a] | Spr. | n.d. | 0.01–1.06 | ~56 days |
| Koops et al. (1997) | The Netherlands | Peat | $N_2O$ | JI | 470[a] | C Cond. | | 1.3 | 31 |
| Lovell and Jarvis (1996) | The United Kingdom | Clay loam | $N_2O$ | JI | 227[r] | C Cond. | | 14 | 38 |
| | | | | | 180[a] | | | 11 | 38 |
| Maljanen et al. (2007) | Finland | | $N_2O$, NO | FP, SC | 583[r] | Aut. | | 0.24 | ~270 |

(*Continued*)

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**Table 4** A summary of reported literature values for nitrous oxide, nitric oxide, and dinitrogen losses from urine patches (as % of N applied)—cont'd

| Study | Location | Soil type | N form | Method[§] | Urine N rate[*] (kg N $ha^{-1}$) | Season[¶] | Soil C (%) | N loss | Duration (days) |
|---|---|---|---|---|---|---|---|---|---|
| Monaghan and Barraclough (1993a) | The United Kingdom | Clay loam | $N_2O$ | L, JI | 477–1186[r] | C Cond. | 5.8 | 1–5 | 30 |
| | | | $N_2$ | | | | | 30–65 | |
| Mueller (1995) | New Zealand | Silt loam | $N_2O$ | FP, SC | 500[a] | Spr. | 2.4 | 0.3 | 90 |
| | | | | | | Sum. | | 0.1 | |
| | | | | | | Aut. | | 0.4 | |
| | | | | | | Win. | | 0.2 | |
| Qiu et al. (2010) | New Zealand | Sandy loam | $N_2O$ | L | 1000[r] | Sum. | 2.4 | 0.76 | ~150 |
| | | | | | | Win. | | 1.27 | ~120 |
| Selbie et al. (in press) | Ireland | Sandy loam | $N_2O$ | L, SC | 0–1000[r] | Aut. | | 0.4 | 80 |
| Sherlock and Goh (1983) | New Zealand | Silt loam | $N_2O$ | JI, SC | 600[r] | C Cond. | 4.2 | <0.5 | 45 |
| Singh et al. (2009) | New Zealand | Sandy loam | $N_2O$ | L | 144[r] | C Cond. | 3.5 | 0.74 | 50 |
| | | | | | 290 | | | 1.09 | |
| | | | | | 570 | | | 3.57 | |

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| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| van der Weerden et al. (2011) | New Zealand | Silt loam | $N_2O$ | FP, SC | 500[r] | Aut. | 5.27 | 0.3 | 125–132 |
| | | Average of three soils | | | 549 | Spr. | | 0.26 | 166–173 |
| van Groenigen et al. (2005b) | The Netherlands | Loamy fine sand | $N_2O$ | FP[‖] | 373[a] | Spr. | 4 | 1.55 | ~90 |
| Vermoesen et al. (1997) | Belgium | Sandy loam | $N_2O$, $N_2$ | FP, SC, JI | 320–555[r] | Spr./Sum. | n.d. | 0.13–1.1 | 19–35 |
| Wachendorf et al. (2008) | Germany | Sandy loam | $N_2O$ | L, SC | 1030[r] | Aut. | 4 | 0.05 | 171 |
| Williams et al. (1999) | The United Kingdom | Coarse silt | $N_2O$ | L, JI | 190[r] | C Cond. | | 7 | 42 |
| Williamson and Jarvis (1997) | The United Kingdom | Clay loam | $N_2O$ | FP, SC | 60[r] | Aut. | | 5 | 37 |
| Yamulki et al. (1998) | The United Kingdom | Clay loam | $N_2O$ | FP, SC | 540[r] | Aut., Spr. | | 0.1–1.4 | 60–417 |

[¶]Aut. = autumn, Wint. = winter, Spr = spring, Sum. = summer, C Cond. = controlled conditions.
[*]r = real urine, a = artificial urine, where reported.
[§]FP = field plots, SC = static chamber, JI = Jar incubation.
[‖]method used on these field plots was vented chamber, photo-acoustic infrared analyzer.

in a study using $^{15}N$-labeled urine, intact soil monoliths, and a box incubation technique. The $N_2$ loss from 1000 kg N $ha^{-1}$ as urine applied to intact soil cores was calculated by Clough et al. (1998) to be: 0.59, 1.31, 2.02, and 0.29% of the $^{15}N$ applied for the silt loam, sandy loam, peat, and clay soils, respectively. These $N_2$ fluxes were consistently higher than the $N_2O$ fluxes, and the ratio of $^{15}N_2$:$N_2O$–N varied between soils, from 6.2 to 33.2. Using the acetylene incubation technique, de Klein and van Logtestijn (1994) reported a total denitrification ($N_2O + N_2$) loss of 18% of applied N, most of which was $N_2O$. Mueller (1995) measured $N_2$ fluxes from field plots using the static chamber method with urine labeled to 50 atom% but the flux remained below the analytical limit of detection most of the time. Reasons for the wide variation in $N_2$ emissions from urine-affected soil are unclear and both the source processes and the quantity of $N_2$ emitted warrant further investigation.

Reported NO emissions from urine patches in the literature suggest that this gas makes a minor contribution to total gaseous N fluxes (Clough et al., 2003b; Maljanen et al., 2007; Dixon et al., 2010). An exception is the study of Bronson et al. (1999), who measured NO emissions $\sim$10 times higher than $N_2O$ emissions in a study of summer-applied urine to a sandy soil. The authors attributed the high NO flux to nitrification, rather than to chemodenitrification (Davidson, 1992).

#### 3.2.2.1 Nitrification Inhibitors

One of the main reported $N_2O$ mitigation options is the use of a nitrification inhibitor, such as dicyandiamide (DCD) or 3,4-dimethylpyrazole phosphate (de Klein et al., 2011; Di and Cameron, 2012; Selbie et al., 2014). By slowing the rate of nitrification, DCD reduces the accumulation of $NO_3^-$, which is the main source of N substrate for denitrification. Total $N_2O$ emissions have been reduced by 60–85% with application of DCD to urine-treated soils (Di and Cameron, 2002c, 2003, 2006). The use of "double inhibitor," the combined application of urease and nitrification inhibitors, was more effective at reducing $N_2O$ emissions than nitrification inhibitor alone applied to autumn and spring urine patches (Zaman and Blennerhassett, 2010).

#### 3.2.2.2 Soil Additives

Soil additives such as biochar and lime aim to shift the ratio of $N_2O$:$N_2$ in favor of benign $N_2$. Taghizadeh-Toosi et al. (2011) reported a 70% reduction in the $N_2O$ emission factor where 30 t $ha^{-1}$ of biochar was

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incorporated into the soil, although the mechanisms for the reduction were not clear. Lime addition increased $N_2$ but had no effect on $N_2O$ emissions from urine-affected soil (Zaman and Nguyen, 2010).

## 3.3 Nitrogen Leaching

Much of the urinary N transported through the soil profile via drainage water to below the root zone is ultimately transferred into groundwater and/or surface water systems. The main form is as $NO_3^-$—N, although $NO_4^+$—N and urea–N have been detected in drainage from urine patches. Leaching of dissolved organic N (DON) also occurs, but is not as well documented as other N forms.

### 3.3.1 The Process and Factors Affecting Leaching

Soluble N is transported in drainage water via a combination of three primary mechanisms collectively termed *combined convective–diffusive–dispersive transport*: convection, where dissolved $NO_3^-$—N moves with the mass flow of water in the soil ("piston displacement"); diffusion, where the uneven distribution of N in solution results in a concentration gradient and the movement of N from areas of high concentrations to lower concentrations; and hydrodynamic dispersion, which is the mixing of soil solute by the mechanical action of water flow through the soil. Additionally, preferential flow of water through surface-connected macropores can occur if deposition rate exceeds soil infiltration rate (e.g., during some urination events) (Cameron and Haynes, 1986). Macropore flow can lead to rapid and extensive N leaching, and losses of $NH_4$–N and urea (which are normally quickly transformed) have been reported soon after urine application.

