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Equitably slicing the pie: Water policy and allocation

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ABSTRACT

Non-point source pollution is deteriorating water quality throughout the world. New Zealand is addressing this issue by regulating land-based nutrient losses, with debates over how to allocate limits across a heterogeneous landscape. We develop a spatially explicit economic land use model to investigate efficiency and equity issues from seven approaches to allocate nutrient discharges across two New Zealand watersheds. We find that the preferred allocation differs across land use, land characteristics, and regulation stringency; and that there is no universal 'best' allocation option. Therefore, decision-makers should focus on, at least, efficiency and equity, and on how to compensate those most affected.

1. Introduction

Pollution to air and waterways from land use creates environmental externalities that can be difficult to manage. Environmental externalities are not a new concept but the dilemma of how to deal with them continues to plague governments. An approach more frequently used by governments to manage these externalities is regulation, often because voluntary approaches are not sufficient to address the problem or because imperfect markets fail to send the appropriate signals to those creating the externality.

Most regulatory signals, to date, have involved taxing inputs such as fertiliser and pesticide taxes (e.g. Ekins, 1999; Pearce and Koundouri, 2003) or outputs such as manure (Ekins, 1999; van Eerdt et al., 2005), restricting input use such as fertiliser (Helfand, 1991; Lally et al., 2009) or specifying the use of different technologies or management practices (e.g. Helfand, 1991; Saffouri, 2005; Maine Department of Agriculture, Conservation and Forestry, 2014). However, many of these regulatory approaches have had limited success (OECD, 2012), and more recently greater attention is being paid to more performance-related regulatory approaches such as establishing an overall target or cap on the production of the externality (e.g. pollution) or the overall impact of the externality. Capping greenhouse gas (GHG) (European Commission, 2015) or sulphur dioxide emissions (USEPA, 2015) to the atmosphere, discharges to waterbodies that degrade water quality (NSW EPA, 2015; PA DEP, 2015) or water takes from a river system are some examples.

Once an overall cap is imposed it is then allocated between the various sources of an externality, or users of a resource in the case of water takes. The cap is a legal limit on, for example, the quantity of a certain type of pollutant that can be emitted within a given time period, usually a year. Allocation is a key component of this type of regulation so sources know the limit they are operating to on a firm-level basis. If trading is also allowed, such as in a cap-and-trade programme (Stavins, 2001; Tietenburg, 2006), then the allocated amount provides the baseline to determine how much pollution reduction a firm has to sell or needs to purchase to meet their allocated allowance.

The initial allocation, however, can have significant distributional impacts on wealth as it likely leads to differential compliance costs or a transfer of wealth between existing sources. Therefore, the debate on the appropriate allocation approach to use is complex as each allocation approach will have a different effect on who loses the most today and who potentially loses the most in the future, i.e. have different equity implications. In most cases, it is not possible to determine an allocation approach that both maximises economic efficiency and is considered equitable by all affected parties.

Our paper explores the implications of allocating caps to meet water quality goals using a comparative assessment of two watersheds in New Zealand where statutory processes are in place to improve watershed

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water quality (Environment Canterbury, 2015). Both watersheds face the challenge of how to allocate water pollution caps between the major sources of pollution, in this instance non-point source nutrient discharge. This approach allows us to draw some general conclusions on the choice of allocation approach for non-point source discharges using efficiency and equity considerations and to compare approaches across watersheds and different regulatory stringencies. While these discussions are playing out in New Zealand – a global policy leader for the topic of non-point source allocation – there are broader implications for watersheds globally where non-point pollution discharges are major contributors to water quality degradation and voluntary approaches are not achieving the necessary improvements.

While a global literature exists on allocation for GHG and sulphur dioxide emissions and evaluations of these approaches (e.g. Svendsen, 1998; Ellerman and Buchner, 2008), there is relatively little knowledge on allocating nutrient discharges. This is particularly the case for non-point sources, which are the main sources of nutrient pollution in many watersheds around the world. Any literature that has looked at the allocation of nutrient discharges to regulate non-point sources has done so in the context of a single watershed (Kampas and White, 2003; Kampas and Mamalis, 2006; Kerr and Lock, 2009; WRC, 2007; Anastasiadis et al., 2014; Daigneault et al., 2013, 2014) or in the context of a single sector (e.g. Doole et al., 2013).

The studies outlined above assessed a number of allocation approaches. These typically included variations based on the existing level of pollutant discharge (e.g. grandfathering or emissions-based allocations), approaches that used current or historical pollution rates to assign an allocated amount to individuals based on share of profit, land or discharge, or approaches based on bargaining power. The latter category of allocation tends to be more theoretical and difficult to apply in practice as the information to assign such an allocation is likely unobserved by policy-makers (e.g. Kampas and White, 2003). The use of land characteristics to allocate discharge allowances is also starting to be explored (Carran et al., 2007; Daigneault et al., 2013, 2014) which attempts to better match land use with the capability of the land.

