EFFICACY OF BMPS TO AMEND NITRATE CONTAMINATION IN GROUNDWATER SYSTEMS – A CANADIAN EXPERIENCE FOR COMPARISON TO NEW ZEALAND

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Introduction

Groundwater nitrate contamination stemming from longstanding agricultural practices remains a worldwide and increasing problem with serious economic (Oneil and Raucher 1990) and health effects (USEPA 1990). Of all agricultural contaminants, nitrate is the most widespread in exceeding national water standards (Spalding and Exner 1993; Goss, Barry et al. 1998). In agricultural areas of North America, for example, between 5-46 % of domestic wells in aquifers exceed the 45 mg NO_3 / L drinking water standard (Hamilton and Helsel 1995; Goss, Barry et al. 1998; USEPA 2002). Generally, nitrate in groundwater originates from long-term or historical point (i.e. waste lagoons, animal waste storage) and non-point (i.e. fertilizers, manure spreading, and fertigation) nutrient sources associated with local agricultural practices (USGS, 1996). The origin, fate and transport of nitrate in aquifers has been extensively studied over the past decades (Hendry, Gillham et al. 1983; Spalding and Exner 1993; Goss, Barry et al. 1998; Nolan and Stoner 2000; USEPA 2002; Visser, Dubus et al. 2009).

Since the 1990s, best management programs (BMPs) have been promulgated to minimize or amend the environmental impact of agricultural nutrients on ground and surface waters. These programs generally focus on producer based management of nutrient inputs in an attempt to strike a balance between economic considerations and improving the quality of the receiving ground and surface waters (*cf.* Turpin, Bontems et al. 2005). Most BMPs attempt to optimize the handling, timing, and application on the land surface in order to optimize crop nutrient uptake efficiency, reduce leaching of nutrients into ground and surface waters, and concomitantly provide real economic and agronomic efficiencies to producers. Currently, increasingly complex “systems models” (e.g. APSIM now being applied by AgResearch in New Zealand) are being promoted as tools to help guide producer BMP development (*cf.* Turpin et al., 2005). In contrast, nutrient budgeting tools such as New Zealand’s OVERSEER™ are not widely used in Canada.

While improved nutrient management can provide economic benefits and efficiencies to producers, we were unable to locate any studies in the scientific literature (search period = 1980-2011) where a BMP had demonstrably *improved* deteriorated aquifer water quality (Boyer and Pasquarell 1996; Stites and Kraft 2000; Stites and Kraft 2001; Neill, Gutierrez et al. 2004; Boyer 2005; Visser, Dubus et al. 2009). Thus, the long-term efficacy of BMPs to improve aquifer water quality remains to be scientifically demonstrated. This does not mean that BMPs are ineffectual, but rather that critical scientific examination of the real efficacy of BMPs is warranted to determine if abatement in the aquifer of concern may be expected to occur within realistic timeframes, and to help better focus BMP guidelines and accompanying groundwater quality monitoring strategies.

Unfortunately, there are relatively few aquifer systems in the world that are sufficiently suited to studying the efficacy of BMPs to improve groundwater quality due to a general lack of extant aquifer nutrient monitoring programs (Wassenaar, Hendry et al. 2006; Visser, Dubus et al. 2009). The Abbotsford aquifer in Canada (Figure 1) is a major water source where water quality monitoring began in the early 1950s. Otherwise excellent water quality in the aquifer is shown to be widely degraded with nitrates since the 1970s. Currently, nitrate contamination in the aquifer remains widespread, with many wells exceeding national standards up to 3-fold (Liebscher, Hi et al. 1992; Stuart, Rich et al. 1995; Wassenaar 1995; Cox and Kahle 1999; Mitchell, Babcock et al. 2003). Nitrate contamination in this aquifer remains a serious international trans-boundary water quality concern in B.C. and Washington more than 4 decades after the nitrate problem was first identified. Similar decadal-scale nitrate trends were noted for European aquifers (Visser, Dubus et al. 2009).

