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Hot Topic #8 Reconsidering the role of the bovine urine patch in the groundwater quality debate

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Hot Topics have been conceived to bring New Zealand agricultural and horticultural issues that need debate to the fore in a style and timeliness that cannot be achieved through traditional scientific publication approaches.

1. The dairy cow urine patch has been cited as the source of nitrate pollution of waterways.
2. Re-evaluation of data indicates assumptions in early calculations that are not supported by subsequent research.
3. Attention in this Hot Topic is restricted to the dairy urine patch: a single excretion on to newly grazed pasture.
4. The authors then compare the impact of this process on groundwater with other forms of rural land use and show that the urine patch has negligible environmental impact compared to the importance of dairy production for human nutritional security.

The impact of dairy cows on the environment in New Zealand, particularly the association with groundwater quality, meets the description of a 'Hot Topic'. This was highlighted by the release of the 'Our Fresh Water' report in April 2023 by the Ministry for the Environment *(https://environment.govt.nz/publications/our-freshwater-2023*), which was subsequently (and rapidly) followed by a press release from Greenpeace (*https://www.greenpeace.org/aotearoa/press-release/mfe-report-water-contamination-from-intensive-dairy*) pointing an accusatory finger at dairy farming.

Popular publications from a range of authoritative sources state that dairy cow urination deposits nitrogen at a rate equivalent to 1000 kg/ha of N (e.g., *https://waikatoregion.govt.nz/assets/WRC/WRC-2019/CNM-factsheet-nitrogen-leaching_4-v2.pdf*; *https://www.agmatters.nz/goals/reduce-nitrous-oxide/*), a number that excites the interest of the public at large, and which has driven regulation-setting authorities to focus on dairy industry groundwater impacts to the exclusion of other potential contributors to environmental nitrogen loading. In this "Hot Topic" review, we question the validity of that 1000 kg/ha of N figure.

The fundamental premise is that it is dairy farmers that have caused nitrate (NO^{3-}) levels to increase in groundwater. But have they? Historical groundwater data from the Canterbury Plains indicate localised high nitrate levels having occurred over many decades (Marshall, 1974, 1978; Rutter & Rutter, 2019), and long before the dairy boom commenced in the 1980s (Pangborn, 2012). Further, re-examination of the original source of the 'equivalent of 1000 kg/ha' (Haynes & Williams, 1993) indicates assumption that have not been supported by subsequent research. This suggests that dairy farmers might be being pressured into destocking or changing their farming enterprise completely to stay economically viable, but with no resultant improvement in groundwater.

The problem for policymakers or anyone with an interest, is that groundwater nutrient levels will be an integral of all nutrient leaching in the catchment. In the 1980s, Keeney & Gregg (1982) wrote "high nitrate levels (10 mg NO^{3-}N/litre) have been found in groundwaters in the Waikato and in Canterbury. No attempt has been made to identify the source of this nitrate, although Baber & Wilson suggested that intensively grazed grass-clover swards were a likely source in the Waikato". While correlations between changes in land-use practice and groundwater nutrient levels are observable, correlation is not the same as causation, and ascertaining the major cause (typically causes) of a problem, may require considerable research effort.

A principal cause of groundwater nitrate has been thought to be the cow urine patch, because of the putative nitrogen loading occasioned by that event. This is somewhat peculiar to New Zealand (though Ireland is now following suit), as very few countries graze their cattle outside on pasture for 365 days of the year. Pasture-derived forage typically contains more elemental nitrogen (not the common atmospheric gas N2, but instead in protein and other compounds that contain nitrogen e.g., DNA, and hereafter referred to as N) than a cow requires for maintenance of its metabolism and life processes, and to meet the demands of production (e.g. reproduction, growth, lactation), so surplus nitrogen is excreted, largely in urine patches in liquid form. In other countries, where dairy cows are housed for at least some of the year and fed mixed rations, better control over dietary nitrogen supply to the cow can be obtained than is possible on pasture, and better capture and management of urine and faecal waste can be achieved.