Solute transport mechanisms are described in further detail by Cameron and Haynes (1986), Hillel (1998), and Cameron et al. (2013). These mechanisms are equally applicable to transport of N from all sources. Factors specific to leaching from urine patches are discussed below.

#### 3.3.1.1 Rate of Urine N Application

Di and Cameron (2007) demonstrated an increase in N leaching with increased urine patch N loading, described by a quadratic function, but which approximated to a linear relationship. This near-linearity very likely occurred because the urine was applied in autumn when N uptake was low and the amount leached was largely determined by the amount of drainage. Shorten and Pleasants (2007) modeled the effects of urine patch overlap on N leaching risk. They estimated that 38, 61, and 71% of the N in single,

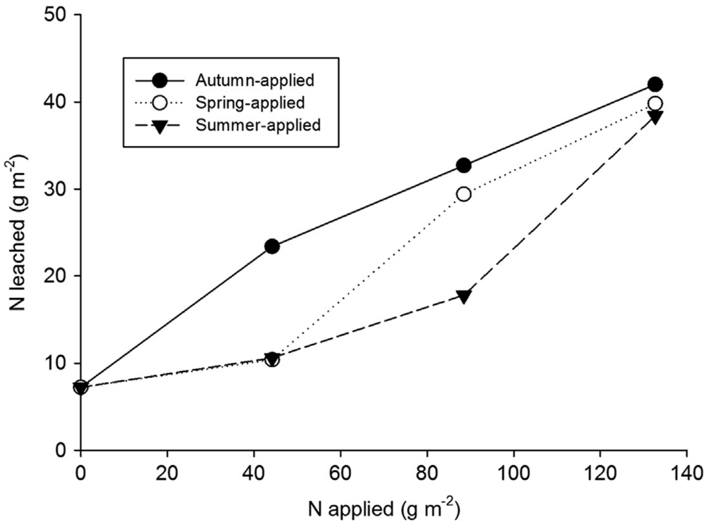


**Figure 3** Relationship between rate and time of urinary N application and resultant N leaching. *Adapted from Stout (2003a).*

double, and triple urine patches would be leached for the climate/soil-type combination they investigated; this equates to a linear relationship between kg N ha$^{-1}$ leached and the urine patch N load. Data adapted from Stout (2003a) also confirmed a close-to-linear relationship between N loading and N leached for autumn-applied urine (Figure 3). There was a more curvilinear relationship for the summer application, probably due to a greater proportion of N removed by pasture at lower N application rates (supported by pasture N uptake data reported by Stout (2003b)), where N uptake for spring/summer-applied urine increased by up to a factor of 10 compared to autumn applications). Trends after the spring application were less clear, even though we would have expected them to be similar to the summer application.

Urine patches may receive other N inputs in the form of subsequent fertilizer or effluent additions, which add to the N load of a patch that already generally has N in excess of pasture requirements. Silva et al. (2005) measured more N leaching from a urine patch that received 400 kg N ha$^{-1}$ as fertilizer over a year than from urine alone. Using $^{15}$N-labeled fertilizer, Buckthought (2014) suggested that additional N leaching was not necessarily fertilizer N, but a substitution in drainage by urinary N for the added fertilizer.

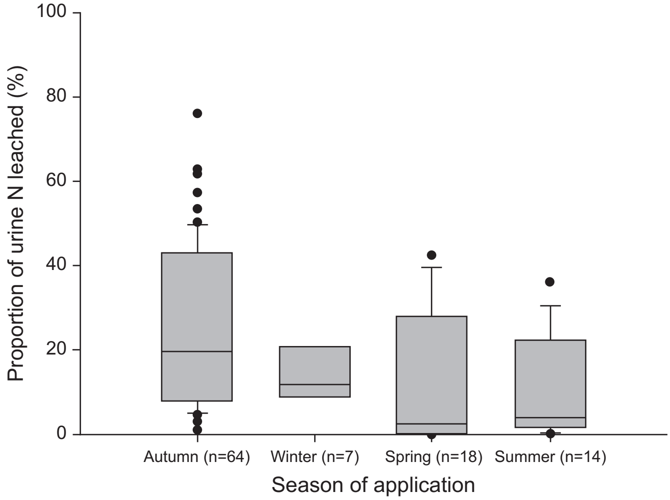


**Figure 4** Summary of $NO_3$–N leaching losses by season of application, data from Table 5. Losses are reported as a percentage of the urinary N applied. The horizontal line is the median, and upper and lower limits of box represent the 75% and 25% percentiles. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, respectively. Outlying points are plotted as closed circles.

#### 3.3.1.2 Deposition Time

Autumn is considered the critical period for urine deposition because in most locations it is typically the start of the main drainage period and plant uptake slows as temperatures decrease. However, urine deposited at other times is also prone to leaching in autumn/winter if N loading exceeds uptake (and other removal processes) during the growing season (Figure 4).

Pakrou and Dillon (2004) and Stout (2003a) measured significant losses of N in winter drainage from urine applied in the previous spring or summer. Decau et al. (2003, 2004) noted leaching of urinary N during winter from summer applications, but not from spring applications. De Klein and Ledgard (2001) concluded that 20% of autumn-soil mineral N was derived from urine deposited in spring/early summer.

#### 3.3.1.3 Irrigation

Leaching from urine patches can also occur at other times of the year if drainage occurs, e.g., from summer storms or surplus irrigation. Carefully managed irrigation during dry summer months can potentially decrease the excess N in a urine patch by increasing pasture growth and N uptake when these might be otherwise limited by lack of soil moisture. However, where irrigation is applied to excess (e.g., "flood irrigation," a relatively common practice until recently in some areas of Australia and New Zealand), this increases summer drainage (Di and Cameron, 2002a) and N leaching from urine patches. Pakrou and Dillon (2004) compared N leaching from irrigated and unirrigated lysimeters. Although irrigation water was only applied in summer months, this was sufficient to reduce the water deficit of the soil profile and increase the annual volume of leachate (and N leaching from the urine patch). Differences between the two treatments were greater in drier years. Wang (2008) determined that increased N leaching was caused by transport of $NO_3$–N in greater drainage from the urine patch.

### 3.3.2 Typical N Losses

#### 3.3.2.1 Nitrate–N

Table 5 summarizes reported measurements of $NO_3$–N leaching at the urine patch scale collated from 22 experiments (103 data points). Average $NO_3$–N leaching loss was 20% of applied (median value 15%). Most experiments focused on autumn applications (Figure 4). Losses were reported at a range of depths. Urine N application rates were in the range 300–1604 kg N $ha^{-1}$ (the upper value representing urine overlap). Forty percent of urine applications were >900 kg N $ha^{-1}$ and only 2% were ≤300 kg N $ha^{-1}$. Statistical analysis of these combined data using a mixed model demonstrated a significant effect of season of application ($P < 0.05$) and lysimeter depth ($P < 0.05$) on the proportion of N leached as $NO_3$–N. Model-predicted values for the proportion of $NO_3$–N leached in each application season, assuming average depth and drainage, were as follows: autumn 24%; winter 20%; spring 17%; summer 16% of applied N. Leaching from spring and summer urine applications was particularly variable, however, with reported values of <5% in half of the measurements. Although the model-predicted (and mean) loss was 24% of autumn-applied urinary N, reported values ranged between 1 and 76%.