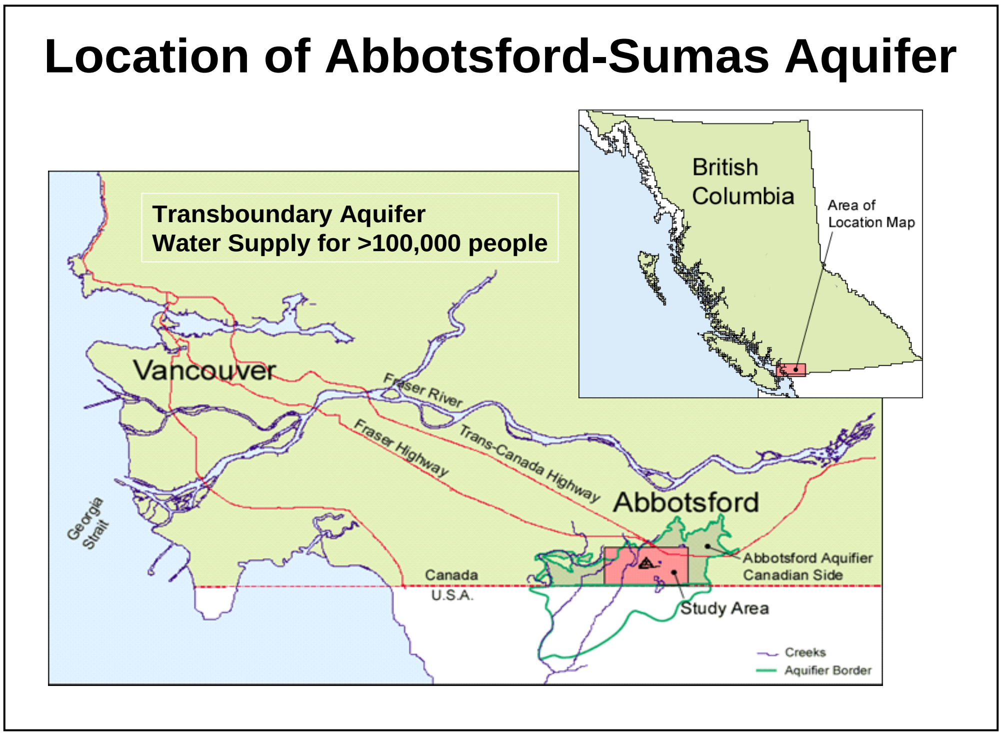

Figure 1. Location of Abbotsford Aquifer, Canada.

On both sides of the Canada-US border, agricultural land use above the Abbotsford aquifer is dominated by raspberry fields (~80% of land use) and large poultry operations and dairy farms (Nelson 1992; Mitchell, Babcock et al. 2003). Previous investigations showed that nitrate contamination stemmed from poor agricultural land use practices, such as roadside and uncovered animal waste stockpiling and manured over-fertilization of raspberry fields (Kohut, Sather et al. 1989; Zebarth, Dean et al. 1994; Wassenaar 1995; Wassenaar, Hendry et al. 2006). A nitrogen budget estimated that a seven-fold excess of manure N was historically applied to raspberry fields in combined form of animal waste and fertilizers, with subsequent leaching of excess soil nitrate through the permeable sands and gravels during fall rains and with irrigation (Zebarth, Dean et al. 1994) . Furthermore, nitrates in the aquifer undergo little biogeochemical transformation because the aquifer has little intrinsic capacity to support widespread geomicrobial denitrification (Wassenaar 1995; Tesoriero, Liebscher et al. 2000).

In 1992 the government of British Columbia implemented a BMP known as the *Code of Agricultural Practice for Waste Management* (B.C. Ministry of Water 1992). This provincial BMP mandated that animal waste must be stored in contained storage facilities, temporary onsite manure storage be less than two weeks, outside manure storage be covered between October and March to avoid nutrient leaching during the fall and winter rains, and that producer applications of manure and fertilizers be optimized to closely meet crop requirements, and not cause contamination of surface or groundwater . About the same time as the Provincial BMP *Code* was established, the voluntary Abbotsford-Sumas aquifer *Groundwater Protection Program* was begun by local poultry producers to develop alternate markets for animal waste. By developing alternate markets for animal waste previously disposed on the aquifer ground surface, the industry hypothesized that nitrate concentrations in the aquifer could be amended. Of approximately 573,000m^3 of animal waste produced in the area annually, the SPFG currently removes approximately 7% (38,000m^3) to alternate markets. Similarly, the raspberry industry implemented a BMP approach to reduce residual fertilizer and manure soil nitrate leaching into the aquifer during late summer by planting between-row cover crops, and by diversification to blueberries with lower nutrient requirements.