Key elements of the debate

From a human dietary perspective, dairy products can me*et al*most all nutritional needs very efficiently. Of the 29 nutrients considered to be important by the nutritional Delta Model (Smith *et al*., 2022), milk contributes to the global availability of 28, particularly high-quality protein and indispensable amino acids, several vitamins, and minerals such as calcium (Smith *et al*., 2022). Regardless of their qualities though, are dairy products the most efficient way of obtaining high quality nutrition?

The efficiency of production of any food is best determined by the number of people whose nutrition needs are met relative to resources. Milk performs well for land and money use (Coles *et al*., 2016), and water use (deduced from a comparison of the data of Brown *et al*., 2005 with e.g., Coles *et al*., 1997). However, the public domain, and to a lesser extent the academic literature, is littered with assertions that dairy production resource use efficiency is compromised by the "incredibly" (the word used advisedly) high rates of N excretion from cows and consequent loss from the production system. Of late, this has been claimed to be the major cause of both surface and groundwater nutrient pollution, and it inevitably results in the naive assertion that disposal of all dairy cows would alleviate the problem.

The ongoing vocal criticism of the dairy industry has affected public perception, and now, two decades after the launch of the "Dirty Dairying" campaign of the Fish and Game Council, the perception of the polluting dairy cow lives on, be it correct or not. In part driven by public interest, regulations are being implemented across the country with the aim of improving water quality (e.g., *https://environment.govt.nz/acts-and-regulations/freshwater-implementation-guidance/*), including the capping of nitrogenous fertilizer use in agricultural and horticultural production systems (*https://environment.govt.nz/acts-and-regulations/freshwater-implementation-guidance/agriculture-and-horticulture/synthetic-nitrogen-fertiliser-cap-in-place-from-1-july/*). The aim is well-intentioned, and may be effective, but unless dairy cow urine patches are the sole cause of the problem, then reducing or eliminating cows altogether, will likely make little difference.

This Hot Topic considers research undertaken over the last thirty or so years, linking above- and below-ground information to identify the questions that should be asked (and answered) before the dairy industry comes under any more fire. It also considers some of the other likely contributors of nitrate to groundwater to illustrate that the challenge is complex, and that more research is needed.

Rethinking the data on dairy cows and the impact of urinary nitrogen

Nitrogen excretion from dairy cows

Research published in the last few years has examined New Zealand dairy cow nitrogen excretion in detail. Some results from above-ground trials, which have included changing the composition of pasture, are summarised in Table 1.

The amount of N excreted per cow per day varied from a high of 251 g to a low of 98 g, nearly a two-and-a-half-fold difference. The N concentration in urine was also highly variable, ranging from a high of 6.4 g of N per litre of urine to a low of 2.2 g/litre. This is a nearly three-fold difference. The inclusion of plantain in the diet appears to be associated with not only reduced N output per cow, regardless of stage of lactation, but also with reduced urinary N concentration. Yet more variability is found in the volume of urine excreted at each event. The range reported in Table 1 is 2.30 litres to 3.34 litres: a 1.5-fold difference.

Table 1. Dairy cow nitrogen excretion and urination data on pasture and plantain.

Study	Time	Treatment	N Excreted (g/cow/d)	Urination (events/ cow/d)	Volume/ event (l)	N/event (g)	N/litre of urine (g)
Box *et al., 2017*	Early-lactation	Pasture	205	15.5	2.75	13.2	4.7
		50:50	114	11.5	2.82	9.9	3.4
		Plantain	119	18.3	3.09	6.5	2.2
	Late-lactation	Pasture	251	18.0	3.23	13.9	5.4
		50:50	213	18.3	2.87	11.6	3.6
		Plantain	177	20.0	3.34	8.9	2.4
Bryant *et al., 2018*	Late-lactation	Pasture	195	12.1	2.39	16.1	6.4
		Forb mix	187	12.2	2.30	15.3	6.0
Navarette *et al., 2022*	Spring	RGWC	125				3.3*
		PL	110				2.8
		PLCL	124				2.9
	Early summer	RGWC	144				4.3
		PL	142				3.9
		PLCL	163				4.1
	Late summer	RGWC	211				4.4
		PL	154				2.9
		PLCL	163				4.0
	Autumn	RGWC	142				5.5
		PL	98				3.3
		PLCL	132				3.7

*Estimated from graph year one. RGWC = ryegrass and white clover, PL = plantain, CL = clover mix.

