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ECONOMIC INSTRUMENTS FOR NONPOINT SOURCE WATER POLLUTION: OPTIONS FOR THE SWAN-CANNING RIVER SYSTEM

ABSTRACT

The management of nonpoint source water pollution presents an immense challenge to economists and policy makers alike. A complex array of physical, economic, political and institutional barriers lie between theoretically appealing textbook prescriptions and their transition into successful real-world solutions. Underlying beliefs about property rights, interest group politics and the transaction costs associated with designing and implementing successful measures have all played a particularly critical role. Building on the theoretical literature and the lessons provided by the practical use of economic instruments for nonpoint source water pollution management around the world, this paper considers these issues in the context of the Swan-Canning river system in Perth. Four innovative economic instruments for the management of nonpoint source nutrient pollution in that system are discussed: auctioned best management practice payments; best management practice incentive charges; an urban nonpoint source emissions offset bank; and a catchment based licensing/trading program.

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1. Introduction

In 1209, Genghis Khan is said to have decreed that his people keep their stock out of the rivers of Karakorum to preserve water quality (McMullen 2002). Nearly 800 years later, however, the management of nonpoint source water pollution has become a far more challenging proposition for democratic governments around the world. While a combination of traditional regulatory and economic mechanisms have made major inroads into point source water pollution from large enterprises, this is far from being the case with nonpoint source water pollution. The latter is now a very serious problem, for not only are nonpoint sources notoriously resistant to public intervention, but in many cases the aggregate level of pollution they cause significantly exceeds that of point sources (Faeth 2000).[1]

Yet despite widespread acknowledgment of the significant threat posed by nonpoint source pollution for water quality in many major catchments and river systems, generations of successive governments have failed to follow the unlikely lead provided by Genghis Khan. Legislative and regulatory approaches to this form of pollution are thin on the ground. Furthermore, the uptake of economic instruments has been equally unimpressive. While economists have proudly espoused the merits of various economic instruments for addressing point source pollution [2] since Arthur Pigou's seminal 1920 publication *The Economics of Welfare*, their suitability for addressing nonpoint source pollution remains questionable at best.

In general, governments have been reluctant to adopt coercive measures to address nonpoint source water pollution. Suasive measures such as education and information programs aimed at communicating the nature of the environmental problem and encouraging the voluntary adoption of best management practices have historically led the way in most countries, including Australia.

Recent developments, however, suggest a policy change may be in the wind. Several major government decisions indicate that the demand for economic solutions to nonpoint source water pollution (and environmental problems more generally) is gradually increasing.[3] The failure of suasive measures to bring about desired reductions in nonpoint source water pollution, combined with increasing pressure to deliver cost-effective environmental solutions, appear to have shifted the spotlight onto potential economic solutions.

While demand for new and improved economic remedies to nonpoint source water pollution has grown, it is fair to say that supply has not. Indeed, it is just twenty years since the first systematic analysis of the role of economics in nonpoint pollution control was undertaken by Griffin and Bromley (1982) and most analysis in this field has only occurred during the last decade. During this time a number of economic instruments have been proposed. Most of these have failed to make the transition from the 'ideal world' that forms the basis for their examination in the economic texts to widespread practical application in the real world.

[1] It is estimated that 82 per cent of total nitrogen discharges and 84 per cent of total phosphorus discharges into waterways in the United States come from nonpoint sources (Faeth 2000). There is little reason to doubt that the comparative figures for Australia would be much different. NSW EPA (2002) for instance, suggest that nonpoint sources contribute 80% of phosphorus and 90% of nitrogen pollution in the Warragamba catchment in Sydney.

[2] It should be noted that despite the continual affirmations of many economists that economic instruments provide the most efficient approach to managing point source water pollution, the practical experience with emission charges and tradable emission permits may be considered modest at best (Hahn 1989; Hahn and Hester 1989; Stavins 2001).

[3] This may be evidenced by the $10 million portion of the 2001 National Action Plan for Salinity and Water Quality earmarked for pilots of various economic instruments around Australia.

This paper looks at some of the reasons why these instruments have failed to attract the eye of environmental managers and policy makers. More specifically, the physical, economic, political and institutional challenges posed by excessive nonpoint source water pollution are examined and several potential solutions are considered.

The paper is presented in three general parts. The first part, Section 2, discusses the nature of nonpoint source pollution and the implications of the inherent characteristics of this form of pollution for policy design.