A decade after BMP deployment a compliance report (B.C. Ministry of Water 2005) revealed that 76% of farms operating on the Abbotsford aquifer still did not have an adequate nutrient management plan for their operations. Most of the compliance issues identified were implicated as key contributors to nitrate contamination of groundwater – the improper storage and spreading of solid and liquid animal wastes. Whereas large commercial operations were in better compliance (86%) over small farms (50%), the relative contribution of either type of non-compliant operation to nitrate leaching was unknown. Previously, the efficacy of a BMP in amending aquifer-scale nitrate contamination was conducted by interpreting the trends and sources of nitrate in the Abbotsford aquifer approximately one decade after implementation (1995-2004 (Wassenaar, Hendry et al. 2006). A summary of those findings are reviewed here, and the Canadian use of BMPs is compared to New Zealand's emerging approaches in our concluding statements.

Materials and Methods

The Abbotsford aquifer reaches about 60 m in thickness in the north-east part of the aquifer, with depth to the water table ranging from a few meters to over 10 meters (Kohut, Sather et al. 1989; Liebscher, Hi et al. 1992; Cox and Kahle 1999). Flow directions in the aquifer are to the south, southeast, and southwest at velocities of about 450 m/a (Liebscher, Hi et al. 1992). The aquifer is highly vulnerable to contamination due to its coarse sediments, and because the area receives precipitation of about 1500 mm/yr, of which 75% occurs between October and March when nutrient uptake by crops is the lowest and nitrate leaching potential is the highest. It is estimated that between 40 and 80% of the annual precipitation recharges the aquifer (Kohut 1987; Cox and Kahle 1999).

Groundwater samples were collected from a combination of research monitoring wells, domestic, and municipal wells, and surface water bodies (Wassenaar 1995; Wassenaar, Hendry et al. 2006). Nitrate isotope assays were conducted using the method described by Casciotti et al. (2000, 2001). Groundwater ages were determined using the $^3H/^3He$ dating method (Tolstikhin and Kamenskiy 1969; Solomon and Cook 2000). Long-term nitrate data were obtained from Environment Canada (1989-2004). Statistical analyses were done using Minitab, and multivariate time series analyses were conducted using the Mann-Kendall method for the determination of trends (Lettenmaier 1988; Thas, Van Vooren et al. 1998).

Results

Temporal nitrate concentration data showed 64% of monitoring wells increased in nitrate concentrations between 1993-2004. The increase ranged from a minimum of 0.4 mg NO_3/L to a maximum of 67.3 mg NO_3/L. The average increase was 28 mg NO_3/L . Of the wells that showed a decrease in nitrate (37%), the decrease ranged from a minimum of 0.3 mg NO_3/L to a maximum of 40.7 mg NO_3/L. Overall, nitrate concentrations in the monitoring wells increased by an average of 14 mg NO_3/L over the decade. The majority of domestic wells re-sampled (71 %) showed a decline in nitrate concentrations over the decade with individual well decreases ranging from 2.4 mg to 138.9 mg NO_3/L . Of these, 29 % that showed an increase that ranged from 0.4 to 62 mg NO_3/L with an average overall increase of 30 mg NO_3/L. On average, the nitrate concentration in domestic wells decreased by 12 mg NO_3/L over a decade (Wassenaar 1995; Wassenaar, Hendry et al. 2006).

For domestic and monitoring wells combined the data yielded an average overall increase in NO_3 of ~3.0 mg NO_3/L in the aquifer. However, a paired t-test revealed that there was no significant difference between the mean nitrate concentrations in the aquifer between both time periods (p>0.1, n=57). These average nitrate concentrations for 2004 and 1993 exceed Canadian and US drinking water standards by more than 15%. The percentage of domestic and monitoring wells that exceeded nitrate drinking water standards was 59% in 2004, virtually unchanged from 58% in 1993. For 1993 and 2004 there were no significant trends in NO_3 concentration in the aquifer with well depth. Nitrate concentrations were spatially variable across the aquifer at both dates due to rapid groundwater flow and variable nitrate inputs across the landscape (Wassenaar 1995; Wassenaar, Hendry et al. 2006).

Changes in δ^{15}N of Nitrate

It was previously shown using nitrate isotopes that the majority of nitrate in the aquifer in 1993 could be attributed to animal waste sources (Wassenaar 1995). The δ^{15}N of nitrate derived solely from locally used NH_4 based fertilizers should range between about 0 and -2 ‰. The δ^{15}N of nitrate derived from animal waste in the Abbotsford area, however, should be at least +8 ‰, or higher (Wassenaar 1995), and from other studies of NO_3^- derived from animal wastes, the $\delta^{15}N_{NO3}$ could vary from +8 to +16 ‰ due to ammonia volatilization during storage and transformation in soils (Kreitler 1975; Heaton 1986; Kendall 1998).