#### 3.3.2.2 Ammonium–N, Urea–N

Although N is leached mainly as $NO_3$–N, losses of $NH_4$–N and urea–N can be significant under some circumstances. Menneer et al. (2008a) measured losses of 109-kg $NH_4$–N from a urine application (775 kg N $ha^{-1}$) due to macropore flow and high rainfall/drainage, albeit the lysimeters were only 450-mm deep. Shepherd et al. (2010) measured >30 kg urea-N $ha^{-1}$ leached below 700-mm deep (1000 kg N $ha^{-1}$ applied) in the first drainage

Table 5 A summary of reported $NO_3$–N leaching from individual urine applications

| Reference | Method*/ Depth (mm) | Country | Application time (s) | Urine rate (s) (kg N ha$^{-1}$) | Drainage (mm) | $NO_3$—N leached (kg N ha$^{-1}$) | (%) | No. of treatments |
|---|---|---|---|---|---|---|---|---|
| Cameron et al. (2007) | L 500 | New Zealand | Aut. | 700 | 486—635 | 133—306 | 19—44 | 3 |
| Di and Cameron (2002c) | L 700 | New Zealand | Spr., Aut. | 1000 | 300—600 | 397—502 | 40—50 | 2 |
| Di and Cameron (2004) | L 1200 | New Zealand | Aut. | 1000 | 280 | 85 | 9 | 1 |
| Di and Cameron (2005) | L 1200 | New Zealand | Aut. | 1000 | 300 | 134 | 13 | 1 |
| Di and Cameron (2007) | L 700 | New Zealand | Aut. | 300—1000 | 300 | 60—255 | 20—27 | 3 |
| Di and Cameron (2012) | L 700 | New Zealand | Aut. | 1000 | 550 | 629 | 63 | 1 |
| Di et al. (2002) | L 700 | New Zealand | Aut. | 1000 | 240 | 77—90 | 8—9 | 2 |
| Di et al. (2009) | L 700 | New Zealand | Aut. | 1000 | 335—683 | 68—457 | 7—46 | 5 |
| Fraser et al. (1994) | L 1200 | New Zealand | Wint. | 500 | ~500 | 40 | 8 | 4 |
| Menneer et al. (2008b) | PC 600 | New Zealand | Aut., Wint. | 598 | 215—410 | 86—263 | 14—44 | 2 |
| Menneer et al. (2008a) | L 500 | New Zealand | Aut. | 775 | 348 | 114 | 15 | 1 |
| Moir et al. (2010) | L 300 | New Zealand | Aut. | 300 | 200 | 147 | 49 | 1 |
| Sprosen et al. (2009) | PC 600 | New Zealand | Aut. | 600 | 440 | 275 | 46 | 1 |

(*Continued*)



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**Table 5** A summary of reported $NO_3$—N leaching from individual urine applications—cont'd

| Reference | Method*/ Depth (mm) | Country | Application time (s) | Urine rate (s) (kg N ha$^{-1}$) | Drainage (mm) | $NO_3$—N leached (kg N ha$^{-1}$) | (%) | No. of treatments |
|---|---|---|---|---|---|---|---|---|
| Zaman and Blennerhassett (2010) | L 400 | New Zealand | Aut., Wint. | 600 | NR | 52—81 | 9—14 | 2 |
| Decau et al. (2003) | L 900 | France | Aut., Spr., Sum. | 525 | 65—110 | 1—45 | 1—9 | 9 |
| Decau et al. (2004) | L 900 | France | Aut., Spr., Sum. | 525—825 | 85—143 | 1—57 | 1—7 | 17 |
| Dennis et al. (2012) | L 1000 | Ireland | Aut. | 521 | 374—677 | 5—28 | 1—5 | 6 |
| Pakrou and Dillon (2004) | L 1000 | Australia | Wint., Spr., Aut. | 650—1604 | 310—360 | 25—335 | 3—21 | 7 |
| Wachendorf et al. (2005) | L 800 | Germany | Aut. | 1030 | 239—416 | 494—592 | 48—57 | 2 |
| Silva et al. (2005) | L 700 | New Zealand | Aut. | 1000 | 271—322 | 77—181 | 8—18 | 6 |
| Welten et al. (2013a) | L 600 | New Zealand | Aut. | 600 | 768 | 217 | 36 | 1 |
| Stout (2003a) | L 900 | The United States | Aut., Spr., Sum. | 442—1327 | 240—504 | 69—530 | 16—76 | 18 |
| Shepherd et al. (2014) | L 700 | New Zealand | Aut. | 1000 | 610—1338 | 153—619 | 18—62 | 12 |

*L = lysimeter, PC = porous ceramic cup, NR = not recorded.
Aut. = autumn, Spr. = spring, Wint. = winter, Sum. = summer.

events after application, again attributing this to macropore flow. Most studies used zero-tension lysimeter studies, which might overestimate losses via macropore flow. Silva et al. (2000) detected movement of both $NH_4$–N and urea-N below 700 mm in urine-treated lysimeters but loss of these N forms was eliminated by applying a 0.5-kPa suction to reduce macropore flow.

$NH_4$–N can be leached if the soil is unable to retain it in the upper soil layers. Wachendorf et al. (2005) attributed $NH_4$–N leaching losses of 88–138 kg N $ha^{-1}$ below 800 mm (1030 kg N $ha^{-1}$ applied) to the sandy soil's low CEC, combined with delayed nitrification. Qian and Cai (2007) observed large $NH_4$–N leaching losses from soils with low CEC and low base saturation.

#### 3.3.2.3 Dissolved Organic N

There are few reported measurements of DON leaching after urine application, and it is often unaccounted for or given little attention in agricultural simulation models, despite its potential importance (van Kessel et al., 2009). Wachendorf et al. (2005) measured DON leaching losses of 72–181 kg N $ha^{-1}$ after urine application (1030 kg N $ha^{-1}$) to 800-mm deep lysimeters. Welten et al. (2013a) measured DON losses of 82 kg N $ha^{-1}$ below 600 mm on a sandy soil (600 kg N $ha^{-1}$ applied). van Kessel et al. (2009) surveyed 16 studies and calculated the average DON losses across all treatments and sites to be 12.7 kg N $ha^{-1}$ per year, with a general trend for DON leaching to increase with increasing N inputs, particularly urine application. Urine deposition introduces soluble N directly to the soil and the soil pH rise in the urine patch will increase the solubilization of soil organic matter, further increasing soil DON (Wachendorf et al., 2005). The frequency of urine deposition events and therefore stocking rate can have a considerable impact on DON leaching losses.

### *3.3.3 Management of Leaching*

Dietary manipulation is a common strategy to decrease urinary N excretion and N loads within the urine patch. This includes reduction of N intake (e.g., Petersen et al., 1998) and the use of tannin-containing feeds to decrease urea-N concentration. Salt added to the diet can increase water intake and urination volume (Dijkstra et al., 2013), thus diluting the N load per urine patch. Ledgard et al. (2007) showed that by increasing the salt (NaCl) intake of nonlactating dairy cows consuming grass silage with 0, 200 to 400 g NaCl per day, the urination volume increased from 5 to

12 to 18 L per cow per day, respectively, and the urine N concentration decreased from 10 to 4 and 3 g N $L^{-1}$, respectively.

Other management strategies focus on the urine patch itself. Shepherd et al. (2010) used a carbon additive to immobilize N and decrease $NO_3$–N leaching. Nitrification inhibitors, by slowing nitrification, can decrease $NO_3$–N leaching (and $N_2O$ emissions) from urine patches (Stark and Richards, 2008). Research has mainly focused on use of nitrification inhibitors combined with fertilizer N or animal wastes, but inhibitors can decrease $NO_3$–N leaching when applied directly onto recently deposited urine. Dennis et al. (2012) measured reductions in $NO_3$–N leaching from urine patches of 38–42% under Irish dairy farm conditions after application of the nitrification inhibitor DCD. de Klein et al. (2010) reported a decrease in $NO_3$–N leaching from urine of 33 ± 9% for New Zealand experiments from wet, mild climates and 63 ± 9% for experiments in cooler, drier climates. This sensitivity to environment makes sense given that microbial degradation of DCD will be more rapid at higher temperatures. DCD is also highly mobile in water (Shepherd et al., 2012), which may result in separation of DCD from the $NH_4$–N source (Abdel-Sabour et al., 1990).

## 3.4 Nitrogen Immobilization

### 3.4.1 The Process and Factors Affecting Immobilization

Immobilization is the process by which inorganic N is converted to organic N. This is often termed "microbial immobilization" as almost all the consumed N is utilized by, and incorporated into, microbial cell and tissue constituents (Jansson and Persson, 1982). Plant uptake, residue turnover, and clay mineral fixation are variants of immobilization and, although not usually included in definitions of immobilization, they may influence the estimation of gross process rates. The opposing process is mineralization, or the mobilization/release of organic N into inorganic N forms. Mineralization and immobilization occur simultaneously in soil (Cameron, 1992), termed "mineralization–immobilization turnover" (Jansson and Persson, 1982). The difference between the two *gross* processes will be the *net* process.