The second part comprises Sections 3 and 4 and investigates the contribution that economics has made to the management of nonpoint source water pollution. Section 3 provides a summary of a range of economic instruments for addressing nonpoint source water pollution. This section does not aim to provide a comprehensive discussion of the theoretical merits of all of these instruments, focusing instead on those instruments that have either received the greatest attention in the economic literature or have been widely adopted in practice. Where applicable, a discussion of the practical performance of these instruments is provided. Section 4 looks at the lessons that may be gleaned from the real world approach to nonpoint source water pollution. It considers why governments have generally preferred suasive measures to economic instruments. Three elements in particular are proposed as having played a critical role in shaping the way in which policy makers and governments address the problem of nonpoint source water pollution – underlying beliefs about the property rights to pollute, interest group politics and the high transaction costs associated with designing and implementing efficient and environmentally effective economic instruments.

The third part of this paper comprises Sections 5 and 6 and considers the contribution that economic instruments may make to the management of excessive nonpoint source nitrogen and phosphorus emissions into the Swan-Canning river system in Perth, Western Australia. Drawing on the lessons provided earlier in the paper, four innovative instruments are proposed. The first two of these instruments are aimed at increasing the adoption of best management practices (BMP) by nonpoint source polluters. The first, a system of auctioned grants for the voluntary improvement of management practices is based on the current underlying beliefs regarding the property rights for nonpoint source emissions. The second, a system of BMP incentive charges, is based on the principle that the nonpoint source polluters should pay for their emissions. While it is clear that such an approach would initially face strong opposition from nonpoint source polluters and their lobby groups, the design of this instrument is aimed at mimimising this opposition through gradually phasing in BMP requirements and charges for non-compliance and recycling revenue to the affected industry to reward positive environmental action. The third economic instrument to be examined is a system of emission offsets targeted at urban residential nonpoint emissions. In recognition of the potential for high transaction costs to erode the benefits from such a scheme, a regionally managed offset bank system is considered. A fourth instrument, based on converting groups of individual nonpoint source polluters into single point sources, although not currently practical due to the high policy transaction costs associated with introducing the appropriate institutional arrangements for its successful implementation, is also discussed due to its potential in the longer term.

In Section 7, conclusions are drawn on the contribution of economic instruments to nonpoint source water pollution in general, and more specifically for the Swan-Canning river system.

2. The nature of nonpoint source pollution

The inherent characteristics of nonpoint source pollution and the implications of these characteristics for policy design are well documented (Shortle and Horan 2001; Shortle and Abler 1997; Shortle and Abler 1994; Tomasi et al 1994; Dosi and Moretto 1994; Dosi and Tomasi 1994; Xepapadeas 1994).

Unlike emissions from point sources, which enter the river system at discrete and identifiable locations, emissions from nonpoint sources follow unobservable, indirect and diffuse pathways into the system. Emissions from a farm or urban parkland are typical examples. As a consequence, assigning responsibility for and measuring emissions from individual nonpoint source polluters becomes both extremely difficult and uncertain.[4] This uncertainty surrounding the size of emissions from individual nonpoint sources and the degree of environmental damage caused by these emissions poses a considerable problem for the policy maker. As Shortle and Abler (1997) suggest, it is politically difficult for the policy maker to require nonpoint source polluters to undertake potentially costly actions to reduce emissions when it is difficult to demonstrate the public benefits of such actions. In some cases it may be highly uncertain as to whether the public benefits of such actions even outweigh the costs. Furthermore, experience in the United States suggests that such requirements would not only need to match up with general perceptions of fairness but also be legally defensible. As will be seen in the following section, these problems have played a major role in shaping policy actions to address nonpoint source water pollution, shifting the focus away from instruments based on the polluter pays principle towards moral suasion and government funded subsidisation of improved management practices.

A further difficulty for the policy maker relates to the characteristics of the nonpoint source polluters themselves. In most river systems there are likely to be a large number of small heterogeneous nonpoint sources. This has a number of implications for instrument choice. First, individual emissions cannot be inferred from ambient nutrient concentrations as the later are determined by the joint contributions of many unmeasured nonpoint sources. Second, the transaction costs associated with monitoring emissions at the individual polluter level (if this were to become technically feasible) and of administering measures aimed at reducing nonpoint source pollution are very high. These costs increase even further the greater the degree of heterogeneity between nonpoint source polluters. Unfortunately, in most instances variation in site characteristics has a considerable influence on the likely environmental effectiveness and cost of specific actions. In many cases this precludes the use of uniform technological prescriptions, instead demanding the adoption of actions tailored to the characteristics of each individual polluter.