The δ^{15}N of nitrate in the Abbotsford aquifer was +9.8 ± 3.9 ‰ and +9.3±5.1 ‰ in 1993 and 2004 respectively, suggesting little if any net change in the overall nitrogen source(s) of nitrate contamination. Where we had directly comparable well data, a *t*-test revealed no significant difference in the mean δ^{15}N of nitrate between 1993 and 2004 (p>0.1, n=53). The δ^{15}N data reaffirmed the interpretation that animal waste (δ^{15}N >+8 ‰) is the prevailing source of NO_3^- in the aquifer (Figure 3). Ammonium-based or nitrate fertilizers do not appear to be a significant contributor to prevailing extensive nitrate contamination, although δ^{15}N values between +2 ‰ and +8 ‰ suggest a mixture of manure and fertilizer NO_3^-. As in 1993, we found no significant correlation between δ^{15}N and NO_3^- or between δ^{15}N and depth in the aquifer in 2004 (Wassenaar, Hendry et al. 2006).

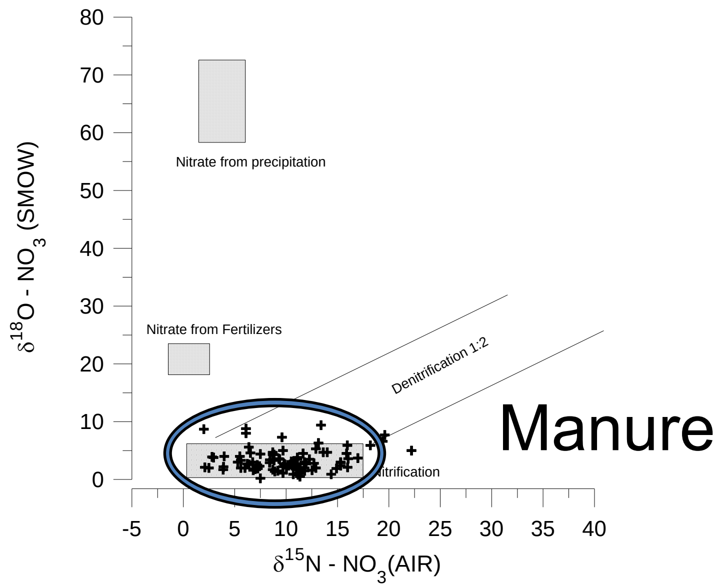

Figure 2. Nitrate oxygen and nitrogen isotopes in 1993 (Wassenaar, 1995).