Generally, it is assumed that the C:N ratio of an input to the soil results in either net mineralization (C:N < 20:1) or net immobilization (C:N > 20:1). Urine deposition, with a C:N ratio of ~2:1, should result in net mineralization. Following urine deposition, there is a flush of soil microbial activity, causing (net) immobilization (Holland and During, 1977). Between 9 and 20% of N can be immobilized within the first 24 h following urine application (Keeney and Macgregor, 1978; Williams and Haynes, 1994).

This is thought to be due to the sudden availability of labile C sources, which soil microbes use for respiration. Lovell and Jarvis (1996) found no specific effect of urine application on soil microbial biomass N or C contents, but instead observed a >50% increase in microbial respiration. Moreover, urinary N immobilization of 38% was reported by Thompson and Fillery (1998) from a soil with a low C content (<1%). Solubilization of C following urine addition to soil was shown by Lambie et al. (2012) where 25% of urine-C was degraded following incubation of soil cores at 25 °C for 28 days. The same study also found a positive priming effect (C solubilization) with real urine, and a negative priming effect with artificial urine. Monaghan and Barraclough (1993) showed solubilization of C in urine-affected soil, which they attributed to the high pH conditions following urea hydrolysis. These findings indicate substantial changes in the soil conditions following urine application (Haynes and Williams, 1992).

### *3.4.2 Typical N Removal and Management of Immobilization*

Two general types of study have been used to estimate N immobilization in pastoral soils, both of which use $^{15}N$ tracer techniques: (1) N balance (e.g., Clough et al., 1998) and (2) gross N transformation studies (e.g., Murphy et al., 2003). Table 6 summarizes N balance studies where immobilization was estimated using method 1, discussed above. The average N immobilization was 26% of applied N (range 10–63%). The length of the experiments was between 22 days and 2 years. Immobilization was attributed primarily to microbial assimilation, since plant uptake was measured in most cases. However, immobilization calculated using a $^{15}N$ balance approach usually includes N fixed by clay minerals and soil organic matter, as well as the N recycled in plant residues, depending on the length of the experiment. N immobilization beneath the urine patch generally takes place in the topsoil. Fraser et al., 1994 recovered 79% of immobilized N in the top 20-cm soil. Clough et al. (1996) recovered the majority of the 10–23% immobilized N in the top 5-cm soil.

Immobilization usually peaks within 4 weeks following urine application. Pakrou and Dillon (1995) observed decreased N immobilization with time, from 53% of N applied 7 days after urine to 14% after 84 days. Williams and Haynes (2000) measured 30–40% and 20% of urine N immobilized in the first 2 weeks in long- and short-term pastures, respectively. For the next 30 weeks, the proportion of urine N immobilized was relatively constant between 20 and 30%, and showed a slight decline between 40 and 50 weeks, to approximately 20%. Condon et al. (2004) reported a

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**Table 6** Summary of studies reporting N immobilization[*] from urine applied to soil

| Reference | Location | Soil type | Trial type | N rate (kg N ha$^{-1}$) | Soil C (%) | Immobilization (%) | N uptake (%) | Trial length (days) |
|---|---|---|---|---|---|---|---|---|
| Ball et al. (1979) | New Zealand | Silt loam | Field plots | 300–600[§] | n.d. | 30–35[#] | 22–37% | 53 |
| Bronson et al. (1999) | Australia | Sand | Lysimeter | 205[§] | n.d. | 16[‖] | n.d. | 28 |
| Carran et al. (1982) | New Zealand | Silt loam | Field plots | 300[§] | n.d. | 35–50[#] | 15–20 | 130 |
| Clough et al. (1996) | New Zealand | Silt loam, peat | Lysimeter | 500[¶] | 7.7–48 | 10–23[‖] | 11–35 | 150 |
| Clough et al. (1998) | New Zealand | Silt loam, peat, sandy loam, clay | Lysimeter | 1000[¶] | 4–28 | 21–24[‖] | 22–31 | 406 |
| Condon et al. (2004) | Australia | Sandy loam | Pot trial | 510[§] | 1.5 | 20–25[#] | n.d. | 32 |
| Decau et al. (2003) | The United States | Clay loam, loam, silt loam | Lysimeter | 520[§] | 1.1–2.2 | 25–31[‖] | 37–49 | 730 |
| Fraser et al. (1994) | New Zealand | Silt loam | Lysimeter | 500[¶] | 2.6–3.1 | 20[‖] | 43 | 365 |
| Holland and During (1977) | New Zealand | Sandy loam | Field plots | 400[¶] | 5.3 | 19[#] | 30–40 | 42 |
| Jensen et al. (1999) | Denmark | Loamy sand, sandy loam | Lysimeter | 184–194[§] | 1.4–1.7 | 54[‖] | 35 | 730 |
| Keeney and Macgregor (1978) | New Zealand | Silt loam | Lysimeter | 362[¶] | n.d. | 13[‖] | 25 | 22 |

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| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Leterme et al. (2003) | France | Silt loam | Field plots | 549–675[§] | n.d. | 21–31[‖] | 30–65 | 350 |
| Pakrou and Dillon (1995) | Australia | Sandy loam | Field plot and lysimeter | 1090–1370[§] | n.d. | 14–63[‖] | 3–20 | 80 |
| Saarijärvi and Virkajrvi (2009) | Finland | Regosol/ medium texture | Field plots | 500[§] | 0.01 | 30[#] | 19 | 330 |
| Shepherd et al. (2010) | New Zealand | Silt loam | Lysimeter | 500[§] | 6.5 | 19[‖] | 48 | 247 |
| Sorensen and Jensen (1996) | Denmark | Sand, sandy loam | Field plots | 204[§] | 0.6–1.3 | 14–35[‖] | 51–62 | 150 |
| Thompson and Fillery (1998) | Australia | Loamy sand | Field plots | 123–259[§] | 0.7 | 25–34[‖] | 47–53 | 365 |
| Wachendorf and Joergensen (2011) | Germany | Fine sand | Field plots | 1030[§] | 4.3 | 13–17[‖] | n.d. | 730 |
| Whitehead and Bristow (1990) | The United Kingdom | Clay loam | Field plots | 740[§] | 3.3 | 21[‖] | 19 | 321 |
| Williams and Haynes (1994) | New Zealand | Silt loam | Field plots | 230–290[§] | 3.3–3.5 | 21–22[‖] | 12–19 | 29 |
| Williams and Haynes (2000) | New Zealand | Silt loam | Field plots | 770[§] | 3.3–3.6 | 20–35[‖] | 35–50 | 350 |

n.d. not determined.
[*] "Immobilization" includes N assimilated by microbes, fixed by clay minerals and humic complexes or similar, and excludes plant uptake.
[§] real urine.
[¶] artificial urine.
[‖] $^{15}$N balance.
[#] other method.

similar trend, with immobilization rates equivalent to 20–25% of applied N between 3 and 32 days after urine application. Recently immobilized N under urine patches can remineralize (Williams and Haynes, 1997, 2000), which has implications for the length of experiments and timing of measurements. Studies reporting changes in immobilization rates with time are rare and, instead, a percentage recovery in the organic N fraction is reported at a single point in time, usually the end of the experiment.

The forms of immobilized urine N may change over time. Whitehead and Bristow (1990) noted that $^{15}N$ recovery fluctuated widely in all fractions in the first 50 days following urine application. Microbial biomass N peaked initially (<50 days) followed by a slow decline. Humified organic matter N recovery peaked at approximately 120 days after urine application, followed by a trough at 190 days. In total $^{15}N$ recovery terms, the fractions followed the order: humified organic matter > microbial biomass > macro organic matter > roots.

There has been little information reported to date on the effect of urine on N immobilization, except as a component of a $^{15}N$ balance (Table 6), which provides an estimate of the *gross* amount only. An estimate of *net* immobilization would provide far greater insight into the overall effect of urine on soil N. This area represents a significant research gap in pastoral systems.

