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Community Governance: An Alternative Approach to Regulation and Market Mechanisms for Management of Nitrogen Loss

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Summary

The National Policy Statement on Freshwater Management now requires that water quality objectives and limits be set for all water bodies in New Zealand. Where objectives result in the development of nitrate limits for waterways and development pressure is likely to make these limits difficult to achieve, systems are needed to avoid over-allocation. This paper proposes a multi-level governance model for managing this load over time, with a focus on community self-governance and building a flexible system for managing leaching risk, given the very high levels of uncertainty in linking nutrient losses to objectives.

Keywords

Diffuse pollution, risk management, market-based instruments, commons resource management, community governance.

Introduction

Following the success of initiatives over the last 20 years to reduce point-source discharges (e.g. industrial and municipal wastes) to water bodies, there is now increasing interest in measures to improve controls on diffuse nutrient losses from agricultural systems (OECD, 2012). The challenge is in designing policies which promote equity and innovation, balance environmental and market risks, and enable growth. Market-based instruments have gained particular attention as a means to provide for flexible and efficient regulation.

A weakness of some formulations of market-based instruments is that they can undermine intrinsic motivations and hinder the development of cultural sustainability (Burton & Paragahawewa, 2011). In some cases, markets can even cause civic-minded individuals to behave selfishly (Bowles, 2008). Internationally, there are many examples of culturally sustainable resource management, based on common property approaches (Ostrom, 1996). However, one of the key attributes of successful common property approaches is the salience of the resource condition to the livelihood of its users (Ostrom, 2008). For example, if a fishery becomes degraded, all users are negatively affected. For the management of nitrate losses from agriculture, the relationship does not hold – degradation of a waterway often may not directly affect the users. Despite this, we know that external drivers can be sufficient to precipitate collective action, though it has not been widespread. One of the key barriers to the effective implementation of common property approaches currently is the absence of effective appropriation and provision rules to ensure common goals are achieved. In developing rules to support community governance, there may be much that we can learn from market-based instruments.

This paper utilises New Zealand dairy farming examples and perspectives to discuss the potential merits and suitability of applying a performance-based approach to community governance of agricultural nutrient losses as a means of managing growth. It begins with by reviewing some key issues from the current policy context in New Zealand. Secondly, some emerging policy options are examined. Finally, a potential institutional structure for community governance of a catchment-based system is proposed.

Review of Current Policy Context

Performance-based Instruments

Different types of environmental management instruments can be characterised in three broad categories (Coglianese & Lazer, 2003):

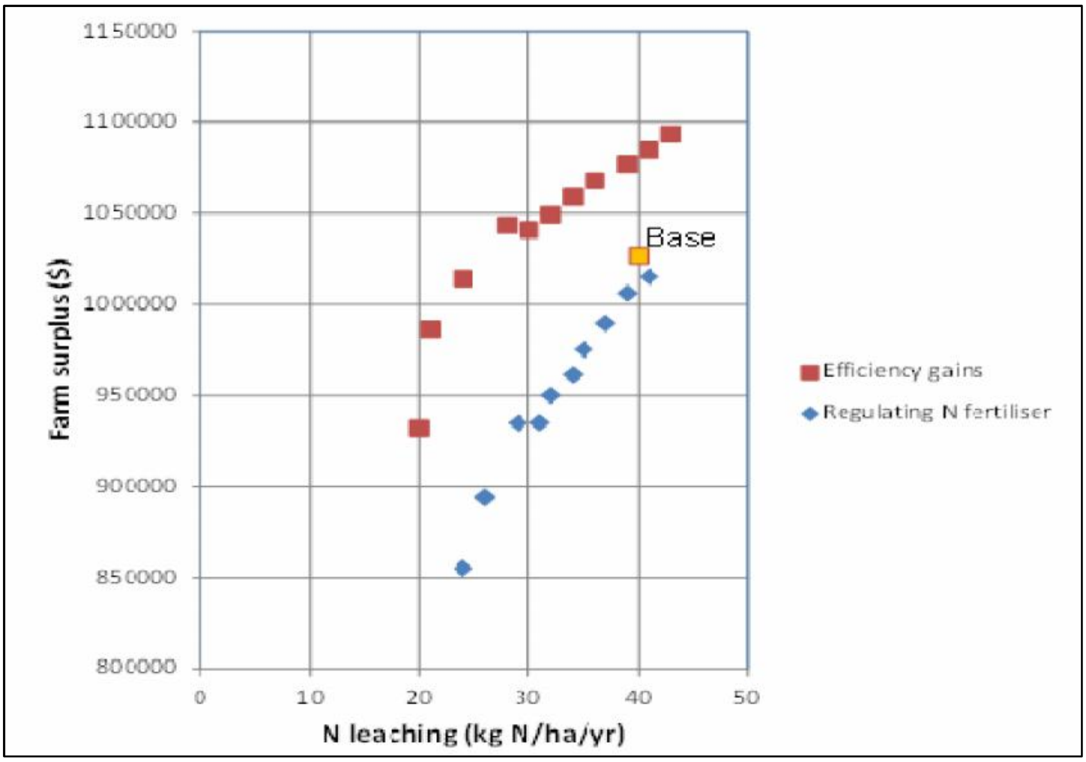
- Technology-based (implementing particular technologies, e.g. low-rate dairy effluent irrigators, riparian strips)
- Management-based (focused on actions taken, e.g. particular stock wintering practices)
- Performance-based (focused on actual outputs, e.g. kgN leached)

Environmental regulations in the European Union have tended to focus on technology or management-based instruments that promote good management practices. While this has been effective in some areas, in others policies have failed (for instance, in controlling nitrate leaching) (OECD, 2007). Economic instruments such as taxes have been applied to inputs such as fertilisers, but with mixed results. In some cases, focusing on inputs rather than output performance has made matters worse, incentivising shifts to cropping systems that require less fertiliser, but increase leaching (Randhir & Lee, 1997).