A final complication that arises with nonpoint source pollution relates to the uncertainty surrounding the way in which nonpoint source polluters will respond to given policies. One of the main arguments that supports the use of economic instruments over regulatory approaches to pollution control is that the polluter holds a greater degree of information about their marginal abatement costs than does the policy maker. With large point source polluters this assumption is likely to hold. It is commonly accepted that they have the level of technical expertise and resources available to determine the various types of abatement options available to them. They

[4] While science is making constant advances in terms of being able to measure the level of emissions from nonpoint sources, the current reality is that doing so in most cases is still either technically infeasible or prohibitively expensive.

also have a reasonable understanding of the cost of implementing these options. The same cannot be said for nonpoint source polluters. Take the example of a small farm. Without ever having been required to consider or undertake pollution abatement it is unlikely that the farmer will have made any effort to try to estimate the costs of various abatement options. Moreover, they may not even be aware what options are available to them. The level of technical expertise and time required to make such estimations is simply not available in most cases. As a consequence, polluters cannot be counted on to respond to economic incentives to undertake pollution abatement. This suggests that an education/information program and ongoing technical extension aimed at polluters should be included in any policy mix that includes economic instruments for nonpoint source pollution control.

Together, these characteristics of nonpoint source water pollution present a considerable challenge to both policy makers and economists. Nevertheless, over the last twenty years a substantial and growing body of literature has developed, seeking to identify suitable economic instruments for managing nonpoint source water pollution. The following section provides a review and a critical analysis of the literature. Particular focus is placed on the potential of a number of proposed economic instruments to translate theory into practice. Where applicable, a discussion of the practical performance of these instruments is also provided.

3. Economic instruments for nonpoint source pollution control

Economic instruments aimed at addressing point source water pollution, such as emission charges and tradable emission permits, are designed to use measured emissions from individual polluters as their compliance base. This luxury is not available when addressing nonpoint source water pollution. Rather, an alternative compliance base that is both correlated with actual emissions from individual polluters and enforceable is required. The search for a suitable compliance base has been the focus of attention in the nonpoint source literature. Three basic options have been proposed: inputs or management practices that are correlated with emission levels; ambient pollution levels; and sophisticated emission proxies.

A range of economic instruments for nonpoint source water pollution control, using these suggested compliance bases have been proposed. These include: incentive mechanisms aimed directly at input use, such as input taxes and subsidies; incentive mechanisms based on ambient pollution levels, such as ambient taxes, random fines and nonpoint source tournaments; and instruments based on estimated emissions from nonpoint source polluters, such as estimated emissions trading (point/nonpoint trading).[5] A summary of these instruments is provided in Table 1. A discussion of a number of these instruments and how they have fared in practice is provided below.

This section does not aim to provide a comprehensive discussion of the theoretical merits of all of these instruments, focusing instead on those instruments that have either received the greatest attention in the economic literature or have been widely adopted in practice. A more detailed discussion of a broader range of instruments may be found in Shortle and Abler (1997) and Shortle and Horan (2001).

[5] The use of emission proxies as a base was first proposed by Griffin and Bromley (1982). The paper in which they proposed this ‘indirect approach’ is considered to be the first systematic analysis of the role of economics in nonpoint pollution control.

Table 1 Nonpoint pollution control instruments

	Compliance measure		
Mechanism	Inputs/Practices	Emissions Proxies	Ambient Taxes
Taxes/subsidies	Charge on fertiliser purchase. Cost-sharing or other subsidies for inputs or practices that reduce pollution. Crop land retirement subsidies.	Charges on modeled nutrient loadings. Charges on nutrient applications in excess of crop needs. Charges on estimated net soil loss.	Ambient taxes.
Standards	Restrictions on fertiliser application rates. Mandatory use of pollution control practices.	Restrictions on modeled nutrient loadings. Regulations on nutrient applications in excess of crop needs.	
Markets	Input trading.	Estimated emissions trading – point/nonpoint and nonpoint/nonpoint.	
Contracts	Land retirement contracts. Conservation or nutrient management practices contracts.		
Liability Rules	Negligence liability rules.		Strict or negligence liability rules.

