NITROGEN IN WASTEWATER AND ITS ROLE IN CONSTRAINING ON-SITE PLANNING

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Abstract

The requirement to minimise the impact of domestic effluent on public and environmental health is an important component of best management practice for on-site wastewater management. The numerous guidelines prepared by state and local regulators recognise the importance of adequate distribution and assimilation of nitrogen from domestic wastewater. As this element is an essential microbial and plant nutrient there are many pathways for reducing it in the effluent while preventing its loss off-site. Many on-site wastewater management guidelines seem singularly driven by the need to prevent the assimilation of nitrogen on the basis of poor science, rather than its assimilation using best management practices and current research data.

This paper reviews the current literature with respect to the nitrogen in domestic wastewater, the sources and possible pathways from product purchase through household use/consumption and the treatment train into the soil application area. Many of the sources of this element are from both the human diet and human metabolism and reduction at source is mostly unachievable. Reduction in other areas will be examined.

The outcome of the review is to compare and contrast the disparity of views with regards to its significance in imposing unrealistically large areas for assimilation of nitrogen back into the environment. Is the concern about nitrogen based upon poor science or is the risk to human health justified?

Keywords

denitrification, nitrogen, nutrient assimilation, nutrient balances

1 Introduction

1.1 Nitrogen References in Guidelines

In any nutrient balance for land application of wastewater or treated wastewater (effluent), the considerations of preventing nitrogen from contaminating groundwater are often given priority over the potential impact of other wastewater contaminants such as sodium. In NSW, the EPA (1995) states that "*from an environmental perspective, nitrate is the most critical form of nitrogen. Its solubility, mobility and stability mean that it is readily leached to groundwaters, it has an active role in the eutrophication process and, in drinking water, it poses a threat to human and animal health*". This statement takes poetic licence in that it is unclear as to why nitrate is most critical. Nitrate is not stable as it is readily adsorbed by plants and microorganisms and immobilised as part of their protein, and it can be reduced to nitrous oxide, nitric oxide or nitrogen gas through denitrification. If nitrate was highly mobile, then the accumulation of nitrate under fallow as suggested by Leeper and Uren (1993) would not occur. The relationship with human and animal health is quantitative, while nitrite is known to be more toxic at similar concentrations as later discussed in detail.

The EPA classifies 'low strength' effluent as having a total nitrogen (TN) concentration of less than 50 mg N/L, which is five times the threshold for drinking water. These guidelines also state that "*to ensure long term sustainability, the total nitrogen applied to the site from effluent and mineralisation should balance the nitrogen uptake of the vegetation cultivated*". When this statement is accepted as fact (without reference to current research), in the absence of an understanding of the total nitrogen cycle, and applied by regulatory authorities to on-site wastewater management, the size of the application area required for nitrogen assimilation expands to often ridiculously large areas. The statement also avoids any suggestion that nitrogen can be stored in the soil in forms that are not readily leached. Anaerobic soil conditions can quickly trigger denitrification and gaseous loss of N (Tisdale *et al*., 1985).

The Environment and Health Protection Guidelines (DLG, 1998) suggest a nominal nitrogen loading of 25 mg/m^2/day (91 kg/ha/year) be applied to land application systems. When this loading rate is compared with typical plant and microbial uptake rates, it is difficult to understand the scientific basis for the guideline value. While the value is called 'nominal' the author has had some authorities impose this as the maximum loading rate. The value is simply the mean of the range of TN for uptake values of perennial pasture referred to by EPA (1995) as 65-130 kg/ha.yr, which references a NSW Agriculture 1991 Feedlot Manual.

The revised Feedlot Manual (NSW Agriculture, 1997) indicates that for an irrigated perennial ryegrass pasture, growing actively March to December, the expected nitrogen uptake rate is 420 kg/ha.yr. Kikuyu is expected to remove 520 kg/ha.yr. Removal of the aerial portion of the grasses is required to remove the nitrogen from the application area. It would follow, although not discussed in the guidelines, that a further quantity of nitrogen would be stored in the root system as organic nitrogen, in the microbial biomass and leaching of nitrogen would be restricted to only a portion of the nitrate-N. Which of the values is accurate and why is there no uniformity between agencies?

The NSW Recycling Guidelines (NSWRWCC, 1992) simply defers to EPA (1995) when making reference to use of recycled water for landscape applications. However, there is no indication that the home owners using recycled water are in any way limited to the quantities of water they use to maintain lawns and gardens. Is nitrogen in the urban area from recycled water not a limitation on its application rate?