Performance-based approaches to the management of agricultural nutrient losses have gained increasing attention in recent years, particularly in the context of applying market-based instruments. There are a number of strengths to focusing on performance that are not realised from a focus on particular technologies or management practices, which have resulted in increased interest in water quality trading as a policy option.

Focusing on performance incentivises innovation and encourages the maximisation of production within set environmental limits (i.e. increasing efficiency). This may be of benefit to both farmers and the wider public, providing highly desirable 'win-win outcomes'. While regulating inputs and practices may cause large production losses for a given environmental benefit, improving efficiency within a performance standard can result in co-benefits. For example, modifying feed conversion efficiency enables higher production and reduced leaching without defining inputs or practices. Figure 1 shows an example of differences in cost-abatement curves for an input-controlled scenario and a farm system optimisation process, based on modelling of a farm in the Waituna Lagoon catchment in Southland (B. Ridler, unpublished data).

Figure 1: Input Controls versus Farm System Optimisation



In addition to these economic and environmental efficiency gains, it has also been suggested that the use of performance-based approaches encourages the cultural sustainability of environmental management, since those involved are responsible for achieving their own targets, but without a prescribed process, and thus have greater ownership of the results (Burton & Paragahawewa, 2011). Performance-based approaches also allow for experimentation and learning, building new skillsets in environmental management. It is important to recognise that many good management practices may have significant initial capital outlay or other transitional costs. While theoretically providing incentives for adoption, many of these "win-win" actions may have relatively marginal incentives. There are a number of barriers which inhibit adoption, including education costs, information costs, risk aversion and complexity (Pannell, 2006). If the deterrent of these costs is to be overcome, then building a culture that values better environmental performance must be a central consideration in policy-making. While efficiency may be an important driver, institutions which support group learning and cooperation are also likely to add significant value to policy design.

Transferability and Water Quality Markets

Farm systems can be highly heterogeneous. Applying a "one size fits all" approach is thus highly likely to result in policy failure in the long-term. Flexible systems (such as market-based instruments) that enable producers to make strategic decisions about their level of environmental responsibility and support the transfer of rights are also likely to result in significant efficiency gains (Jack et al., 2008). Internationally, market-based instruments have been focused primarily on price-based schemes (such as fertiliser taxes across Europe), quantity-based input quota (such as the Danish nitrogen quota system) or on baseline-and-credit water quality trading systems[1] (the majority being in the United States) (OECD, 2012).

In terms of quantity-based instruments, New Zealand is unique in that it has the world's only agricultural cap-and-trade water quality market, in the Lake Taupō catchment. Baseline-and-credit systems have received little attention in policy debate in New Zealand, possibly due to the relatively small number of point sources that could be used to create demand for offsets. A further limitation on the potential for implementation of baseline and credit schemes has come from the National Policy Statement on Freshwater Management, which requires the setting of water quality limits and the avoidance of their over-allocation – effectively setting an overall cap on a water body, rather than particular sources. In catchments under development pressure, there is potential for the reductions in nutrient loads due to offsets from point sources to be overwhelmed by increased diffuse sources of nutrients, resulting in a limit being breached. While baseline-and-credit systems could work well in the absence of development pressure, there has been a significant amount of interest in capping agricultural sources as a method of meeting limits in some areas experiencing significant growth (for instance, in the discussion of

[1] Baseline-and-credit markets differ from cap-and-trade, in that not all sources are capped. Usually, only point sources will be capped, but they will be able to offset their discharges by purchasing credits from non-capped agricultural sources. Credits are generated by implementing measures above and beyond a good practice "baseline".

transferable "Nitrogen Discharge Allowances" for Canterbury). This represents a focus on performance-based instruments for agricultural systems that is quite unique internationally.

Applying market-based instruments to nutrient loss performance is an understandably attractive policy option. However, effective implementation of a performance-based market approach to regulating diffuse nutrient losses is dependent on three conditions (Shortle, 2012):

1. Nutrient losses can be accurately measured for each farmer.
2. Nutrient losses can be largely controlled by each farmer.
3. The spatial location of nutrient losses does not affect environmental outcomes.

Spatial location may or may not apply in a meaningful sense, depending on the environmental context. This condition can in some cases be managed through market design and has been well-addressed elsewhere (Anastasiadis et al., 2012; Jack et al., 2008; Selman et al., 2009). The first two conditions are rather more fundamental. Measurement of the actual nutrient losses from a given agricultural operation is not practical or cost effective with current technology (Shortle, 2012). This has led to increasing interest in various models which may estimate nutrient losses – in particular the OVERSEER™ model in New Zealand (Ledgard et al., 2001). The use of models in this way creates a fundamental problem, in that the incentive to innovate that comes from a market-based approach is limited by the model's capacity. This being said, model-based policies may still theoretically drive uptake of good practice, if the model is able to provide a reasonable reflection of known best practice.

In relation to the second point, there may even be some advantages to the focus on models. While nutrient cycle modelling may provide an estimate of leaching risk, actual leaching will vary enormously from year-to-year owing to the highly-stochastic nature of climate variables. While given pastoral systems and management practice may allow an estimated loss, the actual loss may vary significantly, due to variables beyond a farmer's control such as temperature and rainfall. For example, if long-term average leaching is estimated at 58kgN/ha/year, the actual leaching may range from 13 to 161kgN/ha/year, while holding management practice constant[2].