Source: Shortle and Horan (2001)

Input based instruments

Inputs into the production process and management practices undertaken by the polluters are often correlated with emission levels (Shortle and Horan 2001). This relationship enables inputs and management practices to be used as an alternate base for pollution control instruments.

Several studies have provided theoretical support for the use of input taxes for nonpoint pollution control. Griffin and Bromley (1982) are accredited as being the first to examine the theoretical merits of such an approach. Assuming nonstochastic nonpoint externalities and perfectly estimated emissions they demonstrated that taxing inputs (or management practices) that are known to increase nonpoint source pollution (or conversely a subsidy on all inputs or management practices which will reduce it) is theoretically capable of producing the same outcomes as optimal pigouvian taxes and subsidies. Shortle and Dunn (1994) considered the efficiency of such an approach taking into account the uncertainty that surrounds nonpoint source emissions and their impact on ambient pollution levels and demonstrated that an efficient solution could still be achieved as long as firm specific taxes were applied on all inputs that contributed to the ambient pollution level.

Delivering the efficiency benefits that such an approach offers in theory is extremely difficult in practice. It is both administratively complex and costly to monitor and verify all the inputs and management practices for each nonpoint source polluter in a catchment. Moreover, uncertainties surrounding the relationship between input use and the actual impact on ambient pollution levels are often considerable (Shortle and Abler 1997).

In recognition of these practical obstacles a number of related options have been suggested. The first involves truncating the input tax/subsidy system to a subset of inputs/management practices that remain closely correlated with the level of emissions but are less costly to monitor. While reducing policy transaction costs [6] this approach may have undesired 'spillover' impacts such as encouraging substitution into other environmentally damaging inputs (Randhir and Lee 1997). The second option involves shifting the basis for compliance monitoring away from observed use by the nonpoint source polluter to the point of sale of the input or even to the manufacturer of the polluting product covered by the scheme (Shortle and Horan 2001). While this approach has been used to control pesticide application it has received relatively little support in the nonpoint source economic literature as it fails to provide a transparent signal to nonpoint source polluters about the external costs of their production decisions (Shortle and Horan 2001).

Despite their theoretical appeal, input taxes aimed at reducing nonpoint source water pollution have rarely been adopted in practice. The Scandinavian counties have been the exception. Sweden imposes significant sales taxes on commercial fertilisers containing nitrogen and phosphorus while Norway has previously imposed sales taxes on the nitrogen component of fertiliser (Vatn et al 2002; Shortle and Abler 1997). To date, the imposition of fertiliser taxes in these countries has failed to achieve the desired environmental outcomes (Ecotec 2001). This lack of success may be attributed to the fact that fertiliser consumption tends to be relatively unresponsive to price changes (i.e. it is highly price inelastic). Subsequently, very large taxes are required if the desired reductions in fertiliser applications are to be achieved. Imposing taxes of this magnitude tends to generate strong opposition from affected stakeholders, however, and rarely garners the necessary political support for their introduction. In Norway for instance, it was estimated that a tax of between 100 and 300 per cent would be necessary to reduce the incidence of algal blooms (Simonsen 1989 cited in Vatn et al 2002). Strong lobbying from farmer groups concerned about the impact of the tax on their financial viability resulted in a fertiliser tax of just eight per cent being introduced in 1992. After continuing parliamentary debate (stretching over nearly a decade) it was decided that the eight per cent tax increased farm costs by too much and was failing to provide a sufficient level of environmental benefit. It was finally removed in 1999 (Vatn et al 2002).

Analogous to input taxes, input subsidies (in their textbook form) are payments made to a polluter for each unit of emissions that they reduce. In comparison to input taxes, economists have tended to strongly oppose the adoption of input subsidies as a form of pollution control. Their primary concerns relate to the suggestion that subsidies may lead to long-run dynamic inefficiency and actually increase overall levels of pollution by providing a financial stimulus for the polluting industry and encouraging new entrants (Baumol and Oates 1988; O'Shea 2002). A further objection is based on the grounds that subsidies are diametrically opposed to the polluter pays principle (Panayotou 1998).