Given the variability described in Table 1, and the impact of feed type and season, it isn't surprising that confusion around the impact of urinary N has occurred.

The area covered by a urination

The area of the urine patch upon which the above 2.2 to 6.4 grams of N per litre of urine is deposited, also needs further analysis. The surface tension of urine is similar to that of water (Perryman & Selous, 1935) and of the solution of potassium bromide used by Williams & Haynes (1994) as a tracer (Khurshid *et al*., 2009). On an impermeable surface, water spreads until the depth of the pool falls to 3.6 mm. That means a 3-litre urination would spread outwards to cover about 0.8 m^2, with practical observations from spreading water on pasture suggesting a static depth of 5 mm.

Figures in the literature for area covered by a urine event include 0.38- 0.42 m^2 (a circle of approximately 700-millimetre diameter for an average 0.4 m^2; Williams & Haynes, 1994), 0.34-0.40 m^2 (Moir *et al*., 2010) and 0.2 m^2 (Box *et al*., 2017). A simple calculation from Williams & Haynes (1994) data supports the observation: 2000 mL spread over 0.4 m^2 gives a depth of 5 mm. This implies that the pasture sward and soil below it absorbs the equivalent of 1.6 mm while the liquid is spreading. Thought of another way, what can be absorbed in 20 seconds (the mean duration of a mammalian urination) is only 32% of the total applied, leaving the balance to soak in as the soil is wetted.

Adding to the complications is that the N application rate per unit of area urinated upon might be quite variable depending on soil roughness and the type and extent of pasture cover[1]. Although suggestions have been made about 'splash zones' when larger leaved plants like plantain and chicory are included in pasture, the 'splash' is unlikely to be beyond the zone of uptake. However, an 'edge effect' (extra pasture growth beyond the area of the urine patch) extends 100-150 mm around the wetted area (creating an impact zone of diameter 0.7 m^2), nearly doubling the effective size of the patch. An increase of (up to) 40% in N uptake from urine N deposited in the wetted area, when uptake from the edge of the urine patch is included, has been reported (Phillips & Shepherd, 2013). The size of the zone of uptake beyond the wetted area (i.e. including the edge effect) was not factored into the original research in the 1990s, yet has implications for calculations of N-loss.

The depth of penetration by the urine

Notwithstanding the question mark on urine patch size, the other variables Williams & Haynes (1994) described align with those reported in Table 1. Assuming dissolved bromide ions trace the physical movement of urine through the soil profile, soil coring revealed that bromide penetrated to a depth of 400 mm after the 'urination event'. This was beyond the main rooting zone for the pasture at the time, suggesting that the urine 'water' had gone beyond its potential use by plants. The depth of penetration was suggested to be affected by flow through macropores (Clothier & Heiler, 1983). Given the surface tension of water, these macropores would need to be greater than 3.6 mm in diameter to allow this flow.

Other factors to consider in this experiment, include that it was undertaken on a soil profile maintained at 'field potential', which means that the bromide was simply diffusing down a concentration gradient. However, the diffusion rate of urinary N (it is applied as urea) is subject to other considerations, as it is biologically important in the soil-plant continuum (bromide ions are not), and its mean downwards diffusion rate will therefore be lower (Holland & During, 1977).