It is a well-established principle of common law that in order for there to be a duty to avoid something, impacts need to be reasonably foreseeable. While actual environmental impacts might be calculated retrospectively, predicting the influence of climate for any given year is more difficult. While it would be theoretically possible to design software for real-time modelling of leaching, there remains a question as to whether this would be a desirable way to regulate nitrogen losses. Under precision irrigation, it may be possible to manage leaching risk at this level of time-bound specificity. However, precision management of nitrogen loss risk is

[2] Source: Graeme Doole (University of Waikato). This scenario represents modelling of nitrogen leaching for 100 years given a stocking rate of 3.08 cows, 105 kgN applied in fertiliser and 0.37 tonnes of maize silage per hectare. The data represents nitrogen leaching at 150 cm on an allophanic soil in the Waikato region (Hamilton climate).

likely to be problematic in conventional rain-fed farm systems. Business planning would become very difficult, production would become more volatile and farming decisions would need to be almost entirely focused on environmental impacts. Given the limited ability of most farmers to foresee actual nutrient losses, we cannot reasonably expect farmers to be held responsible for performance in terms of actual environmental impacts. However, we can design policies around farmers' risk management performance. In effect, creating tradable responsibilities for the level of risk management effort farmers are willing to undertake.

While a risk-management approach is far from providing any kind of environmental certainty, we can develop policy based on models that delivers some of the benefits of a true water quality market. What such a policy enables is for farmers to make strategic decisions about their duty of care, allowing some equalisation of mitigation effort and a reasonable level of environmental risk management. However, given the high level of uncertainty in actual environmental outcomes, it is important not to imply spurious precision or assign undue importance to precise compliance.

Key Determinants of Policy Suitability

The viability of trading in nutrient leaching risk management is only half of the equation. The second half is concerned with whether such an approach is preferable. This depends on the objectives for the water body in question, the nature of the contaminant risk being traded and whether the catchment is likely to experience development pressure which makes biophysical limits difficult to achieve.

Nitrogen versus Phosphorus

Phosphate and nitrogen behave very differently as contaminants and thus have different uncertainty characteristics in modelling. The behaviour of phosphorus is well understood in waterways, but we have higher uncertainty in understanding how it behaves at an enterprise level. Diffuse phosphorus loss is heavily influenced by transport factors such as erosion, surface runoff and preferential subsurface flow (Sharpley et al., 2001). Because of this, there are unlikely to be benefits from a focus on performance, regardless of model sophistication. The transaction costs and complication associated with designing a market with such a high level of spatial granularity are unlikely to make the potential cost-savings worthwhile. In addition, as phosphorus binds to soil particles, there are a range of options at a farmer's disposal which enable cost-effective mitigation using a simpler management-based approach. Farmers may choose to simply mitigate rather than engage in trading activity.

Conversely, nitrogen loss can be better modelled at the enterprise level, but we have higher uncertainty about its effect on waterways. Nitrogen is much more costly to manage than phosphorus and while higher nutrient use efficiency allows for greater production for a given environmental effect, rates of nitrate leaching will tend to bear some relationship to the overall intensity of production in a catchment (Clapcott et al., 2011). The greater accuracy of nitrogen modelling at the enterprise level, combined with its lower level of substitutability, make nitrogen risk management more suitable for market design than phosphorus. For these reasons, nitrogen losses will be the focus of this paper.

Development Pressure

In many mature dairying areas of the North Island (for example, Taranaki) the management of growth is not a significant issue. In these cases incremental improvement in the efficiency of production and quality of management practice in a catchment will mean that water quality and ecological health improves over time (Wilcock et al., 2009). In many cases, this will be sufficient to ensure that water body objectives are being met. In some cases, an important environmental goal may mean that it is worthwhile setting industry standards for the farmers involved and targeting extension towards achieving those standards. In the absence of development pressure, markets are unlikely to deliver sufficient benefits to be justified, unless a catchment is so over-allocated that reductions are needed beyond what is reasonably achievable by farmers through good practice. In these cases, baseline and credit schemes may be able to deliver significant gains.

However, where there is growth occurring alongside potential for nitrate concentrations to become a problem in the future, the use of good management practices in the absence of controls on the overall intensity of production in a catchment cannot deliver the required environmental results – even if the catchment is currently in an under-allocated state. At a catchment scale, no single farm can be considered a cause of degradation due to nitrate loads. It is the sum-total effect of all land uses in a catchment that produce effects on water quality. Where there is a relatively small amount of intensive land use in a catchment, rates of nitrate leaching are not particularly important, as their total effect will remain small and any limits for a waterway can be met with relative ease. However, as the scale and intensity of production increases, the total nitrate loading will increase and pursuing good practice and higher rates of nutrient use efficiency become increasingly important if catchment limits are to be met.

If land use intensifies further, even with best practice, limits will be breached. This leaves two options:

- Abandon the limits that have been set
- Retroactively impose controls on land use intensity in the catchment

From the perspective of the wider public interest, abandoning limits is unlikely to be viable. There is thus significant risk that retroactive controls will be placed on land use in the catchment. From the point of view of intensive producers that have been in the catchment for some time, this represents a threat to their livelihood for degradation of water quality that was beyond their control. Where there is potential for this kind of over-allocation to occur, unbridled expansion may penalise existing operators. Without the ability to exclude new resource users from the assimilative capacity of a water body, current users have no control over the condition of the resource, regardless of their efficiency or responsible use of the resource.

Flexible negative incentives that prevent undesirable change such as tradable permit mechanisms perhaps represent the best method of avoiding over-allocation. Growth can then still be possible, but some guarantee can be provided that existing users' efforts to farm more sustainably are not wasted.

Risks Associated with Capping Nitrogen Losses

The risks associated with capping agricultural nitrate leaching and allowing trading in leaching risk management are very poorly understood. Indeed, there is only one such market in the world (in the Lake Taupō catchment) and its implementation is so recent that few conclusions can be drawn about its impacts (Shortle, 2012). Where policies are introduced to limit nitrate leaching, this limitation will devalue land in many cases (especially in the case of scarce allocations for leaching rights). These impacts on land prices have been explored elsewhere (Kerr & Lock, 2009). However, there are a number of other possible risks which are worth highlighting.