The term subsidy is used much more widely in the popular economic literature, however, and refers to any form of payment made to a polluter to encourage them to undertake management actions that will reduce their level of emissions (Industry Commission 1997; Barde 1997). In this broader sense subsidies include cost sharing, grants, low interest loans and tax

[6] As explained in section 5 of this paper, policy transaction costs include the costs associated with: research, information gathering and analysis; enactment of enabling legislation (including lobbying and public participation costs); design and implementation of the policy (including costs of regulatory delay); support and administration of the on-going program; contracting costs; monitoring/detection costs (of both the efficiency of the outcome and the level of compliance); and prosecution/inducement costs (McCann and Easter 1998; 2002).

concessions. If considered in these forms, subsidies are one of the most commonly adopted economic instruments for nonpoint source pollution management.

In the United States, for example, a number of voluntary programs that provide cost-share payments for approved management practices are run at both the federal and state level and form the backbone of efforts to reduce nonpoint source water pollution in that country. These include the: continuous signup component of the Conservation Reserve Program (CRP); Environmental Quality Incentives Program (EQIP); Agricultural Management Assistance Program (AMA); Wetlands Reserve Program (WRP); and the Soil and Water Conservation Assistance Program (SWCA). Cost-share assistance ranges from 50 per cent for the continuous signup component of the CRP through to 75 per cent for the other programs (USDA 2002a; USDA 2002b; USDA 2002c; USDA 2002d; USDA 2002e).[7]

The approach adopted in Australia is similar to that in the United States with cost-share assistance and other forms of subsidies (including tax deductions and low interest loans) provided through the Commonwealth funded Natural Heritage Trust (NHT).

The strong uptake of this form of instrument might suggest that it is capable of delivering on both environmental and economic grounds. This is not necessarily the case. As cost-share programs are not directly aimed at internalising the externality caused by the polluter's management decisions they are unlikely to deliver the same level of economic efficiency gains as many other economic instruments. Moreover, being voluntary, they do not guarantee significant environmental improvements, as the proportion of total polluters participating in the program may be small.[8]

So, why has this form of instrument tended to spearhead the attack on nonpoint source water pollution? The answer is essentially political. Subsidies are not based on the polluter pays principle. As such, they enjoy strong support from affected nonpoint source polluters and avoid political confrontations regarding changes to property rights. The impact that underlying beliefs about property rights has on instrument choice is discussed in greater detail in Section 4 of this paper.

Ambient based instruments

Ambient based instruments treat the problem of nonpoint source water pollution as one of 'moral hazard' [9] and are aimed at inducing efficient pollution abatement decisions from individual nonpoint source polluters. Since first suggested by Kathleen Segerson in 1988 a substantial body of theoretical literature that explores various types of ambient based economic instruments has evolved.

Segerson (1988) proposed that nonpoint source polluters receive subsidies if ambient pollution levels were measured to be below a desired target level and be taxed if they were

[7] The SWCA is the only on of these programs subject to competitive bidding. While such bidding is a feature of the CRP it does not apply for the continuous signup portion of the program. The competitive bidding element of the EQIP was removed as part of changes made in the 2002 US Farm Bill.

[8] Polluters may be unwilling to participate in such programs due to the amount of time and effort required on their behalf (Humphries et al 2002).

[9] Moral hazard assumes that the characteristics of individual polluters are assumed to be known with certainty while their actions, including the level of pollution abatement they undertake, are not.

measured to be above. Under a number of restrictive conditions[10] it was demonstrated that this model could deliver an economically efficient environmental outcome. In practice, Segerson's model faces a number of barriers. The largest relates to the perceived fairness of applying this approach. The model does not discriminate between those who have undertaken costly actions to reduce their emissions and those who have not. This feature would be likely to engender considerable opposition from affected polluters and acts to limit the political appeal of the instrument.

Other ambient based economic instruments face similar barriers. One such approach, developed by Xepapadeas (1991), is centered on a system of random fines whereby a polluter is chosen at random and fined if ambient pollution levels are measured to be above the desired target level. While this instrument would provide some incentive to polluters to jointly undertake pollution control to avoid the risk of being punished, it is also politically unpalatable. In the extreme it could result in firms who have met best management practice pollution abatement requirements being fined while others who have undertaken no pollution abatement remain unpunished (Govindasamy et al 1994).