A subsequent experiment was undertaken in the late-autumn and winter seasons by Williams & Haynes (1994). These are generally assumed to be the highest risk seasons for nitrate leaching because of the lower temperatures, slower plant uptake of N, and higher rainfall and drainage. The experiment involved placing soil solution samplers at 300- and 600-mm soil depth, and the application of ^{15}N-labelled (radioactively labelled) urine (Williams & Haynes, 1994). They reported that 6% of the applied ^{15}N was leached beyond the 600 mm soil depth, and the total amount of 15N leached was equivalent to 11% of the applied urine-N in typical cattle urine patches.[2] The simplest explanation for this observation is that N absorbed by pasture plants and used for root growth is eventually lost to the soil when new deep roots, having exhausted the available nutrients in the soil volume into which they have grown, senesce, die and are degraded by soil micro-organisms and invertebrates. The N those roots contain is thus released for reabsorption or leaching.

The authors commented that they were surprised the leaching loss was so low, but cited Fraser *et al*. (2013) who recorded a loss of 8% below 1100 mm from an application of urine at a rate of 500 kg/ha N, and Whitehead & Bristow (1990) who measured a loss of 16% below 300 mm from an application rate of 744 kg/ha N. Note that these application rates are several-fold higher than is the case during real urinations, as will be shown below.

Since the biological activity of the pasture plants would be expected to remove a more-or-less fixed amount of soil N when recovering from grazing, the lower provision of N would be reflected entirely in lower N available for leaching. Williams & Haynes (1994) suggested that a combination of ammonia volatilisation, denitrification, microbial N immobilisation and rapid plant uptake might minimise

[1] Urinations usually occur on pasture being grazed, reducing the likelihood that variation in pasture cover will affect the area over which urine will spread.

[2] This suggests that there are soil processes stimulated by addition of urine that drive leaching of non-urine N beyond that occurring under urine-free pasture. This seems improbable

leaching losses in urine patch areas of pasture.

In support of this rapid N-uptake by plants (absorption through their roots as amino acids, nitrate ions, nitrite ions, or ammonium ions), urine is delivered at a time when leaf area index (LAI; the one-sided green leaf area per unit ground surface area) has been markedly reduced by grazing. Good pasture management is focused on restoring LAI as rapidly as possible after grazing. The plant achieves this by using turgor pressure to expand basal cells in leaf tissue and grow new leaves from basal meristem as quickly as possible (Prioul *et al*., 1980). Thus, the extra water provided when urine is excreted will benefit the (grazed) plants on whose roots it is deposited. Expanding leaf area will be accompanied by increasing photosynthetic activity, requiring the plant to recover N from root reserves and from the soil.

Although winter grazing has been considered an extra problem because plant uptake of N is reduced (because of reduced plant growth), the sward being consumed during a winter grazing episode has been grown in situ, and it is logical to assume that readily available soil N has been depleted by that growth. This leaves depleted N-binding loci in the soil which the urine urea will replenish, because urea is strongly bound to charged entities because it forms an electric dipole, much like water. Lower soil temperatures, as will occur in the colder months, will also reduce the rate of urease activity, reducing the relative mobility of the applied urea N.

Of further note is that research (Balvert & Shepherd, 2015) has suggested that the commonly used tool to assess N flux, called lysimeters, potentially underestimate pasture N uptake and therefore potentially overestimate N leaching because 30-40% of the pasture N uptake from urine deposited on the wetted zone come from outside the wetted zone (the zone of influence courtesy of the edge effect). This means that in a lysimeter sized for the urine patch, 30-40% of the potential uptake could be missed.

Taken together, this suggests the contribution of N from dairy cow urine to groundwater is not the 'equivalent of 1000 kg/ha' as is regularly cited and used to argue that the urination of the cattle is the primary cause of elevated groundwater nitrate. The origin of the 1000 kg/ha figure therefore needs to be investigated and its validity tested.

The 1000 kg/ha first appears in Haynes & Williams (1993) and the relevant table from that paper is transferred verbatim below:

The urine concentration at 10 g/L is markedly higher than those reported in the various studies summarised in Table 1 (where it ranges from 2.2 to 6.4 g/L), and the area covered (0.2 m^2) could be less than half that described above (ranges from 0.2 to 0.42 m^2). These are however very important numbers for determining N-loading and hence the potential for leaching to occur. Thus, with a halved urine concentration and potentially doubling of the urine patch area, the application rate would not be any higher than 250 kg/ha N, which is also quite comparable to the 280 kg/ha N that Haynes & Williams (1993) claim is recycled from urine. The effect of the plants in the zone of influence described above is also important (Balvert & Shepherd, 2015), which further reduces effective N fertilisation by an unknown amount.