1.2 Reported Case Studies

Thompson (2000) states that microbial biomass is both the catalyst of decomposition and an important source/sink of carbon and inorganic nutrients, especially nitrogen. He also tabulates data to show that under a continuous wheat crop the amount of soil nitrogen is 2700 kg N/ha (270 kg N/m^2) while the nitrogen in biomass is 95 kg N/ha. As perennial pasture provides all-year growth, rates at least similar to a wheat crop would not be unexpected.

In the FILTER technique of using flood irrigation and subsurface drains to strip out nutrients from municipal wastewater, Biswas *et al.*, (1999) reported that their CSIRO Griffith (southern NSW) trial removed 57% of the total nitrogen after filtering through 1 m natural soil. For the rye grass crop, 182 kg N/ha was removed while the water percolating through the FILTER had less than 10 mg N/L (drinking water level). In a FILTER trial at Gatton (SE Qld), 90% of the TN load was removed in a seven day treatment. Nitrate levels in the water collected from under the FILTER system were usually $\leq$ 5 mg N/L (Gardner *et al*., 2000).

Whitehead *et al*., (2001) in their review of five case studies, showed that there was no consistent association between nitrate and bacterial contamination in either surface or groundwater, nor did they find a clear correlation between level of contamination and on-site system density. Cromer (2001) showed that while nitrate levels increase close to the trenches, that downgradient of the trenches' levels of nitrate fell to less than 1 mg N/L.

Cromer did measure each of the three nitrogen species (NH_4^+, NO_2^-, NO_3^-) in the groundwater but organic N (Norg-N) has to be derived from his data by calculation. From the data it has been calculated that for the 12 bores over three sampling events, the Norg-N was from 100% to 1% of the TN load with a median 70%. It would be difficult under such circumstances to correlate the TN in the bores to septic drainfields as Norg-N is not readily transported in the same manner as NO_3-N.

Under perennial crops and pastures, Kuhn *et al*., (2001) suggest that soil nitrate levels will normally be low, as nitrate is taken up by plant roots as it is mineralised and there is little opportunity for it to accumulate in the soil. ANZECC & ARMCANZ (2000) state that the TN applied to the soil in irrigation water should balance the N uptake of the harvestable portion of the crop plus the acceptable concentration in drinking water (23 mg N/L). This does not appear in either the EPA (1995) guidelines or the DLG (1998) guidelines. It is unclear as to why the N applied in irrigation water should only balance the plant uptake.

2 Nitrogen in Drinking Water

Much of the public health concern about nitrogen in drinking water is based upon an incident of methaemoglobinaemia (blue-baby syndrome) reported by Comly in 1945 ("L'Hirondel & L'Hirondel, 2002) describing recurring episodes of cyanosis in two infants after ingestion of water containing large amounts of nitrates. Research has since shown a poor correlation between the nitrate content of drinking water and infant methaemoglobinaemia, but a strong correlation with unhygienic well water. Currently nearly 22% of the cultivated land in the EU has nitrate above 50 mg/L as nitrate, without a link to methaemoglobinaemia ("L'Hirondel & L'Hirondel, 2002).

It is interesting to note that the Australian Drinking Water Guidelines (ADWG) (NHMRC, 1996) specifically state that:

> *"the toxicity of nitrate to humans is thought to be solely due to the reduction to nitrite. The major biological effect of nitrite in humans is its involvement in the oxidation of normal haemoglobin to methaemoglobin which is unable to transport oxygen to the tissues. This condition is called methaemoglobinaemia. Young infants are more susceptible to methaemoglobin than older children and adults".*

L'Hirondel & L'Hirondel (2002), medical researchers, state that "WHO, US and EC regulations on nitrate in potable water and food are not supported by science. They should be re-examined" and concluded that "the history of nitrate is that of a world-scale scientific error that has lasted for more than 50 years. The time has come to rectify this regrettable and costly misunderstanding". The document goes on to indicate the natural beneficial effects of nitrate in humans and its insight into these effects is worth reading. If this text is only partly correct, our present regulations are ridiculously conservative.

The ADWG (NHMRC, 1996) give a health value for nitrate of 50 mg as nitrate (11.3 mg N/L) and for nitrite of 3 mg/L as nitrite (0.9 mg N/L). The 50 mg/L as nitrate has been set to protect bottle fed infants under three months of age as discussed previously. The guidelines further complicate the issue by stating that up to 100 mg/L as nitrate (22.5 mg N/L) can be safely consumed by adults and children over three months of age. Reference to research by Avery (1999), cited in ANZECC & ARMCANZ (2000) suggests that upon re-examination of methaemoglobinaemia in infants, 100 mg NO_3^-/L (23 mg N/L) would probably not increase the health risk to infants. Why do the guidelines not reflect the references they use?