In response to the potential problems faced by the ambient based instruments proposed by Segerson and Xepapadeas, Govindasamy et al (1994) proposed the use of nonpoint tournaments whereby polluters are ranked based on the level of pollution control they have undertaken. Bottom ranked polluters are penalised if ambient pollution levels are above socially desired levels and top ranked polluters are rewarded if ambient pollution levels are below socially desirable levels. By basing penalties and rewards on actions rather than emissions the information requirement for the policy maker is substantially reduced and political acceptability increased. How polluters would be ranked and whether or not they would find this system acceptable remains uncertain. Moreover, whether it would deliver environmental benefits greater than the transaction costs associated with implementing such a radical system is questionable.

Not surprisingly, despite having some theoretical appeal, ambient based instruments are yet to be applied in practice (Shortle and Abler 1997). The high cost and uncertainty of monitoring ambient pollution levels and the significant political limitations of the approach have been put forward as two likely reasons for this outcome (Shortle and Abler 1997; Shortle and Horan 2001). Lack of stakeholder and political support appears to be a severe concern. It stems from concerns regarding fairness (or lack of it) as individuals are taxed based on the actions of the group rather than their own pollution control efforts. These and other concerns have led analysts to suggest that ambient instruments may only be suitable in restrictive circumstances where: the watershed is small; polluters are homogenous; water quality may be monitored reliably and at low cost; and where there are short time lags between the polluting activity and the impact on water quality (Weersink et al 1998 cited in Shortle and Horan 2001). Given the restrictive nature of these conditions and the political opposition that such an approach would be likely to engender, ambient based instruments must be considered to be of no practical interest at this stage.

Emission proxies

Emission proxies are estimates of emissions based on observed behaviour. Simple proxies may be based on the expected emissions resulting from a single input use, such as fertiliser

[10] For instance, it is assumed that polluters are able to observe and control their own emissions as well as having knowledge on how the actions of other polluters will affect the ambient pollution level.

application. More sophisticated proxies may use models which try and replicate the impact that various management choices will have on emissions in a particular physical environment.[11] If emission proxies are considered to closely predict actual emissions the range of economic instruments used to address point source pollution may also be considered by the policy maker for the management of nonpoint source pollution. These include emission charges and emissions trading. Emission charges based on proxies have yet to be adopted in practice. Uncertainty over the ability of emission proxies to reflect actual emissions and strong resistance from nonpoint source polluters to such an approach substantially erode their political appeal. In comparison, nonpoint source emission trading based on emission proxies has been applied in practice, predominantly in the United States. A detailed discussion of this instrument is provided below.

Nonpoint source emissions trading

The concept of nonpoint source emissions trading has evolved as a consequence of both the institutional context of the United States water pollution control laws and the failure of early point source emissions trading programs for water pollution control (Shortle and Abler 1997). Emissions trading programs to address point source water pollution were first introduced in the United States in response to stringent regulations placed on emissions from point source polluters under the *Federal Water Pollution Control Act 1972*.

The first of these trading programs, the Fox River tradable discharge scheme, was introduced in 1981 amid considerable fanfare from environmental economists. Modeling undertaken prior to the implementation of the scheme suggested that trading would deliver annual cost savings of around US$7 million over a more prescriptive regulatory approach (US EPA 1996). These savings did not eventuate. In fact, the actual cost savings have been minimal and in all the time the scheme has been in existence just one trade has taken place (Hahn 1989; Kraemer and Banholzer 1999). The problems of the Fox River scheme have been repeated in point source water pollution markets all over the United States (Stavins 2001; Environomics 1999). 'Thin' markets [12] and a lack of variability in polluters' marginal abatement costs were considered to be two of the reasons for this outcome.

The inclusion of nonpoint source polluters in tradable emission markets was seen as a possible solution to both of these problems and as a way of improving the overall efficiency of the market (Shortle and Horan 2001). The potential for improved efficiency derives from both the increase in market 'depth' as a result of including a large number of nonpoint source polluters and from the fact that nonpoint source polluters tend to have lower marginal abatement costs than point source polluters (US EPA 1996; Shortle and Abler 1997). While point source polluters may require the introduction of new and potentially expensive technological methods to reduce their emissions, nonpoint source polluters may be able to reduce theirs through relatively simple and low cost changes in management practices, such as reducing fertiliser application rates. The fact that point source emissions have been the focus of regulators attentions over a longer period than nonpoint source emissions has also played a role in creating the difference in marginal abatement costs. While the easy yards may have already been taken with point source emission reductions this may not be the case for nonpoint sources that have largely remained off the environmental authorities radar screen.

[11] The Universal Soil Loss Equation is an example of a sophisticated proxy.