Manipulating the C:N ratio in the soil may provide an opportunity to increase immobilization as a potential N loss mitigation for urine patches. Shepherd et al. (2010) reported soil $^{15}N$ recoveries in the range 27–51% of urine N applied in urine + sucrose treatments, which was higher than the recovery of 19% of urine N applied in the urine + sawdust treatments. The higher immobilization was thought to be due to the sucrose being a more readily available C source for microbial assimilation and there was less N leaching as a result.

## 3.5 Pasture Nitrogen Uptake

### 3.5.1 The Process and Factors Affecting N Uptake

Pasture plants primarily use inorganic N, both $NH_4^+$ and $NO_3^-$ forms. However, $NO_3^-$ must be reduced back to $NH_4^+$ once inside the plant, thus requiring higher energy inputs and thereby reducing the efficiency of plant N utilization (Haynes, 1986). Because $NO_3^-$ is more mobile than $NH_4^+$ in the transpiration stream, and because nitrification quickly depletes the soil of $NH_4^+$, $NO_3$–N is often more available for uptake by plants. Plants can also take up some organic N compounds directly via their root systems, or in association with certain types of mycorrhizal fungi (Nasholm et al., 1998; Hodge et al., 2000; Harrison et al., 2007). Plants can also absorb N (primarily gaseous $NH_3$) through their leaves (Sommer and Jensen, 1991; Whitehead, 1995), but this contributes <5% of total plant N uptake.

Nitrogen uptake by pasture is of course influenced by temperature, sunlight hours, soil moisture, and the amount of plant available N in the soil. The optimum temperature for perennial ryegrass (*Lolium perenne* L.) is in the range of 15–18 °C with a minimum of 5 °C and a maximum of up to 35 °C (Whitehead, 1995). Longer sunlight hours allow for greater plant N uptake due to greater capacity for photosynthesis. Factors specific to pasture N uptake from urine patches are discussed below.

#### 3.5.1.1 Edge Effects

As previously explained, the urine patch comprises a wetted area and the area immediately outside the wetted area. For example, Decau et al. (2003) used $^{15}N$-labeled urine patches (wetted area of 0.4 $m^2$) and measured pasture recovery of urinary N as ~20 g N $m^{-2}$ in the wetted area and ~5 g $m^{-2}$ from the edge area. The pasture urinary N uptake outside the wetted area was attributed to soil N diffusion; urinary N did not diffuse beyond 200 mm from the edge of the urine patch. Deenen and Middelkoop (1992) found that the effective area was confined to within 150 mm from the edge of the urine patch. Measurements of affected pasture on a volcanic soil similarly suggest that the effective area extends c.150 mm beyond the wetted area of the urine patch (Shepherd et al., 2014). Buckthought (2014) applied $^{15}N$ labeled to an area 60 cm in diameter (0.28 $m^2$) and recovered 30% of the $^{15}N$ in the wetted area and 17% within 25 cm of the edge of the wetted area. Moir et al. (2011) observed seasonal variation in the pasture response to urine whereby the urine-affected areas tended to be larger from spring/summer deposited urine, and smaller in winter and autumn. These differences were mainly attributed to rapid winter $NO_3^-{-}N$ leaching when soils are draining (less N available for plant uptake) but other factors may have included animal N intake in feed (less feed in winter), animal water intake (less water ingested in winter and therefore lower urine volumes), and higher spring/summer pasture growth rates (Moir et al., 2011).

Pasture in the edge area can therefore play an important role in removal of urinary N from the urine patch. Data from Decau et al. (2003) suggest that uptake could increase by c.25% when the edge contribution is accounted for. Based on data from Buckthought (2014), an extra c.50% of urinary N was recovered in the outer zone compared with the wetted area. This questions the effect of lysimeters on estimating N leaching losses when urine is applied over the entire surface and pasture outside of the lysimeter is unable to access this N.

#### 3.5.1.2 Urine Scorch

Urine "scorch" reduces the potential for removal of N from the urine patch. It occurs when the high N content and temporary rise in soil pH during urea hydrolysis causes ammonia toxicity which, coupled with the high salt content of the urine, has a detrimental effect on plant roots (Richards and Wolton, 1975). Generally, the higher the N concentration and ionic strength of urine, the larger the extent of pasture scorching (Groenwold and Keuning, 1988). Recovery of pasture from urine scorch can take up to 10 months (Dale, 1961; Deenen and Middelkoop, 1992). Another consequence of urine scorch is the gradual ingress of weed species and the deterioration of the sward quality (Keuning, 1980).

### *3.5.2 Typical Pasture N Uptake and Opportunities to Improve Uptake*

Pasture growth usually visibly increases, mainly due to the added N and K and the visible effect is usually present for about 3 months (Whitehead, 1995). Nitrogen is further enhanced by "luxury uptake" when soil mineral N concentrations are high. The growth response is dominated by the grass component of the pasture, as clover is a poor competitor for the available N, having a depressing effect on the clover content (Ball et al., 1979; Haynes, 1981; Di and Cameron, 2002b). Assessing "typical" amounts of urinary N uptake by pasture is affected by the risk of "priming" where urine application might stimulate even larger amounts of plant uptake from the background soil N pool. We have therefore restricted our analysis of urinary N uptake to where $^{15}$N-labeled urine was applied.

Plant response to urine is greatest where growth is less likely to be restricted by environmental conditions such as low soil moisture (i.e., summer) or temperatures (i.e., winter). For example, Decau et al. (2003) measured pasture recovery of urinary $^{15}$N as 58, 42, and 32% for spring, summer, and autumn urine applications, respectively. Clough et al. (1998) applied urinary N at 1000 kg N ha$^{-1}$ in winter and recovered 21–34% of applied $^{15}$N, depending on soil type (this includes $^{15}$N in roots but this amount was <1%). Di et al. (2002) applied 1000 kg N ha$^{-1}$ in autumn and recovered 29–39% of applied $^{15}$N in above-ground pasture and another

20% in roots. Fraser et al. (1994) applied 500 kg N $ha^{-1}$ and recovered 44% in pasture. Management interventions can alter the pasture N recovery. Silva et al. (2005) recovered 39% of $^{15}N$ in above-ground pasture from urine (1000 kg N $ha^{-1}$) applied in autumn, which decreased to 29% when dairy effluent (400 kg N $ha^{-1}$) was applied as well. An additional 20% was recovered in roots in both treatments. Shepherd et al. (2010) applied 500 kg N $ha^{-1}$ as urine and recovered 48% of $^{15}N$ in pasture; this decreased to 19% when sucrose was applied to the soil to enhance immobilization of urinary N. With uptake being as much as 50% of applied N, this represents a major removal process from the urine patch.

#### 3.5.2.1 Pasture Species

Most work to date has focused on pasture comprising perennial ryegrass/clover, but other pasture species might be more effective at removing N, either through improved root architecture (Crush et al., 2007) or growing more actively in winter. Supporting this theory, Malcolm et al. (2014) measured 24–54% less N leaching from Italian ryegrass/white clover pasture than from other pasture species they compared (including perennial ryegrass/white clover). Dry matter yield and winter N uptake were greater from the Italian ryegrass/white clover pasture and it was argued that this pointed to high plant winter activity (plant growth/root metabolic activity) as being more important than specific root architecture.

#### 3.5.2.2 Process Inhibitors

Process inhibitors aim to decrease N losses and therefore it might be expected that N saved from loss could be utilized by the pasture. Di and Cameron (2002c, 2004) measured significantly greater N uptake (and dry matter yield) of pasture from urine-treated lysimeters when the nitrification inhibitor DCD was applied. Menneer et al. (2008a) recovered 20% of applied $^{15}N$ from an autumn urine application (775 kg N $ha^{-1}$) but this increased to 27% where urease and nitrification inhibitors were added to the urine to decrease both $NH_3$ volatilization and $NO_3^-$ leaching. Shepherd et al. (2014) found a positive correlation between N saved from leaching by DCD application and increased pasture N uptake in a lysimeter study (i.e., equivalent to urine patch level).