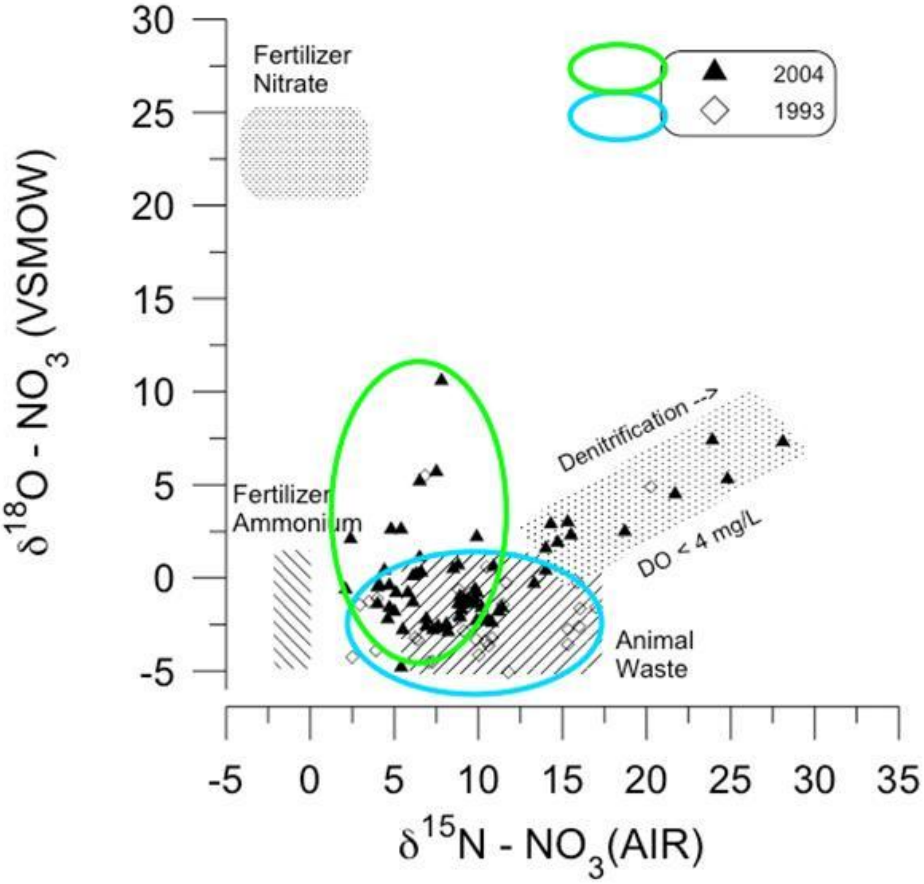

Figure 3. Comparing nitrate isotope changes in the aquifer between 2004 and 1993.

Although the overall nitrate and $\delta^{15}N$ data suggested that nitrate concentrations in the aquifer and sources of nitrate contamination did not change significantly over the decade. The lack of water quality improvements cannot be directly interpreted as a failure of BMPs without knowing the "age" of the groundwater associated with the nitrate. It is possible that many wells contain nitrate that entered the groundwater long before the implementation of BMPs, and so could lead to misleading conclusions. $^{3}H/^{3}He$ isotopic analyses were used to age-date groundwater samples from 19 research and monitoring piezometers instrumented in the aquifer. The $^{3}H/^{3}He$ age dating method was chosen because of its ability to accurately (±0.5 y) age date water that entered the groundwater system within the past 40 years (Tolstikhin and Kamenskiy 1969; Solomon and Cook 2000), thus allowing one to bracket nitrate trend assessments to times before and after establishment of the BMPs. Groundwater ages in the Abbotsford aquifer ranged from 0.8 to 33 years. BMPs implemented a decade ago ought to have a significant positive impact on groundwater samples that are at most less than 5 years old, allowing for the fact that travel time through a variable thickness of unsaturated zone needs has to be accounted for. One would not anticipate water quality improvements resulting from BMPs in groundwater samples that are a decade or older, other than by naturally occurring intrinsic biogeochemical processes such as *in situ* denitrification or mixing.

Nitrate and $\delta^{15}N$ Trends with Groundwater Age

Long-term temporal trends in nitrate concentrations from two groups of monitoring piezometers were examined; those with groundwater ages determined using $^{3}H/^{3}He$ to be older than 10 years old, and those groundwater ages less than 5 years old. For ground water older than 10 years old, and where the impact of BMPs was not expected to occur, a Mann-Kendall non-parametric test for detection of trends in a time series (Kendall, 1975, Lettermeier, 1988) revealed no significant trend up or downward in nitrate concentrations over the past decade (p>0.1). .

Where the groundwater age was determined to be less than 5 years from $^{3}H/^{3}He$ ages, *and* where the data was constrained for trends since BMP implementation, the Mann-Kendall test revealed a significant increase in nitrate levels ($p<0.05$). When broken down further on a seasonal basis, this trend of increasing nitrate over the past 5 years was weighted to those groundwater samples collected the months of September to December, and to a lesser extent the growing season months, supporting the view that fall and winter rains cause rapid leaching of nitrate into the aquifer.

The increasing trend in nitrate concentrations in the youngest of groundwater in the aquifer does not support the hypothesis that the BMP Code was (yet) having a positive effect in improving groundwater quality, particularly over a timeframe where improvements in water quality were expected. The increasing nitrate trends may be interpreted in light of changes in nitrate sources stemming from BMP implementation. These findings were not unexpected. The past decade has seen a BMP driven shift away from utilization of animal waste as a fertilizer, towards managed nutrient applications through the use of fertilizer banding (B.C. Ministry of Water 1992; Kowalenko, Keng et al. 2000; Rempel, Strik et al. 2004). Fertilizer banding has the potential to allow for residual nitrogen leaching into groundwater (Zebarth, Dean et al. 2002). Previously, it was shown that the $\delta^{15}N$ of nitrate extracted from soil beneath raspberry fields was considerably lower than nitrates extracted from soil beneath manure piles (Wassenaar 1995). Thus, the comparative 1993-2004 $\delta^{15}N$ data, though relatively few, suggest that the shift from animal wastes to fertilizer use may be leading to worsened groundwater contamination of the Abbotsford aquifer due to leaching of unused chemical nitrogen fertilizers.