The first (and perhaps most obvious) is the erosion of competitiveness. Capping nitrogen loss limits producers' ability to intensify in the face of increasing costs, unless they are able to acquire additional loss permits. For producers that are primarily price-takers trading in commodities, this may be difficult. Seeking higher-value markets may work for some producers, but there is unlikely to be sufficient demand for such products for this to be a viable option for all producers.

The second and more subtle risk is the effect that trading in nitrogen loss may have on farm systems. In a capped environment, the capacity of producers to intensify is determined by their ability to acquire additional nitrogen loss permits. This makes the amount of profit a producer is able to generate for each kilogram of nitrogen leached an important component of farm system optimisation. In dry-stock operations for example, breeding beef animals becomes far less attractive than finishing them, as finishing offers farmers far greater returns per kilogram of nitrogen leached. This is not a problem in and of its self, but raises the question of where sufficient animals for finishing will be bred if nitrogen capping becomes widespread. There is also some evidence that the introduction of nitrogen capping and tradability accelerates conversion from sheep and beef to dairy farming, as the higher returns make for a more sustainable business due to the ability to acquire additional nitrogen loss permits (Bartle, 2011).

There are a number of potential scenarios which could eventuate for dairy operations in a nitrogen trading scheme. Profit per kilogram of nitrogen leached could mean a tendency towards low to medium-intensity systems. Depending on commodity prices, an alternative scenario may be that off-pasture dairying based on cropping and cut-and-carry operations becomes optimal. This would have potentially negative impacts on soil health and higher sediment and phosphate loss due to increased cultivation. In addition to further erosion of industry competitive advantage, the increased cost associated with infrastructure requirements would make progression to farm ownership far more difficult. The corresponding increase in corporate farming may have significant impacts on the composition of rural communities, with adverse social consequences.

Currently there is a spread from very low, to very high-intensity systems, which underpin the sector's ability to manage climate (Clark et al., 2012) and price risks.

While commodity prices are high, high-input systems can chase big gains – but when prices fall, the blow is cushioned by lower intensity systems. The overall industry risk profile is well-spread. If imperatives relating to profit per kilogram of nitrogen result in a level of farm system convergence, this will increase the risk faced by the industry as a whole – and consequentially New Zealand, given the large share of export earnings generated by the sector.

Emerging Policy Options

Improving Policy Efficiency

Market-based instruments can create efficiency gains if transaction costs are low, particularly where there are heterogeneous marginal abatement costs (Jack et al., 2008). However, this can only be realised if they are appropriately structured and implemented (Selman et al., 2009). If transaction costs are not kept low and trading efficiency kept high, then there the potential benefits of a market system will not be realised (McDonald et al., 2010).

An important determinant of market efficiency is whether monitoring occurs ahead of any trading activity, or afterwards (ex-ante or ex-post, respectively). Systems that require monitoring and approval of discharges and mitigation changes before every trade significantly increase transaction costs and decrease trading efficiency (McDonald et al., 2010). The only cap-and-trade market for diffuse agricultural nutrient losses internationally is that of Lake Taupō, which is managed ex-ante. Theoretically speaking, designing markets with ex-post monitoring should be able to greatly increase efficiency and the overall amount of trading that occurs. However, there may also be other important considerations.

Firstly, farmers are often risk-averse, which may have a large impact on their response to a particular regulation (Bontems & Thomas, 2006). Some may prefer to not engage in trading, preferring a stable stream of allowances. Unlike regulated point sources in most water quality markets, the vagaries of climate variability and international commodity prices mean that the ability to adapt management strategies according to exogenous conditions forms an important part of rural resilience. The use of a multi-year rolling average may go some way towards addressing this. However, even if allowances are not needed, farmers may prefer to hoard them as a risk management strategy, rather than risk non-compliance. There is some evidence of this occurring in the Taupō nitrogen market (Bartle, 2011). This has potential to drastically reduce overall resource use efficiency. One way to deal with this is to allow trading for the season to occur at the end of the monitoring period to settle deficits and surpluses (kgN), in addition to trading for basal, year on year allowances (kgN/y). For example:

> Farmer A has been allocated 35kgN/ha/y for his farm. Following an increase in nutrient use efficiency, farmer A determines he will only need 30, so sells off 5kgN/ha/y. Bad weather means farmer A has to invest in extra imported feed in order to maintain stock condition. This means that at the end of the season, the farm is estimated to have leached 31kgN/ha. Farmer A then buys

> an extra 1kg/ha of leaching rights *for the season* off his neighbour who didn't use them that year – this means both are compliant without having to change their overall basal allowance.

This is a good way of solving some problems as it means producers can make long-term strategic decisions about the best level of nutrient loss management for their farm, while retaining a degree of flexibility. It also means producers can farm right up to the limit without fear of non-compliance, thus avoiding hoarding or under-production. The problem is that such adverse weather events are likely to have catchment-wide effects. So what happens in the case of one of these events? In the first instance, the price for any surplus allowances would become very high due to excess demand. This would leave many farmers with little choice but to cut production later in the season in order to remain compliant. Depending on the timing of such adverse events, input prices, commodity prices and the mitigation options that remain available later in the season, this may threaten the viability of some businesses. By adding nitrogen loss constraints, we have removed a significant amount of resilience from the socio-ecological system.