#### 3.5.2.3 Urine Spread

Nitrogen losses can be exacerbated when urinary N load exceeds pasture requirement. Thus, diluting urinary N concentration, for example, by the use of dietary salt supplementation would benefit pasture N utilization by decreasing the N load per urine patch. Bryant et al. (2007) estimated that improved urine spread could decrease N leaching by as much as 48% in their case study.

# 4. SCALE ISSUES IN UNDERSTANDING THE EFFECTS OF URINE PATCHES

The implications for farm-level N management result from the aggregated effects of all urine patches deposited on the farm. This provides challenges in quantifying (including modeling) these aggregated effects, as well as developing mitigations that can operate within a farm to decrease N loss from the urine patch while not decreasing farm productivity or profitability.

## 4.1 Spatial Distribution of Urine Patches

### *4.1.1 Scaling to the Paddock Level*

Urine patch coverage in a paddock depends on urine patch area (driven by urination volume) and urine patch frequency (driven by stocking rate and feed and water intake) (Pleasants et al., 2007). Estimates of coverage range from 4% to 29% of the grazed pasture area per year (Richards and Wolton, 1975; Williams and Haynes, 1994; Whitehead, 2000; White et al., 2001; Dennis et al., 2011; Moir et al., 2011) although quantitative measurement is difficult. Improved methods are now available for measurement of urine patch distribution. Moir et al. (2011) used a real-time kinematic global positioning system (RTK-GPS) and estimated the mean number of urine patches annually deposited on a dairy farm to be 6240 (±124) patches $ha^{-1}$ (range 1300–2100). Using the same method, Dennis et al. (2011) reported 0.359 urine depositions per grazing hour, covering 14.1–20.7% of the soil surface annually. Visual assessment of the urine patch coverage (the "pasture response area") will overestimate the actual area receiving urine, i.e., the "wetted area." The use of "urine sensors" combined with GPS provides information on spatial and temporal distribution of urination events (Betteridge et al., 2010b, Betteridge et al., 2013).

Distribution is uneven across a paddock or land unit. Animal congregation sites such as sheltered areas, around water troughs, gateways, on ridges or hills, and in areas where hay or silage are fed out, receive higher urine loads than other areas (Haynes and Williams, 1993a). A hill country study showed that 55% of the sheep urine was deposited in only 15–31% of the total land area (Saggar et al., 1990). In another study in hill country, 50% of beef cows' urination events were within 5–15% of the paddock area, which corresponded to low elevation, low slope areas (Betteridge et al., 2010a). Grazing management further modifies this distribution. Rotational grazing (animals grazed at very high stocking rates for short periods to improve feed utilization, especially in winter) increases urine patch overlap and N leaching risk, compared with grazing animals at lower stocking rates for an extended period (Pleasants et al., 2007).

#### 4.1.1.1 Estimating Paddock-Scale Urinary N Deposition

The total amount of urinary N excretion is generally estimated through either empirical (regression) or mechanistic animal metabolic modeling, linking per animal or herd dietary intakes to N excretion (Hirooka, 2010). These approaches allow an estimate of urinary N load deposition at a range of spatial or temporal scales when linked to data about animal movements and management.

Romera et al. (2012) linked a mechanistic urinary N excretion model with a model predicting urinary volume to then estimate the number of, and N concentration in, urine patches deposited on a paddock as a prior step to calculating $NO_3$–N leaching risk. More simply, daily urinary N excretion has been related to daily N intake either as curvilinear (Castillo et al., 2000) or linear (Vellinga et al., 2001) functions and also to milk urea–N and dietary crude protein (CP; Spek et al., 2013).

Amounts of annual urinary N deposition to the paddock can be large, with 125–250 kg N ha$^{-1}$ returned directly to the sward in urine (Whitehead, 1986). Lantinga et al. (1987) (750 cow grazing days ha$^{-1}$, 16 kg DM intake per day per cow) estimated c.250 kg N ha$^{-1}$ returned to the paddock as urine. The OVERSEER® Nutrient Budgets model (Selbie et al., 2013) can estimate the monthly urinary N excretion of a ruminant herd based on dietary N intake, CP content, and production (meat, wool, or milk). Figure 5 provides an example of the calculated monthly urinary N deposition across a block of land stocked at 3.2 dairy cows ha$^{-1}$, grazed all year round and fed predominantly ryegrass/white clover pasture (spring calving; Sept/Oct in southern hemisphere). This equates to c.350 kg N ha$^{-1}$ deposited as urine, but the pattern is highly seasonal with the least returned in winter and most in spring/early summer when the cows are lactating and consuming more high-quality pasture. This seasonality has consequences for scaling up individual urine patch N losses to the paddock and farm level; even though $NO_3$–N leaching losses can be large

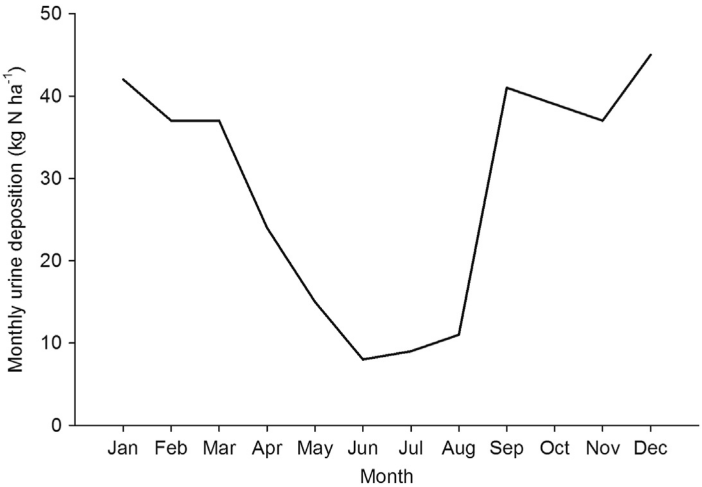


**Figure 5** Example of the annual distribution of urinary N return to a block of land stocked at 3.2 cows ha$^{-1}$ under New Zealand conditions, based on calculations using the OVERSEER model (Wheeler et al., 2008).

from individual urine patches deposited in autumn and winter, this is a period when less N is deposited.

#### *4.1.2 Scaling to the Farm Level*

Urine is also deposited on raceways, stock handling facilities in the milking yard, and in animal housing. The more time animals are in these "nonproductive" farm areas, the less urine is directly voided onto soil, although much will be captured as effluent, slurry, or manure and redistributed around the farm this way (Ledgard, 2001). The proportion of excreta deposited in these nonproductive areas will depend on the farm system but has been estimated at 10–38% of the total excreta for dairy farms (Nguyen and Goh, 1994). The mobility of the grazing animal means that it facilitates the transfer of nutrients and fertility around the farm. Whole-farm models (WFMs) need to be able to account for these transfers, as discussed below.

### 4.2 Modeling at the Paddock and Farm Scale

There are many process-based models available that mechanistically model N. However, in order to specifically model N dynamics at the urine patch scale, simulation of preferential flow and two-dimensional movement of water and solutes should be included (Wang, 2008). It is also important to take account of characteristics of the urine patch that differ, say, from a fertilizer application; movement of N to depth during deposition and lateral access to nutrients by pasture outside the wetted area.

#### 4.2.1 Paddock Scale

Preferably, the approach for estimating paddock-scale urine N deposition is to aggregate the effects in an individual urine patch, to give an explicit representation of the heterogeneous urine deposition, yet many models assume uniform spatial return (Snow et al., 2009). Assuming a uniform spatial return of urine deposition may underestimate the N-removal process in question. Aggregating urine patches to the paddock scale requires some estimate of the impacts of overlapping. Cichota et al. (2013) identified two contrasting circumstances for overlap: (1) when urine patches are deposited within short time intervals resulting in an additive effect and (2) when the interval between depositions is longer and urine patches become functionally independent. Furthermore, the estimates of N leaching losses using mean urine patches, derived from mean urine volume and N concentration, differ from that estimated using variable urine patches as deposited by animals (Li et al., 2012).