Taken together, this may have led to at least a four-fold overestimate in application rate per urine patch (1000 kg/ha N versus 250 kg/ha N), and which has subsequently led to investigations with results that are considerably at odds with reality.

Typical Application Rates of Major Nutrients Applied to Sheep and Cattle Urine Patches

Nutrient	Urine concentration (g litre^{-1})	Application rate per urine patch (kg ha^{-1})[a]		Annual output per cattle beast (kg anima1^{-1} yr1)[b]	Annual output from cattle per hectare of farm (kg ha^{-1} yr^{-1})[c]
		Sheep	Cattle		
N	10	500	1000	73	183
S	0.35	18	35	2.6	6.5
K	9	450	900	66	165

[a] Assuming that for sheep and cattle a urination consists of 0.15 and 2.0 litres applied to an area of 0.03 and 0.20 m^2, respectively.

[b] Assuming cattle urinate 10 times per day.

[c] Assuming 2.5 cattle per hectare.

From urine patch to loss into groundwater

Williams & Haynes (1994) stated that 'the overall effect of applying urine to the soil was that leaching loss of N amounted to 11% of that added'. Using the calculated figure of "280 kg/ha N recycled annually in urine on grazed irrigated dairy pasture" (Haynes & Williams, 1993), they estimated that an 11% leaching loss assumed for the whole year, would be 31 kg/ha N from urine patches plus 1 kg/ha N from the area of soil not affected by urine. As drainage and leaching predominantly occurs in autumn and winter, this will be an overestimate of total losses throughout the year.

In addition, the actual loading in the experiment from which the 6% and 5% (direct and indirect summing to 11% loss) were derived was 11.2 g N in 2 litres, yet the studies summarised in Table 1, suggest a range of 4.4 to 12.8 g in 2 litres. Further, the recovery of 280 kg/ha N does not include the N harvested in animal products (cow and calf growth) and the N excreted in faeces. Time during milking, when urine is collected in effluent sumps, is also omitted. Because of the uncertainty in the calculations in the literature (Haynes & Williams, 1993; Williams & Haynes, 1994), going back to first principles is attractive.

At any single grazing, only 2-3% of the paddock receives urine events, hence over a year, 12 grazings would lead to a maximum of 25% of the sward area receiving urine (Moir *et al*., 2010). This study was undertaken at a stocking rate of 4.5 cows/ha at that time and 3.4 is now the norm for the location (DairyNZ Statistics). The reduction in stocking will further reduce the number of urinations per hectare (by about 25%), and thus the impact.

Using the starting figure that each urination event on pasture has approximately 14 g N at a volume of approximately 2.75 litres (Table 1) and using the surface tension calculation explained earlier (and sized up from Williams & Haynes, 1994 – 2 litres and 0.4 m^2) leads to a wetted area per event of 0.576 m^2. That is = 24.3 gN/m^2 N and 243 kgN/ha. The upper end of potential leaching loss is 11% (6% directly and 5% indirectly, Williams & Haynes, 1994) of the N added = 26.7 kg/ha. But if only a maximum of 25% of pasture area is on average urinated on in a year (Moir *et al*., 2010), then the potential leaching loss is only 6.7 kg/ha N per annum. That is markedly less than 1000 kg/ha N.

Other sources of nitrogen

In the context of urinary nitrogen contributing to N loss and the impact on groundwater, one must also briefly consider other potential sources of N.