Policy Flexibility and Resilience under Uncertainty

Given the potential for significant adverse social and economic consequences under such a breakdown of the production system, it is important to understand the likely environmental impacts to determine whether this could be justified. As explained earlier, the good that is being traded under a modelled nitrogen loss system is an agreed level of risk management – not an actual environmental effect. Actual nitrogen loss is likely to have variance that is outside the level of the potential increase in nitrogen loss that would result from the farmers' adaptation to the adverse weather event. Indeed, cutting production later in the season may have little or no environmental benefit if peak levels of leaching have already occurred. Under this level of uncertainty, it is difficult to justify a strict approach to compliance with the overall environmental cap. A more flexible system is likely to deliver a more socially optimal level of production and mitigation.

Increasing interest is appearing in combinations of price and quantity-based instruments internationally, in particular through the use of threshold taxes, with tradable thresholds (for example, Pezzey & Jotzo, 2010). Combination systems like this work by combining tradable allowances with a fixed price for any emissions over and above those allowances. No charge is levied for emissions that are covered by permits held by the producer. A recent review showed how combinations of price and quantity-based instruments are potentially far more efficient in cases of uncertainty (Lehmann, 2012). So far there appears to be only one examination of threshold taxes in relation to agricultural nitrogen losses (Ramilan et al., 2011), but with promising results.

Setting a price for a threshold tax is likely to be difficult and require adaptation as the policy is implemented, depending on a particular community's attitude to risk. If environmental outcomes are paramount, a high price may be set. At the high end of the price range, a threshold tax could be used as a compliance tool. The high price for additional nitrogen loss allowances could then be used in place of penalties under

Resource Management Act for a strict compliance regime. Alternatively, if the environmental risk is lower (which may particularly be the case for many rivers and streams), the price may be used as a way of adapting limits to changing exogenous conditions. When we go through the limit-setting process, we are attempting to strike a balance between economic and environmental effects. Due to the high levels of uncertainty involved in making these trade-offs, it is highly likely that most limits will not be optimal. Using a combination of price and quantity-based instruments allows for adjustment in the face of uncertainty, so that if there is environmental damage, there is an economic benefit and vice versa.

A Proposed Model for Managing Nitrate Loss

Institutional Design

Clearly-defined responsibilities for nitrogen loss risk management may offer a useful means of managing the risk of over-allocation where there is strong development pressure. Well-designed mechanisms for transfer of these responsibilities have the potential to increase the efficiency of resource use. Combining this with a threshold tax may both improve overall efficiency and increase the resilience of the resource management system under uncertainty. Finally, there are a number of institutional and behavioural factors to consider in designing a system to manage nitrate loss risk.

Where the resource users involved have a sense of ownership of measures to protect the environment, compliance with standards and requirements will be increased, with reduced monitoring and enforcement necessary. Communities can be more effective than governments or markets where requirements are very incomplete or costly to enforce (Bowles & Gintis, 2002). In addition to utilising the superior information that is held by those closest to the issues, governance of this type may also be more conducive to the development of cultural sustainability (Burton & Paragahawewa, 2011). For these reasons, there may be considerable potential in implementing regulatory-type approaches through private governance structures, particularly when those affected are involved in determining the conditions. However, in using a common property approach to the management of nitrogen loads in catchments experiencing development pressure, some means of defining individual responsibilities and enabling the transfer of rights is necessary. While administration of such a system could be usefully managed by a regulator, there is potential for community governance of the scheme as a whole.

A number of necessary (though not necessarily sufficient) characteristics of effective community commons resource management have been identified through research after Elinor Ostrom (Ostrom, 1990), then expanded and empirically tested based on instances of successful and unsuccessful commons management (Cox et al., 2010). The following proposed model is intended to offer one option for integrating these conditions, which may be usefully applied to certain problems under certain conditions.

Community Governance and Cost-sharing

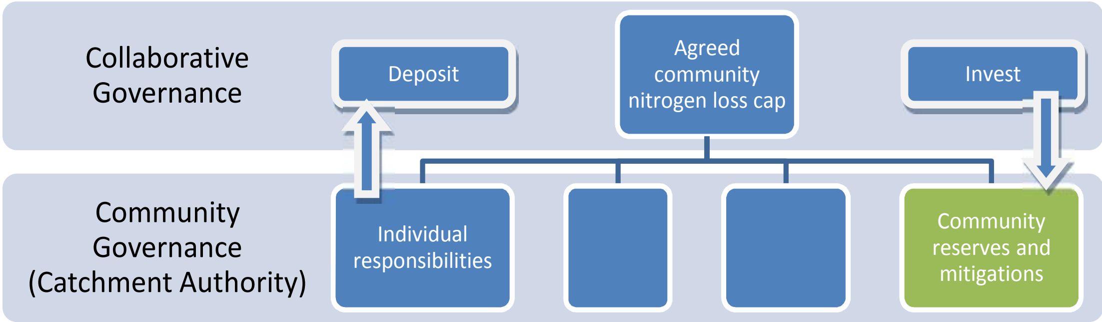
Commons research shows us that successful governance is much more likely to occur where the burden is shared equitably (Cox et al., 2010). When resource users can see that all in the community are working just as hard as each other, social and moral suasion are likely to have a greater impact on behaviour. This higher perceived fairness can mean that participants will be more willing to contribute (Ajzen et al., 2000). In addition, such a community governance system provides a useful foundation for the development of communities of professional practice, where resource users can engage in on-going dialogue to change their thinking and improve their collective environmental risk management performance.

The concept of community governance for nitrogen becomes problematic when voluntary effort fails. If farmers fail to meet their obligations and exceed their allocated nutrient leaching allowance, they create a cost to the community commons. If other participants feel that the burden is not being shared fairly, they are less likely to comply (Bowles, 2008). Thus, a means of fairly allocating this cost is necessary. In designing this, we can draw on mechanisms from market-based instruments – though applying them to community governance rather than regulatory regimes.