#### 4.2.2 Farm Scale

Routine direct measurement of N losses is impractical at a farm level. Simulation models with a strong evidential basis are the best alternative and their use for assessing potential N losses has been increasing worldwide (Cichota and Snow, 2009). Such models can simulate combinations of management and environmental conditions that cannot be easily attained by experimentation, and thus help to better understand how environmental conditions combined with management strategies affect both farm productivity and its environmental impact (Vogeler et al., 2013).

A key task of farm-scale models is to be able to model transfers of nutrients around the farm. As such, there is less emphasis on the processes within urine patches. For example, the nutrient budgeting model OVERSEER models N transfers around the farm (Selbie et al., 2013), and uses a summarizing "transfer function" (Cichota et al., 2012) to estimate the proportion of N leached from urinary N deposited each month. In contrast, Romera et al. (2012) developed a urine patch framework that postprocesses the results of a WFM (Beukes et al., 2010) and runs a mechanistic soil model to simulate the urine patches, including an estimation of spatial distribution of the patches. The difference in complexity of approaches to modeling urine patch effects at the farm scale reflects their end use: OVERSEER is an on-farm decision support tool, whereas the WFM is primarily a research tool.

## 5. MANAGING URINE PATCH NITROGEN IN THE FARM SYSTEM

Numerous papers have assessed farm system N management options to decrease losses from grazed pastures (Rotz et al., 2005; Ledgard et al., 2009; Misselbrook et al., 2013; Monaghan and de Klein, 2014). Mitigations either target the source of N or the transport of the N, as shown in Table 7. Most mitigations focus on managing the source of N. Table 7 also includes an assessment of their stage of development, i.e., whether or not they have been proven to work at the farm system level.

### 5.1 Target: Soil

Process inhibitors (urease and nitrification inhibitors) control $NH_3$ volatilization, $N_2O$ emissions, and $NO_3$–N leaching; a decrease in N loss at the individual urine patch scale has been demonstrated (Di and Cameron, 2004, 2008; de Klein et al., 2011). Urease inhibitors have not been applied at the paddock scale. In contrast, the nitrification inhibitor DCD has been applied to grazed pastures in New Zealand. Monaghan et al. (2009) measured a reduction in N leaching of 21–56% from two to three applications made during autumn–winter over the whole paddock at an experimental farmlet scale. In general, measured effectiveness on N leaching and pasture N response has been variable at this scale, and further work is required to understand this. Broadcasting inhibitors is inefficient given the target is recent urine depositions, which represents only a small proportion of the paddock. Targeting stock camp areas could decrease the required application area (Betteridge et al., 2011). Welten et al. (2013a,b) suggest orally administering DCD to grazing animals, which is then excreted with the urine. Another type of additive is a C-containing compound. Shepherd et al. (2010) used soil-applied C additives to increase N immobilization and decrease N leaching from a urine patch. Efficacy depended on C availability (sucrose better than sawdust) but demonstrated proof of concept. However, there are practical difficulties in adapting this approach to a paddock/farm scale.

### 5.2 Target: Plant

Altering some key traits in forage plants would potentially decrease N losses but most are still at an experimental stage. The concept of a highly

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**Table 7** List of mitigation options with potential to decrease N losses from urine patches

| | | Mechanism | | | Development stage* | | |
|---|---|---|---|---|---|---|---|
| **Target** | **Mitigation description** | **Source** | **Transport** | **Reference** | **Con** | **PoC** | **PoF** |
| Soil | Process modifiers: | | | | | | |
| | Nitrification inhibitors | ✓ | ✓ | Monaghan et al. (2009) | | | ✓ |
| | Urease inhibitor | ✓ | | Zaman et al. (2009) | | ✓ | |
| | N immobilization | ✓ | | Shepherd et al. (2010) | | ✓ | |
| Plant | Grass—composition: | | | | | | |
| | Low N | ✓ | | Ledgard et al. (2009) | ✓ | | |
| | High sugar, high tannin | ✓ | | Edwards et al. (2007) | | ✓ | |
| | Rooting habit | | ✓ | Malcolm et al. (2014) | | ✓ | |
| | Winter active | ✓ | | Nichols and Crush (2007) | ✓ | | |
| | Pasture species mix | ✓ | ✓ | Woodward et al. (2012) | ✓§ | ✓¶ | |
| | Biological nitrification inhibition | | ✓ | Subbarao et al. (2009) | | ✓ | |

(*Continued*)

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**Table 7** List of mitigation options with potential to decrease N losses from urine patches—cont'd

| | | Mechanism | | | Development stage* | | |
|---|---|---|---|---|---|---|---|
| **Target** | **Mitigation description** | **Source** | **Transport** | **Reference** | **Con** | **PoC** | **PoF** |
| Animal | Feed additives | | | | | | |
| | Tannin | ✓ | | Waghorn (2008) | | ✓ | |
| | Salt | ✓ | | Dijkstra et al. (2013) | | ✓ | |
| | DCD | | ✓ | Welten et al. (2013a) | | ✓ | |
| Farm | Restricted grazing | ✓ | | de Klein and Ledgard (2001) | | | ✓ |
| | N fertilizer strategy | ✓ | | Ledgard et al. (1999) | | | ✓ |
| | Stocking rate | ✓ | | Ledgard et al. (1999) | | | ✓ |
| | Use of low-N-supplementary feeds | ✓ | | Tomlinson et al. (1996) | | | ✓ |
| | Move water troughs, etc. | ✓ | | Shepherd and Chambers (2007) | ✓ | | |

* "Con" = Concept "PoC" = Proof of concept; "PoF" = Proof of function, proved to work within a farm system.
§N leaching decreased due to deep rooting.
¶N leaching decreased due to decreased dietary N.

winter-active grass with greater N demand and uptake during times of high leaching activity appears a possibility (Nichols and Crush, 2007) but has not moved beyond this concept. Breeding low N grass has the potential to reduce urinary N excretion (Ledgard et al., 2009) but, as yet, there do not appear to be any suitable low-N productive grasses. An effective nitrification inhibitor has been discovered in the root exudates of a tropical pasture *Bracharia* spp., but further work is required to exploit this function (Subbarao et al., 2009). Snow and White (2013) modeled various plant characteristics and their effect on N leaching risk. Rooting depth was identified as moderately important; the most important factor of those tested was the ability of the plant to intercept light at low pasture mass, thereby increasing the potential for increased growth and N uptake giving increased growth and N uptake. Crush et al. (2007) demonstrated that the rate of plant growth and size of the root system were important regulators of nitrate interception in ryegrass genotypes, and suggested that this supported the emphasis on deeper rooting as a strategy to increase nitrate interception.

An alternative strategy could be to use a pasture species mix that provides some of these features. Malcolm et al. (2014) have demonstrated that Italian ryegrass/clover benefits N leaching because of improved winter growth compared with perennial ryegrass. Although milk yields from cows grazing a diverse pasture (perennial ryegrass, white clover, prairie grass, lucerne, chicory and plantain) and a standard ryegrass pasture were similar (Woodward et al., 2013), in-stall feeding trials showed a halving of urinary N excretion from feeding the species mix (Woodward et al., 2012). Further work is required to determine if the inclusion of deeper rooting species such as chicory and lucerne would also intercept more of the deposited urine.

Grasses with high water-soluble carbon (WSC) content can improve dietary N utilization by ruminants with a resultant decrease in urinary N, although trial data have produced mixed results (Edwards et al., 2007); these authors suggested that the benefits of high WSC content could be negated if there is also a high CP content.

## 5.3 Target: Animal

Mitigation focuses on feed management or additives. Oral administration of DCD has already been described. The potential for salt to act as a diluter of urinary N has also previously been described. Polyphenols, specifically condensed tannins (CT), have been demonstrated to reduce N absorption in the rumen and partition more excretal N to feces (Woodward et al., 2004). However, as with high-sugar grasses, effects are inconsistent. Forages naturally rich in CT (e.g., sainfoin) also perform less well agronomically than grass. Also, if the tannin content is too high, dry matter intake can also be reduced (Mueller-Harvey, 2006).