The ANZECC & ARMCANZ (2000) guidelines point out that the ADWG have not set a health-based guideline for ammonia even though ammonia can be readily oxidised to nitrite then nitrate, both of which the guidelines consider a health risk. The ammonium concentration in irrigation water would be accounted for as part of the TN concentration under the ADWG.

So, why do we still have a 50 mg NO_3^-/L (11.3 mg N/L) limit on drinking water, and why is this level expected to leach to groundwater from land application of irrigation water? Why is nitrate labelling of bottled water neither voluntary nor mandatory.

3 Basis for Nitrogen Loading Rates

The rate of NO_3^- uptake by plants is usually high and occurs by active absorption (Tisdale *et al*., 1985), the NO_3^- entrained with water entering the roots. The NH_4^+ ion is the preferred nitrogen source since energy will be saved using it instead of NO_3^- for synthesis of protein. Ammonium is less subject to losses by leaching and denitrification (neutral pH, depressed by acidity) because it can be captured by the soil's cation exchange capacity or can replace potassium ions (K^+) within the clay lattice.

The ANZECC & ARMCANZ (2000) guidelines have no cumulative contaminant loading limits (CCL) for nitrogen because "nitrogen is a major plant nutrient". The short term trigger value (STV) of nitrogen in irrigation water, which applies to a 20 year period, is set at 25-125 mg/L as nitrate (5.6–28.2 mg N/L) to minimise the risk of contaminating water.

From where did the regulators get the 10 mg N/L which is now different to ADWG of 11.3 mg N/L (50 mg NO_3/L) or the statement by Avery (1999) that a concentration of 22.6 mg N/L (100 mg NO_3/L) may not affect infants as previously thought? If nitrogen is so critical to water quality, why has there been as upward shift of 13% in the threshold level of nitrogen in the current rounding operation between two sets of guidelines. One could be excused for thinking that the numbers were 'comfortable positions' rather than guesses.

4 Sources of Nitrogen in Home

Most of the nitrogen in domestic wastewater is the product of our eating habits and food preparation, body exudates washed off in the bath or shower and products washed from clothes. Cleaning chemicals also contribute organic compounds in varying amounts. These organic compounds require microbial activity to degrade them.

Food preparation including washing of vegetables with small vegetable scraps entering the wastewater system through the sink, adding to the potential nitrogen load in the septic tank. Table 1 indicates the nitrogen contribution to our diet from selected foods. Whether the contribution is from food scraps, waste from plates and dishes or what becomes human waste and the by-products of metabolism, the relative values in Table 1 reflect the significant input from our diet. In most cases it will be unlikely that one would expect dietary changes in the house because of the potential impact of nitrogen on the land application area. It is possible, though, through education and better management of kitchen operations to reduce the waste food scraps and oils that enter the wastewater stream.

L'Hirondel & L'Hirondel (2002) state in northern Europe, beetroot, spinach and lettuce can have levels of nitrate higher than 1000 mg/kg (226 mg N/kg), water cress 610 mg N/kg, with peas and potatoes less than 22 mg N/kg. The authors quote references to show that in USA 87% of the nitrate content of the human diet comes from vegetables, compared to 60% for UK and 78% for France, the remainder coming from water. As the level of nitrate in vegetables is inversely proportion to sunlight, it would be interesting to note the variations for Australian vegetable crops.

Table 1. Sources of nitrogen as proportion of protein in common foods

Food source	g N per 100 g portion	Food source	g N per 100 g portion
beef and lamb (lean meat, cooked)	4.48	peas (fresh or frozen)	0.8
cheese	4.16	Eggs	1.92
Beans (haricot, cooked)	1.12	bread	1.25
cornflakes	1.38	potatoes (cooked)	0.26
rice (cooked)	0.35	apples	0.05
milk (full cream)	1.0	bottled water	not listed

(after Wahlquist, 1986)

In the bathroom wastewater is contaminated with perspiration (sweat) that contains neutral fats and volatile fatty acids, traces of albumen, urea [$CO(NH_2)_2$], sodium chloride, potassium chloride and traces of alkaline phosphates, sugar and ascorbic acid (Osol, 1973) calcium and magnesium salts and nitrogenous compounds (organic N, ammonia-N, urea and amino acids), the latter which vary with diet. Other human wastes include skin, hair, body oils and greases and the 'dirt' from other sources. Hair shampoo, conditioners and other personal care items contain large proportions of very complex organics, the fate of which is unclear.