Conclusions

After a decade of agricultural BMPs aimed at amending nitrate contamination in the Abbotsford aquifer, all of the geochemical, isotopic, and temporal evidence indicate that nitrate contamination in the aquifer may be poised to worsen, not improve. The monitoring data from recent (<5 years old) groundwater revealed a trend of increasing nitrate levels which corresponded with a $\delta^{15}N$ isotopic shift in nitrogen sources away from manure sources towards fertilizers. Comparative dual nitrate isotope analyses also revealed this shift in nitrogen sources from animal waste towards fertilizers occurred in the decade 1993-2004. This trend of increasing nitrate levels and the isotopic shift to fertilizers may be an inadvertent consequence of a BMP driven change away from the application of animal wastes landscape towards the use of chemical fertilizers used in cane production. These findings and previous research (Zebarth, Paul et al. 1999; Zebarth, Dean et al. 2002) suggest that fertilizer banding practices applied to irrigated cane crops may be resulting in more easily leachable residual nutrients in the soil, compared to the soil conditioning and nitrogen cycling properties of animal wastes. It is possible that the best-designed BMPs may be ineffective at preventing or amending widespread nitrate contamination in the Abbotsford aquifer. This potential scenario presents an enormous challenge to local stakeholders who are working together to mutually achieve agricultural objectives, water quality objectives, environmental standards and community expectations.

The efficacy of voluntary BMPs in amending aquifer nitrate contamination has yet to be been successfully demonstrated in the scientific literature. This study showed that, despite best of intentions, agricultural BMPs could also inadvertently contribute to *enhanced* nutrient contamination of groundwater under irrigated settings. BMPs aimed at meeting nutrient objectives for groundwater in agricultural settings must be developed in full partnership with groundwater quality monitoring programs.

The current state of Canadian BMP approaches remain producer-focused (instead of receiving environment) and on managing and optimizing nutrient inputs, with the hope that plant nutrient optimizations will precede anticipated improvements in ground or surface water quality. Unfortunately, the failure or success of input-focused BMPs may take years or decades to determine without a scientific means of tracking and quantifying success. Ideally, BMPs ought to be developed as part of a comprehensive agricultural nutrient utilization and flux monitoring program. For example, an environmental nutrient management program aimed at protecting groundwater could be developed to include targeted annual monitoring of soil and vadose zone residual nutrients levels concurrently with nutrient level monitoring of shallow groundwater wells. Deficiencies in the efficacy of the BMPs to prevent nutrient leaching through the soil and vadose zone would be more quickly identified, and appropriately adjusted in consultation with local agricultural producers and agronomists. Finally, voluntary participation in BMPs may lead to the noncompliance issues, whereby the effect of non-compliant “offenders” in highly vulnerable aquifer settings has to potential to jeopardize overall water quality objectives.

From a New Zealand perspective, it can be argued that approaches of considering and in some cases implementing nutrient caps, within an appropriate management and trading scheme, appears more likely to generate positive outcomes in water quality. New Zealand’s approaches have, in general, addressed many of the issues that appear to have undermined the success of BMPs. These include a focus on managing the receiving environment, and on evaluating success using monitoring programmes. Interestingly, other aspects of New Zealand’s approach may create some risk. In contrast to Canada’s desire to understand a transboundary issue through science, New Zealand has avoided transboundary water issues by aligning Regional Councils on watershed boundaries. This arrangement has been positive from most perspectives, but the lack of transboundary issues and sheer number of regional authorities has caused New Zealand’s science focus to be dispersed and focused more on monitoring concentrations and processes than in developing new tools, such as isotopic tracers for tracking sources and processes (such as denitrification). Without the development of these tools, uncertainty appears likely to remain large – particularly given extremely limited understanding of groundwater in many regions. As yet, efforts to manage water quality within New Zealand’s legislative and institutional framework can be described as innovative, but uncertain. In contrast to cap and trade systems, future certainty has been a hallmark of regulatory systems. We advocate the ongoing development of New Zealand-specific versions of measurement-based tools that can be applied from the development through to the implementation of policy as an important step in improving certainty for stakeholders in water quality management.

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