As explained earlier, threshold taxes can work well in combination with quantity-based instruments. However, while the threshold may reinforce the concept of moral norm for leaching, the tax could be potentially counterproductive to building cultural sustainability – i.e. the extrinsic motivation of the fine may undermine the intrinsic motivation to "do the right thing" (Gneezy & Rustichini, 2000). This could work quite differently in the context of community governance, as the influence of authority is less likely to have long-term effectiveness than the influence of people we like (Halpern et al., 2004). Since the payment is required by other members of the user group, it is more likely to reinforce intrinsic motivations – rather than undermine them as would potentially be the case if a fine was administered by a regulator (Bowles, 2008). In the context of community governance, it is likely to be the social sanction of others in the community that is most effective. However, payments may still form a useful means of cost-sharing.

For example, if a farmer fails to meet their risk management responsibilities in a given year, a deposit could be required into a community fund. The price of this deposit would be determined by calculating a fair price for the amount by which the allocation had been exceeded. Community sanctions of this type are likely to significantly improve compliance with risk management responsibilities (Bowles & Gintis, 2002). As outlined earlier, prices will vary according to community attitudes to risk. This deposit would then be fully refundable to the farmer if spent on mitigation. If the farmer chooses not to invest in mitigation, the deposit could be spent funding proposals for mitigation from other farmers in the catchment. The creation of this offset then reduces the total risk the community faces of exceeding their agreed common cap in the future. This community offset that has been created could be either redistributed equitably among the farmers according to pre-determined rules, or kept in reserve. In this way, any mistakes made by individual farmers would work to reduce the risk of exceeding the common cap faced by the community as a whole.

Figure 2: Community Deposit and Investment Structure



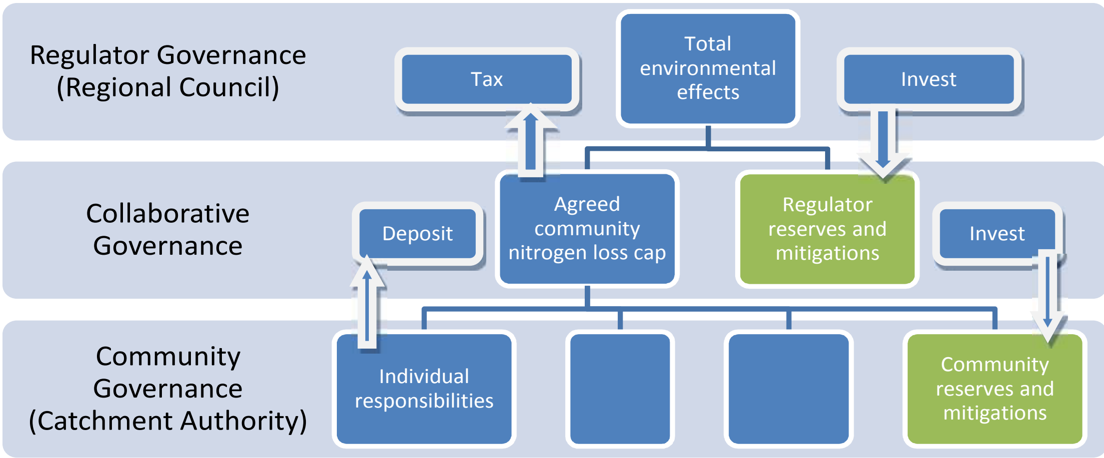
Naturally, the regulatory and legal framework within which such an arrangement is made plays a vital role. Some form of guarantee is necessary to ensure participants do not defect. "Contractual governance" offers a promising solution, whereby participants voluntarily join the scheme, but in doing so enter into legally binding agreements that require them to abide by the scheme's rules (Gunningham, 2009). This also creates an external driver that raises the salience of the issue for resource users.

Regulatory Backstops to Achieve Environmental Bottom Lines

The question that remains is what to do when the community as a whole exceeds their cap and fails to take action to mitigate the effects. A complementary regulatory approach may be of significant value here. In essence, the regulator could work with the community commons resource governance group to form a nested structure. This would enable the regulator to play a key role in ensuring environmental bottom lines are met, while still supporting community self-governance. This kind of nested structure may also prove to be a practical way for Regional Councils to effectively manage overall environmental impacts without creating a large drain on their resources from managing water quality limits.

In the same way that an individual farmer makes a deposit to the community fund to account for exceeding their allocation, a threshold tax could be required from the responsible individual members for contributing to a breach of the common cap. The coercive power of the regulator can be used correctively if contractual governance breaks down, in order to avoid a situation where community governance dissolves into litigation over individual responsibilities. The regulator could then use the revenue generated to invest in mitigation in the catchment and reduce the overall environmental risk. Earmarking and recycling revenue in this way is likely to significantly increase potential support for the model (Kallbekken et al., 2011).

Figure 3: Nested Deposit and Investment Structure



The price of this tax can form a further important lever to adjust for uncertainty. The agreed community discharge cap will be set to balance a particular set of economic, environmental, social and cultural conditions. If the profitability of farming were to change significantly for the better, the balance would shift. Farmers might then choose to go over their allowances for a season, as it may become more profitable to produce more and pay the tax. While exceeding allowances for the season, this provides the community with capital to invest in more permanent mitigation, reducing the risk of environmental effects.

In many catchments, water quality problems may be caused by a combination of nitrogen and phosphorus losses. In such instances there may be some value in allowing for interchangeable mitigations and offsets between the two variables, where co-benefits can be created. For example, it may be common for plant/algal growth in a water body to be phosphate limited (McDowell et al., 2004), but with a need to manage growth to prevent nitrate levels reaching potentially toxic levels (Hickey & Martin, 2009). In this case, the price of going over a nitrogen allowance can be used as a deterrent, but the revenue generated could be reinvested in phosphate mitigation, which would have a greater impact on water quality.