The toilet is the repository for a large proportion of the nitrogen that enters the domestic wastewater stream. Faeces are excretions from the intestines of unabsorbed food, indigestible matter and intestinal secretions (Osol, 1973). Excreted nitrogen is a waste product of protein metabolism (Wahlquist, 1986) and the daily loss is about 160 mg N/kg body weight/day. Urine is excreted by the kidneys at about 1.0–1.5L/day, depending upon liquid intake and has about 40-75 g solids. The solids are mainly 25% urea [$CO(NH_2)_2$], 25% chlorides, 25% sulphates and phosphates and the remainder organic acids, pigments, neutral sulphur and hormones (Osol, 1973). Urea is 46% N rapidly reverts to NH_4^+ and bicarbonate ions in soil.

The laundry is perhaps the least contaminating source for nitrogen. Laundry powders contain very low levels of organic products.

The question arises: how do we reduce the level of nitrogen that enters the domestic wastewater while maintaining healthy eating habits and a high level of personal hygiene?

5 Nitrogen in Septic Tank Effluent

The various species of nitrogen in a single household septic tank vary with the number of occupants, their wastewater generating behaviour, and the biological activities in the tank, including the degree of accumulation of sludge and scum. Patterson *et al.* (2001) reported that for four sampling periods from five septic tanks in Tingha (northern NSW) NH_4-N was 36.1 – 84.1 (avg 57.1) mg N/L; NO_3-N was 1.2 – 5.1 (avg 2.8) mg N/L; total Kjeldahl nitrogen (TKN) was 44.2 – 103.5 (avg 67.1) mg N/L; with a TN of 46.3–108.2 (avg 69.9) mg N/L. The measurement of TKN in water accounts for both the ammonia-N and the Norg-N. By subtraction of ammonia-N from TKN, one is left with the Norg-N.

The variability in TN in any one system can be high. At Tingha, the least variable was 39-83 mg N/L and the highest variability was 55-182 mg N/L. At all times the NO_3-N in the septic tanks was low and the NH_4-N high because of the anaerobic conditions while the Norg-N had a median value of 19% of TN.

Sarac *et al*., 2001 measured TN for 20 homes and reported values for septic tanks of 50–340 (avg 141) mg N/L and for the trenches to those systems of 55-217 (avg 90) mg N/L. Septic tanks are mineralising systems (converting proteins to ammonia) and because of anaerobic conditions, ammonia is favoured over nitrate. The levels of TN are unlikely to be abated from septic tank to trench. The decreased TN in the trench is mainly through denitrification in an anaerobic environment.

However, this concentration of TN should not be used to gauge trench effectiveness, since the trench is only a storage facility – treatment is performed in the adjacent soil profile.

Martens and Geary (1999) measured nitrogen in existing drainfields. They reported values of plus and minus one standard deviation of 30 - 152 mg N/L of ammonia (avg. 91.1 mg N/L), <1 – 5.8 mg N/L of nitrate and nitrite combined (avg. 2.2 mg N/L) and 44- 172 mg N/L of TN (avg. 108 mg N/L). By subtraction the Norg-N averaged 14% of the TN. Since these values are for water in the trench, there was no suggestion that these levels would be found in the soil immediately adjacent to the drainfields.

6 Measurements and reporting

The laboratory methods for the measurement of each species of nitrogen in ANZECC & ARMCANZ (2000) and EPA (1998) refer to Standard Methods (APHA, 1995). In the relevant tests, the reporting of each nitrogen species is in mg/L of nitrogen.

The ANZECC & ARMCANZ (2000) confuse the reporting of nitrogen as either nitrate rather than nitrate-N. In Table 4.2.1.1 of the guidelines the tabulated values are for nitrogen with no reference to the various species. At section 4.3.3.3, nitrate less than 400 mg/L in livestock drinking water should not be harmful to animal health. At the end of the section the statement "*Note that trigger values in the present guidelines are expressed as nitrate and nitrite*" gives new confusion to the whole of the guidelines. If the latter statement is true, then levels of up to 90.3 mg N/L are acceptable in livestock drinking water. As livestock can be sheep, cattle or horses, it is clear that animal mass does not affect the dose. There is, however, no measure of TN (ammonia, nitrite, nitrate and organic-N). The toxicant guideline level for aquaculture is 50 000 mg/L as nitrate (11300 mg N/L).