This paper uses a spatially explicit economic land use model to estimate the impacts of nutrient reduction targets under different allocation approaches in two watersheds. This enables the exploration of how a broader range of land and landowner characteristics affect the efficiency and equity implications of allocation choice by expanding the analysis and discussion to multiple watersheds and quantifying impacts for 17 different land uses across 8 soil types. As this paper finds, identifying an 'optimal' allocation that maximises economic efficiency (i.e. first-best) and is considered equitable by all affected parties is not straightforward.

In this paper, we first discuss the rationale for governments to intervene in resource use and allocation of pollutants within regulatory policy. We then present the methodology that includes a description of the allocation approaches assessed, an overview of the economic land use model we use to compare allocation approaches, and the range of mitigation practices used to manage nutrient discharges in two watersheds. Next, we present baseline land use, net farm revenue, and environmental outputs, followed by results from a series of nutrient reduction policy scenarios to identify possible implications of different allocation approaches. The final sections provide a discussion and the implications of our findings.

1.1. Rationale for Government Intervention

To compare between different allocation approaches and consider the implications of these approaches on various pollution sources it is useful to consider why governments intervene in free market economies. Governments tend to intervene in economies to generate revenue for organised economic activity and to increase welfare (Myles, 1995). The motivation to intervene on welfare grounds relates to whether market failure exists or around welfare optimality. This may also relate to inequalities in income distribution or missing, incomplete or imperfect markets (Sandall et al., 2009). The generation of externalities from economic activity is an example of market failure that could prompt government intervention, and nutrient discharges from agricultural systems are one such example. The justification for policy intervention should be based on whether government can actually improve upon the market (Myles, 1995).

When governments intervene there are two aims that may conflict. One is that policy is implemented at least cost or minimum loss to society, i.e. the policy focuses on efficiency. The other is that there is an even distribution of an economy's resources, i.e. the policy focuses on equity. Both are important considerations, and the design of an optimal policy frequently involves the trade-off between efficiency and equity objectives (Myles, 1995), which can be the case with allocation.

While efficiency is a relatively objective measure, equity is far from straightforward and is more subjective. Often equity discussions are based on a set of principles or criteria (e.g. Rose and Stevens, 1993, Pascual et al., 2010 or Kerr and Lock, 2009) when judging the equity implications of a policy. For the public the perceptions of equity and how the benefits and costs of a policy are distributed between affected parties are important (Howe, 2000). Equity therefore 'represents a normative evaluation of the social desirability of economic and non-economic disbursements, both positive and negative' (Rose and Stevens, 1993, p.125). For this paper we focus on the distributive justice (allocation of economic rewards and responsibilities) aspects of equity rather than procedural justice (participation in decision making) (Konow, 2001; Pascual et al., 2010) as economic analysis can provide some insights into the assessment of equity implications related to the distribution of costs. Kampas and White (2003) also noted that policy-makers should take into account the distributional impacts of different allocation approaches, highlighting the importance of considering both efficiency and equity when deciding between allocation approaches.

2. Allocating Natural Resources

While allocation is relevant for most common pool resources (e.g. GHG and sulphur dioxide emissions and nutrient discharges) and also with the use of resources (e.g. the taking of water from rivers), the analysis in this paper focuses on non-point or diffuse pollutant discharges that impact on water quality and how they can be allocated between various sources in a regulatory setting. To date, research and policies to cap pollutants to water have tended to focus on point source discharges and use existing discharges or some variant on these for the initial allocation (Harrison, 1999; National Center for Environmental Economics, 2002; Woerdman et al., 2009; The Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse, 2010; Greenhalgh and Selman, 2012; WRC, 2011). Point source discharges are relatively easy to measure and monitor as they come directly from a single point like a pipe. Non-point source discharges, on the other hand, are not as easy to measure as they enter water via diffuse pathways. As a result, these emissions are typically estimated rather than explicitly measured (Greenhalgh and Selman, 2012).

2.1. The Allocation Debate

The allocation of point source discharges has also not been without debate. For instance, issues with the over-allocation of allowances or permits to emit GHG emissions resulted in significant windfall gains to many sources in both the UK and EU Emissions Trading Schemes (ETS). Windfall gains occur when a source is allocated more allowances than they need and subsequently sell those additional allowances into the market, thereby profiting from the allocation process. This led to widespread criticism of the schemes and there is a relatively large literature exploring this issue and the failures of the allocation processes that created these gains (e.g. Betz and Sato, 2006; Ellerman and Joskow, 2008; Woerdman et al., 2009; de Bruyn et al., 2010; Sartor et al., 2014).