Multilevel Governance

This allows for different types of governance at the appropriate scales, with mechanisms to reduce both fiscal and environmental risk at each level. A farmer makes individual strategic decisions about their level of risk management by exchanging leaching allowances with other farmers in the catchment. Further exchange is allowed ahead of monitoring to settle any deficits and surpluses. If farmers mistakenly (or intentionally) exceed their allowance, payment to the community fund is required so that all participants share the burden equally. The community can use this to reduce their risk of exceeding the agreed environmental risk management bottom line. The threshold tax gives the regulator a means to guarantee that the environmental bottom line will be met in the long-term, without requiring needless regulation when community governance is performing well.

The use of a regulatory backstop also plays a significant role in supporting the success of community governance. Purely voluntary measures for natural resource management have a poor record of success. However, with regulatory approaches applied to encourage full participation in community governance, successful outcomes are far more likely (Gunningham, 2009). For example, farmers in the catchment who choose to not participate in a community scheme may face a more rigid and administratively costly regulatory framework, operating in isolation under individual consents. Using layered institutions in this way offers a promising strategy for a robust solution (Dietz et al., 2003).

Context-Appropriate Application of the Model

Inappropriate application of any governance model is likely to result in its failure. In order to understand how and when to apply such a model, a more diagnostic approach is necessary (Ostrom, 2007). While this proposed model may offer a solution under certain circumstances, there are a number of conditions which need to be in place in order for it to be successful. For example, in very large catchments with a large number of participants, community governance may become unwieldy and difficult. Deciding what instrument is most appropriate in each case will be dependent on a concrete analysis of the costs and benefits associated with each particular policy option (Coglianese & Lazer, 2003). In particular, where there is no need to manage the risks associated with development pressure, costs of implementation may outweigh the potential benefits of avoiding over-allocation.

Conclusion

Tradable responsibilities for nitrogen loss risk management may offer a useful means of managing the risk of over-allocation where there is strong development pressure. Well-designed markets for these responsibilities have the potential to increase the efficiency of resource use. Combining these markets with threshold taxes may both improve overall efficiency and increase the resilience of the resource management system under uncertainty.

While trading systems can theoretically reduce the costs of mitigation, improve efficiency and allow flexibility, they do not offer any solutions to agricultural water quality problems themselves. The development of mitigation solutions comes from individual farmers' understanding of their contribution and through catchment-level learning, adaptation and adoption of innovative solutions. Part of the promise of community governance structures is in their potential for building the social capital required to produce solutions and disseminate information at a catchment level. While well-defined individual responsibilities for risk management are necessary, implying spurious precision to the measurement of compliance is unlikely to improve management effectiveness under uncertainty. Instead, a focus on sharing costs and responsibilities in relation to a common goal offers a more positive framing, and a potential means for reconciling cooperative and competitive drivers.

This proposed model is new and experimental in nature, and therefore requires further work in order to better understand its potential utility or cost-effectiveness. Initially, feasibility would need to be modelled or tested in a simulation. If results are favourable, a small policy pilot or case study would shed light on potential strengths and pitfalls. If this model is found to have merit, it may form a useful addition to the various policy options which may be considered for managing to limits under certain conditions.

References

Ajzen, I., Rosenthal, L. H., & Brown, T. C. (2000). Effects of Perceived Fairness on Willingness to Pay. Journal of Applied Social Psychology, 30(12), 2439-2450.

Anastasiadis, S., Nauleau, M. L., Kerr, S., Cox, T., & Rutherford, K. (2012). Does Complex Hydrology Require Complex Water Quality Policy? NManager Simulations for Lake Rotorua. Motu Working Papers. Wellington, New Zealand: Motu Economic and Public Policy Research.

Bartle, B. (2011). From Dung to Dollars: Lessons for development and implementation of market based trading instruments in agriculture - the case of Lake Taupo nitrogen trading. Thesis presented for Master of Science in Environmental Management. Auckland, New Zealand: University of Auckland.

Bontems, P., & Thomas, A. (2006). Regulating Nitrogen Pollution with Risk Averse Farmers under Hidden Information and Moral Hazard. American Journal of Agricultural Economics, 88(1), 57-72.

Bowles, S. (2008). Policies Designed for Self-Interested Citizens May Undermine "The Moral Sentiments": Evidence from Economic Experiments. Science, 320(5883), 1605-1609.

Bowles, S., & Gintis, H. (2002). Social Capital and Community Governance. The Economic Journal, 112(483), F419-F436.

Burton, R., & Paragahawewa, U. (2011). Creating culturally sustainable agri-environmental schemes. Journal of Rural Studies, 27, 95-104.

Clapcott, J., Young, R., Goodwin, E., Leathwick, J., & Kelly, D. (2011). Relationships between multiple land-use pressures and individual and combined indicators of stream ecological integrity. DOC Research and Development Series. Wellington, New Zealand: Department of Conservation.

Clark, A.J., Nottage, R.A.C., Wilcocks, L., Lee, J.M., Burke, C., Kalaugher, E., Roche, J., Beukes, P., Lieffering, M., Newton, P.C.D., Li, F.Y., Vibart, R., Teixeira, E.I., Brown, H.E., Fletcher, A.L., Hernandez-Ramirez, G., Soltani, A. Viljanen-Rollinson, S., Horrocks, A., Johnstone, P., Clothier, B., Hall, A., Green, S., Dunningham, A., Kirschbuam, M.U.F., Meason, D., Payn, T., Collins, D.B.G., Woods, R.A., Rouse, H., Duncan, M., Snelder, T. & Cowie, B. (2012). Impacts of Climate Change on Land-based Sectors and Adaptation Options. In A. J. Clark, R. A. C. Nottage & D. Hansford (Eds.), Stakeholder report to the Sustainable Land Management and Climate Change Adaptation Technical Working Group. Wellington, New Zealand: Ministry for Primary Industries.