To start with, the N from urine patches is joining background N loss from soil organic matter activity. To 300 mm depth, New Zealand soils contain on average 100 tonnes of organic carbon per hectare (*https://www.agmatters.nz/goals/maintain-soil-carbon/soil-carbon-science/*). This is associated with approximately 10 tonnes of N (at a C:N of 12 to 8:1). The decomposition of organic matter (and potential for N loss) depends on temperature and moisture, and it is reported that under fertilised moist conditions in New Zealand, half of any added material will have gone in 3.5 to 5 years (Stoner *et al*., 2021). These researchers estimated the mean turnover time for organic matter on fertile, moist pastures to be approximately 15 years, which establishes the potential for a lot of previously accumulated N to be released from soil organic matter.

Soil carbon accumulation has been enabled in some parts of the country following deforestation and the addition of superphosphate that enabled legumes to flourish (Schipper *et al*., 2017). Carbon accumulation has also followed the introduction of dairying on the Canterbury Plains when precision irrigation and the use of nitrogen overcame the limitations to grass photosynthesis and growth and noting that precision irrigation reduces the likelihood of N leaching (Hedley *et al*., 2010).

Milk production also removes N in protein on a daily basis, with this amounting to 112.5 kg/ha/yr N on the Canterbury Plains (based on an average milk solids/ha figure of 1464 kg for North Canterbury; DairyNZ Statistics). Much of the N added to dairy farms is therefore exported, and on the Canterbury Plains the increase in soil organic matter to a depth of 75 mm increases from approximately 3% under sheep and cropping systems, to 5% under dairying (which also creates a 60% increase in soil nitrogen, with added carbon sequestration benefits too).

The loss of N from soil has been estimated at approximately 170 kg/ha/yr N over a 13-year period under bare fallow land (i.e., no plants and no grazing) on the Canterbury Plains (Curtin *et al*., 2021). This N was lost primarily through leaching (Fraser *et al*., 2013). In this respect, an Environment Canterbury report (Aitchison-Earl, 2019) supports the suggestion that arable cropping is a major source of N loss. The implication from this is that removal of dairy cows could increase N loss as the soil returns to a new dynamic equilibrium. Similarly, using the Hurley Pasture Model, Parsons *et al*. (2016) showed that a transition from dairy to dry-stock would decrease food yield, and increase nitrogen release to the environment.

Another source of N is human excretion. In rural areas septic tanks are ubiquitous and the dispersal field must be below the root zone of pasture to achieve resource consent. New Zealand population dietary studies indicate average daily protein intake is 102 g for males and 71 g for females (86.5 g average overall; 31.6 kg/year). This protein is nearly all turned over (i.e. excreted) in the adult population. For the Selwyn District of Canterbury, a region of interest in the context of N-loss into rivers and groundwater, it can be estimated that rural residents now inject 182 tonnes/year of N into groundwater (see Table 2). Selwyn is a rapidly expanding semi-rural/rural region *(https://www.selwyn.govt.nz/news-And-events/news/archived/selwyn-leads-the-nation-in-population-growth-2021*). Historically it was farmed for sheep in mixed-cropping systems, but that has been succeeded by partial conversion to dairy production, as summarised in Table 2.

Table 2. Estimated additions of N to lower vadose zone/groundwater from different land uses in Selwyn District, Canterbury. (Data on area were derived from *https://figure.nz/chart/ZsjUTQ3Hp0I1cDnW-ltXxziVvnyJ5J56D*)

Aside from humans and stock, weedy legumes, including gorse (*Ulex spp.*) and broom (*Cytisus scoparius, Carmichaelia spp.*), can contribute to fixed N in the rootzone. These species are commonly found in un-grazed/unimproved land, especially riverbeds and other waterways. Research undertaken in the Wairarapa suggested that leaching from gorse is probably much higher than currently assumed (Mason *et al.*, 2016). These authors calculated that leaching under gorse was greater than under pasture by a factor of two to three times, and commented that underestimation of this 'invasive' contribution could lead to constraint on agricultural production (because of the erroneous assumption that N-load is caused by agriculture). Many other native species are also capable of supporting nitrogen-fixing microbes in a range of associations (Burrows & Wilson, 2008); and this potential contribution to N-loss is not accounted for in this table.