### 5.4 Target: Farm Management

Farm-level management tackles the urine patch indirectly but the analysis in Table 7 shows that these are mostly "farm-ready" and have been implemented on experimental farms or are used in commercial agriculture. Monaghan and de Klein (2014) provide estimates of effectiveness of many farm-level strategies. Growing or importing low-N-supplement feed such as cereals can decrease urinary N excretion (Tomlinson et al., 1996), although the consequences associated with growing these crops should also be accounted for in a life cycle analysis (Ledgard, 2009). Removing grazing stock at key times (autumn/winter) reduces direct urine deposition and can decrease N leaching by 35–50% (de Klein and Ledgard, 2001) and can decrease soil $N_2O$ emissions. However, increased losses of $NH_3$ may negate some of these benefits (Webb et al., 2005). Decreased stocking rate decreases N leaching (Ledgard et al., 1999), through decreased forage intake (and urinary N excretion); decreasing dairy cow replacement rate is one way of decreasing stocking rate. Decreased N fertilizer inputs can decrease N leaching (reduced pasture production) but can also decrease productivity. The effect of N fertilizer is mainly through the extra amount of forage grown and urine subsequently excreted (Shepherd and Lucci, 2013).

## 6. CONCLUSIONS

The urine patch is the conduit through which much of the N in grazed pasture systems passes. Its contribution to farm N cycling increases with the more time that animals spend on pasture. The focus of this chapter was on N, as a major nutrient required for plant growth and because of its potential impacts on the wider environment. Most of the published research has focused on cattle-based systems, predominantly dairy. This reflects the economic importance of, but also the challenges in, efficient N management in these production systems. The chapter set out to address three questions: (1) what is known about urine patch characteristics and the N cycling processes in the patch; (2) what are the implications for N cycling at the farm and paddock scale; and (3) what mitigations are there to control losses from the urine patch?

The chapter confirms that urine patch characteristics (urine N concentration, urination volume, frequency of urinations per animal, and urine patch area) are highly variable. Few new data have been published on urinary N characteristics since the review of Haynes and Williams (1993b). Urine patch N loading rate is a key metric for quantifying and modeling N cycling through the urine patch. However, it is a derived value with no direct measurement method available, and therefore relies on estimates of the highly variable urine patch characteristics. A wide range of N loading rates has been used in published experiments (300–1604 kg N $ha^{-1}$ in our survey), which partly reflects the high variability, but perhaps also the uncertainty in the measurement and calculation of urine N loading rate.

Differentiating between the wetted and effective areas of a urine patch is important. Quantifying the wetted area allows calculation of the N deposited in a urine patch, whereas quantifying the effective area allows an assessment of the area where pasture N uptake occurs. The effective area is visually noticeable and easier to measure than the wetted area. However, while it enables an estimate of the proportion of the paddock affected by urine, it cannot be used as an assessment of the paddock area *receiving* urine. This has important consequences for scaling up from individual urine patch to paddock scale.

The urine patch processes of $NH_3$ volatilization, $N_2O$ emissions, N leaching, and pasture N uptake have been well researched. The main research gaps relate to quantifying the amounts and source processes producing $N_2$, DON leaching, and N immobilization. For $N_2$, this gap is mainly due to the measurement difficulties inherent with a high atmospheric background of $N_2$. DON leaching from urine patches has been quantified in some published studies but more data are required. Immobilization is confounded by the simultaneously occurring process of mineralization, and there are few studies which measure these processes from urine-affected soil. More research is required to understand both the consequences of these little-investigated processes for the urine patch N budget and the consequences of urine N loss into the environment.

This chapter has highlighted the difficulty in deriving "typical" values for the fate of urinary N due to the influence of individual urine patch characteristics and environmental factors. Average $NH_3$–N losses are c.13%, but losses can range from 1% to 38%, and can exceed 38% in extremely hot and dry conditions. Average $NO_3$–N leaching losses were c.20–24% from autumn/winter depositions and 16–17% from spring/summer depositions (overall average 20%). Nearly all mineral N was leached as

$NO_3$–N. The overall average pasture N uptake was 41%, based on the $^{15}N$ studies we reviewed. Most of the data were from autumn- or winter-applied urine. Broadly, the two major N-removal processes were pasture N uptake and N leaching, and the balance between the two drives the N leaching profile, particularly in the drainage season. A third major source of N removal was immobilization. However, estimates of immobilized urinary N are confounded by mineralization occurring simultaneously with immobilization. Gross N immobilization beneath a urine patch was based on $^{15}N$ studies, from which the immobilization rate was 26% of the applied urinary N. Nitrous oxide emissions tended to be a small component of the urine N budget (average $N_2O$ loss of 2.1% with a range of 0–14%), although environmentally they are important as a greenhouse gas.

The sum of the average for each removal process, $NH_3$ volatilization (13%), $N_2O$ emissions (2%), N leaching (20%), N immobilization (26%), and pasture N uptake (41%), is equal to 102%. Interestingly, this value is close to 100% which could imply a near-complete recovery of applied N. However, this value should be interpreted with caution. The sum total is likely to be higher when other, unaccounted for pathways are measured, such as the contribution of native soil N mineralized in the urine patch. The use of isotopic $^{15}N$ balance methods may provide further insight into the source of N, either applied or native N, assuming that added N interaction effects are taken into consideration.

Empirical approaches to scaling from urine patch to paddock generally use "typical" N removals from the urine patch (lysimeter/small plot study) multiplied by the area or number of urine patches. Urine patch area can be misleading in this calculation. Area tends to be the effective area of the urine patch, but it is the wetted area that is needed for scaling up (i.e., the area that receives urine). This method also requires some inclusion of temporal variability and it also assumes that urine patch characteristics used for scaling up are representative of the animal for the time of deposition.

The variability in urine patch characteristics adds complexity to scaling up from individual patch to paddock, whether it is by using empirical approaches as described above or by the use of models. Although variability will add uncertainty to estimates, say, of N leaching, a critical appraisal of the effects of this variability on estimates is required. For example, Li et al. (2012) estimated that N leaching could be underestimated by 5–8%, when using an "average" urine patch as opposed to varying both volume and N concentration; and that the underestimation would be more for pastures on soils of greater water holding capacity. Further analyses like this are required to fully understand the sensitivity of our modeled estimates of paddock/farm-scale losses to variable urine patch characteristics.

The impact of urine patch overlap on measured and modeled N leaching losses is also important and becomes more so at high stocking rates, or where animals are rotationally grazed, or where they congregate in large numbers in areas within a paddock (stock camp areas). Modeling suggests that not taking account of overlap will underestimate N losses.

Spatial and temporal variation in urinary N deposition in a paddock also provides practical challenges for managing N. The use of nitrification inhibitors to control $NO_3$–N leaching at the paddock level is one example. Effectiveness is high when measured in a lysimeter/small plot scale, equivalent to an individual urine patch applied at a single time. However, in reality, urine patches are deposited in a paddock throughout the year. The solution is to target urine patches that are high contributors to N loss, i.e., patches that are deposited in autumn and winter for N leaching. More than one application of N inhibitor may be required. This illustrates that progressing experimentally from proof of concept to proof of function is a critical step in the evolution of improved N management strategies.

Models are an important tool for scaling from the urine patch to the paddock and to the farm. These models are multidisciplinary, whereby they deal with the animal as the generator and distributor of urine and then the biological and physical processes once the urine "hits the ground." However, farm-scale models are as much about accounting for nutrients transfers around the farm as they are about being able to model urine patch N dynamics.

Given the importance of the urine patch to N loss and inefficiencies in grazed pasture systems, mitigating losses is a priority. The assessment of management that specifically targets the urine patch suggests that there is a large number available. However, our analysis suggests that many are still at the proof of concept stage, with few actually deployed on the farm. Those that are, tend to involve removing stock from pasture at key N loss vulnerable periods (i.e., fewer urine patches voided directly), combined with system efficiencies around per-cow performance or feed and fertilizer N inputs. A further challenge (both at the individual urine patch and at the farm scale) is the avoidance of pollution swapping when trying to control one N loss pathway. This marks the need to measure more N cycling pathways at a range of scales when investigating mitigations.

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