Why is there no standardisation of reporting level of mineral nitrogen in water, that is mg N/L (as NO_3-N) or mg NO_3/L. ANZECC & ARMCANZ (2000) states that "*under the assumption that all forms of nitrogen have the potential to be expressed as nitrate in soil, total nitrogen has been used for setting trigger values*". This is not clear throughout the guidelines and the reporting of nitrogen becomes more confusing.

7 Levels of nitrogen in soil

Organic matter (OM) contains the ratio C:N:S:P as roughly 100:10:1:1 (Probert, 1988). Most of the nitrogen in OM is present as amino acids unavailable to plants and does not leach out. Since the heterotrophic organisms use organic carbon as energy they consume the OM and form fresh OM, competing with plants for the available nitrogen. Autotrophs obtain energy from oxidation of inorganic salts and their carbon from CO_2 in the atmosphere.

Monnett *et al*., (1995) reported that maintaining reclaimed water in the upper microbially active part of the soil profile was important to denitrification. They also stated that denitrification is enhanced by anaerobic conditions and greater nitrate concentrations in that microbially active zone. A supply of NO_3^- or NO_2^- in soil is a prerequisite for denitrification.

High NO_3^- increases the rate of denitrification and exerts a strong influence on the ratio of nitrous oxide to elemental nitrogen in the gases released from soil by denitrification. This finding as it contradicts the processes by which N is accounted for in nutrient balances.

The direction that nitrogen takes in mineralisation or demineralisation is dependent upon the carbon:nitrogen ratio (C:N). If the decomposing OM has a high C:N ratio (above 30:1) the microbes will utilise the NH_4^+ and NO_3^- present in the soil to continue decomposition and mineral nitrogen will not be available to the plants. When C:N is low (below 20:1), there will normally be a release in mineral nitrogen in the soil that is available for plants and microbes.

Large populations of denitrifying organisms are present in arable soil and most numerous in the vicinity of plant roots. Carbonaceous exudates from actively functioning roots are believed to support the denitrifying bacteria in the rhizosphere. The potential for denitrification is immense in most field soils but conditions must arise which cause these organisms to shift from aerobic respiration to a denitrifying metabolism, involving use of NO_3^- as an electron acceptor in the absence of oxygen. Waterlogging is sufficient to cause this change of state. Denitrification can operate in seemingly well-aerated soil, presumably in anaerobic micro-sites where biological oxygen demand exceeds supply.

Levels of nitrogen in soil are generally in the order of > 1000 mg/kg for sandy soils, > 2000 mg/kg for loams and > 2500 mg/kg for clays (Walker and Reuter, 1996). The irrigation of effluent with a TN of 50 mg N/L, when applied at the rate of 6 ML/ha (600 mm per annum), is an increase in soil TN of 12%. The increased application of water alone is likely to increase vegetation mass, increase soil microbial activity and lead to a faster denitrification. Is the intention of the relevant guidelines to suggest that this small addition will cause environmental damage or elevated TN in local groundwater?

8 Conclusions

Two specific guidelines (EPA, 1995 and DLG, 1998) relate to the application of effluent to land, the former for agricultural reuse and the latter for domestic on-site treatment. The requirement for nutrient balances to account for all wastewater nitrogen to be assimilated only by vegetation is seriously flawed. The loss of TN through denitrification appears to be better understood and more widely published than is recognised in the guidelines. The potential for nitrates to leach through a soil profile must be related to soil texture, soil structure, soil permeability and soil depth. Thus, each application area must be judged on its unique soil characteristics rather than a blanket and flawed assimilation rate.

The guidelines introduce confusion, not so the approved methods, to the method of reporting the various species of nitrogen. In each analytical method, whether APHA (1995) (water analysis) or Rayment and Higginson, 1992 (soil analysis) it is clear that the results are reported for nitrogen, that is nitrate as NO_3^+-N, ammonia as NH_4^+-N and TKN as TKN-N. The national guidelines (ANZECC & ARMCANZ, 2000) seriously confuse the issue.

It is regrettable that 11.3 mg N/L (50 mg NO_3^+/N) continues to be used as a restriction on the land application of effluent. Recent research suggests this level is seriously flawed.

The use of nutrient balances based upon flawed guidelines, with inappropriate restrictions on the plant uptake rates, losses through microbial activity and denitrification are hindering best management practice. Through rigid application of these flawed values, irrigation areas are expanded to a point where there is almost no benefit from the nutrients or water. It is time that best management practice is applied to appropriate guidelines.

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