Burial of deadstock, estimated at 2% of animals (Max Enersen, pers. comm., 2023) will provide N directly below the vadose zone. Dairy cattle in the Selwyn District alone would account for 4500 animals annually, providing 68.4

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Land use		**N per unit (kg/year)**	**Total N loss to groundwater (tonnes)**	**Proportion of total (%)**
Dairy (grazed pasture intercepting urine deposits)	51,536 hectares	6.7[3]	343.5	7.3
Cropping (including dairy support)	40,503 hectares	85[4]	3442.8	73.1
Other grazing	109,411 hectares	6.7[7]	733.1	15.6
People				
Rural population (data courtesy of E. Sibbald, Selwyn District Council)	36,000[6] people	5.081[7]	182.9	3.9
Total (tonnes/annum)			**5115.8**	**100.0**

[3] See the section "*From urine patch to loss from a paddock*" above.

[4] See the section "Other sources of nitrogen" above.

[5] See the section "Other sources of nitrogen" above; uses the dairy figure as worst-case scenario and does not include domestic cats or feral animals as they excrete urine in small volumes.

[6] Estimated from 55% of households connected to a reticulated wastewater system, but relatively smaller household sizes compared to rural areas.

[7] See the section "Other sources of nitrogen" above.

tonnes of elemental N, or about one third of that contributed by human waste disposal. This amounts to about an extra 1.3% of the N losses outlined in Table 2. Note that the contribution of cats, dogs and feral animals, which tend to excrete on the soil surface and in small volumes, is not considered significant.

All this discussion should be set in context with the historical "purity" of water and current sources of contaminants. On the Canterbury Plains ECan data indicate high nitrogen concentration in some wells from sampling in the 1940s. Masterate research in the early 1970s (Marshall, 1978) reports that nitrogen was 3.6 ppm in the Leeston Drain 1 km from the source, yet decreased after passing through watercress beds and increased again after each village (Marshall, 1974). Further, high N levels in waterways have been recorded across the decades (Rutter & Rutter, 2019) and long before the dairy boom. Also reported by Environmental and Scientific Research (ESR; Close & Humphries, 2019) is that a survey across the country found caffeine, human contraceptive chemicals, sunscreen ingredients and artificial sweeteners in many wells. This suggests that human excretion is at least part of the problem.

Measuring a change

There is little doubt that New Zealand is trying to improve waterways and groundwater, just like many other countries. However, identifying the sources of N in a catchment is not easy. A review for the Ministry for Primary Industries, Welten *et al*. (2021) reveals the paucity of measured leaching data, and the range of leaching reported under different production systems. Because the measurements are not easy to make, researchers have tended to use figures already published which, as discussed above, might not have been typical.

The next problem is the measurement in the catchment. A just-published paper (McDowell *et al*., 2024) has uncovered the very real problems for regulators trying to monitor water quality. The research indicates that it can take 4 – 50 years to detect a percentage change in the level of a contaminant, due to uncertainty in the starting value as well as in the value achieved because of an intervention. It takes considerable sampling effort and time to establish a baseline before progress can be judged. This suggests that only rational modelling can provide a sensible estimation of a starting value against which to judge progress. But models are subject to assumptions and constraints, so in themselves are prone to bias.

It is logical to assume though, that if there is a commonly held belief that groundwater amenity would be improved if nutrient status is diminished, that sources of those nutrients need also to be diminished. The efforts here around the urine patch are to see if attention to that source will have a useful effect.

Conclusions

This re-analysis of available data suggests that a reconsideration of the role the dairy cow urine patch plays in water quality deterioration, particularly on the Canterbury Plains, is timely. The range in measurements for N concentration in urine, the volume in which it is excreted, and the area each event covers, plus the depth of soil penetration and subsequently the quantity leached, has created confusion. By bringing discussion of the urine patch to the fore and considering what other contributors there might be to N in waterways, it might be possible to focus efforts in areas where action can make a difference. In that context, were environmental lobbyists to get their wish to stop dairy (or any livestock) farming, the effect on water quality may be small to negligible, unlike the economic impact of such a change.

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