Coglianese, C., & Lazer, D. (2003). Management-Based Regulation: Prescribing Private Management to Achieve Public Goals. Law & Society Review, 37(4), 691-730.

Cox, M., Arnold, G., & Villamayor Tomás, S. (2010). A review of Design Principles for Community-based Natural Resource Management. Ecology and Society, 15(4).

Dietz, T., Ostrom, E., & Stern, P. C. (2003). The Struggle to Govern the Commons. Science, 302(5652), 1907-1912.

Gneezy, U., & Rustichini, A. (2000). A Fine Is a Price. The Journal of Legal Studies, 29(1), 1-17.

Gunningham, N. (2009). The New Collaborative Environmental Governance: The Localization of Regulation. Journal of Law and Society, 36(1), 145-166.

Halpern, D., Bates, C., Mulgan, G., Aldridge, S., Beales, G., & Heathfield, A. (2004). Personal Responsibility and Changing Behaviour: the state of knowledge and its implications for public policy. UK Cabinet Office publication available at www.cdi.mecon.gov.ar/biblio/docelec/dp4105.pdf.

Hickey, C. W., & Martin, M. L. (2009). A review of nitrate toxicity to freshwater aquatic species. Christchurch, New Zealand: Environment Canterbury.

Jack, B. K., Kousky, C., & Sims, K. R. E. (2008). Designing Payments for Ecosystem Services: Lessons from Previous Experience with Incentive-Based Mechanisms. Proceedings of the National Academy of Sciences of the United States of America, 105(28), 9465-9470.

Kallbekken, S., Kroll, S., & Cherry, T. L. (2011). Do you not like Pigou, or do you not understand him? Tax aversion and revenue recycling in the lab. Journal of Environmental Economics and Management, 62(1), 53-64.

Kerr, S., & Lock, K. (2009). Nutrient Trading in Lake Rotorua: Cost Sharing and Allowance Allocation. Motu Working Paper. Wellington, New Zealand: Motu Economic Research.

Ledgard, S. F., Thorrold, B. S., Petch, R. A., & Young, J. (2001). Use of OVERSEER as a tool to identify management strategies for reducing nitrate leaching from farms around Lake Taupo. In L. D. Currie & P. Loganathan (Eds.), Precision tools for improving land management (Vol. 14, pp. 187-194). Palmerston North, New Zealand: Fertilizer and Lime Research Centre.

Lehmann, P. (2012). Justifying a Policy Mix for Pollution Control: A Review of Economic Literature. Journal of Economic Surveys. Journal of Economic Surveys 26(1): 71-97.

McDonald, H., Kennedy, M., Ngawhika, S., & Kerr, S. (2010). Trading Efficiency in Water Quality Markets. Paper presented at the NZARES Conference, Nelson, New Zealand.

McDowell, R. W., Biggs, B. J. F., Sharpley, A. N., & Nguyen, L. (2004). Connecting phosphorus loss from agricultural landscapes to surface water quality. Chemistry and Ecology, 20(1), 1-40.

OECD. (2007). Instrument Mixes Addressing Non-point Sources of Water Pollution: OECD Publishing.

OECD. (2012). Water Quality and Agriculture: Meeting the Policy Challenge. OECD Studies on Water: OECD Publishing.

Ostrom, E. (1990). Governing the commons: the evolution of institutions for collective action. Cambridge, UK: Cambridge University Press.

Ostrom, E. (1996). Neither market nor state for the governance of common-pool resources. [Nemercato ne stato nella gestione delle risorse collettive.]. QA, Questione Agraria(64), 7-39.

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. Proceedings of the National Academy of Sciences of the United States of America, 104(39), 15181-15187.

Ostrom, E. (2008). Frameworks and theories of environmental change. Global Environmental Change, 18(2), 249-252.

Pannell, D. J. (2006). Flat Earth Economics: The Far-reaching Consequences of Flat Payoff Functions in Economic Decision Making. Review of Agricultural Economics, 28(4), 553–566.

Pezzey, J. C. V., & Jotzo, F. (2010). Tax-Versus-Trading and Free Emission Shares as Issues for Climate Policy Design. Environmental Economics Research Hub Research Reports. Canberra, Australia: Crawford School of Economics and Government, Australian National University.

Ramilan, T., Scrimgeour, F. G., Levy, G., Marsh, D., & Romera, A. J. (2011). Simulation of alternative dairy farm pollution abatement policies. Environmental Modelling & Software, 26(1), 2-7.

Randhir, T. O., & Lee, J. G. (1997). Economic and Water Quality Impacts of Reducing Nitrogen and Pesticide Use in Agriculture. Agricultural and Resource Economics Review (April 1997).

Selman, M., Greenhalgh, S., Branosky, E., Jones, C., & Guiling, J. (2009). Water Quality Trading Programs: An International Overview. WRI Issue Brief. Washington DC: World Resources Institute.

Sharpley, A. N., McDowell, R. W., Weld, J. L., & Kleinman, P. (2001). Assessing site vulnerability to phosphorus loss in an agricultural watershed. Journal of Environmental Quality, 30(6), 2026-2036.

Shortle, J. (2012). Water Quality Trading in Agriculture: OECD Publishing.

Wilcock, R. J., Betteridge, K., Shearman, D., Fowles, C. R., Scarsbrook, M. R., Thorrold, B. S., & Costall, D. (2009). Riparian protection and on‐farm best management practices for restoration of a lowland stream in an intensive dairy farming catchment: A case study. New Zealand Journal of Marine and Freshwater Research, 43(3), 803-818.