[12] 'Thin' markets refers to markets where there are insufficient potential trading partners to generate the level of trade necessary to deliver efficiency gains.

These factors provide a strong economic rationale for including nonpoint sources in tradable emission markets. The ability to translate the potential benefits that such a scheme offers into practice is not without considerable challenge, however. In particular, complex technical, economic, scientific, regulatory and institutional issues all need to be addressed.

The most important of these issues relates to how to design the system so that trades can occur between polluters with measurable emissions (point source polluters) and those with unobservable emissions (nonpoint source polluters). That is, how can the market operate when the same commodity (observed emissions reductions) cannot be traded? To get around this problem it is necessary for the policy maker to estimate what the likely emissions reductions of different management practices for nonpoint sources would be (through fieldwork and detailed biophysical models) and to grant permits on this basis. This poses a considerable challenge, may be costly to undertake and may not provide proxies that can be adopted with a great level of certainty. Complicating matters even further is the fact that in most cases there is likely to be a great degree of heterogeneity between nonpoint sources. Due to temporal and spatial differences the emissions reduction as a result of the same change in management practices by one source may differ from another source elsewhere in the catchment.

One way of addressing these concerns is to introduce trading ratios for trades between point and nonpoint polluters. In general, trading ratios need to be carefully determined as they have the potential to erode incentives for trade between point and nonpoint source polluters if set too high. If trading ratios are set too high, transaction costs are likely to increase as it becomes more difficult for willing trading partners to be found in the market. High transaction costs are important to the efficient operation of the market as they have the potential to diminish cost-efficiencies gained through trading. While this is an issue for all trading schemes it is of particular importance to point/nonpoint trading schemes due to the large number of small nonpoint source market participants. Given that changed management practices for most nonpoint source emitters is only likely to result in a small emissions reduction (compared to that required from point source emitters) it is likely that the point source will need to coordinate trades with multiple nonpoint source emitters.

Even when transaction costs are low, there are no guarantees that a sufficient level of trade to generate efficiency gains will automatically result. Where nonpoint source polluters are not subject to the same regulatory constraints as the point source polluters they may be hesitant to enter into a trade, even where it would be consistent with their profit maximising behavior, simply to avoid the time and cost taken up by negotiations with the point source emitter. And where trades are entered into, questions regarding the security of environmental outcomes that they will deliver remain as nonpoint source polluters may fail to undertake the management actions required to deliver the agreed reduction in emissions. [13]

As a result of the range of scientific, economic and regulatory challenges outlined above, fully fledged nonpoint emissions trading programs, whereby trading occurs until an equilibrium market price is achieved, are yet to be applied in practice. While interest in including nonpoint source water polluters in existing point source emission trading programs in the United States has

[13] To avoid this potential problem the US EPA (1996) suggest that compliance for any nonpoint sources should be determined by a regulatory authority and based on reasonable assurance that the nonpoint sources will comply with the provisions of the trade. Reasonable assurance, as defined by the US EPA (1996) means that the proposed nonpoint source controls are: technically feasible; specific to the pollution of concern; implemented according to a schedule and within a reasonable time period; and supported by reliable delivery mechanisms and adequate funding.

increased dramatically over the last decade, in practice trading schemes have tended to amount to little more than relatively simple offset schemes. In these schemes any increase in emissions resulting from new developments or expansions of existing developments are able to offset by reducing off-site nonpoint emissions. A summary of operational offset schemes in the United States is provided in Table 2.

While the mechanics of offset schemes appear relatively straight forward, their performance in practice has been fraught with difficulties. In general, the level of trading activity has failed to live up to the expectations of their proponents. In most instances few offset trades have been made and cost-savings have remained relatively small (Environomics 1999). This is exemplified by the performance of the Lake Dillon effluent trading scheme where the first offset trade took fifteen years to complete (see Box 1).

A report prepared by Environomics (1999) for the US EPA advances a number of reasons for the relatively poor performance of offset schemes. These include: low interest in trading from point source polluters; difficulties in identifying a sufficient number of suitable nonpoint sources; high administrative costs; coordination difficulties due to the large number of heterogeneous participants; uncertainty regarding trading rules; reluctance to negotiate trades due to the cost involved; a break down in negotiations between key stakeholders in setting trading ratios; a lack

Table 2 US tradable emissions permits offset schemes for water pollution

Waterbody/location	**Pollutant**	**Number of participants**	**Trade type/ratio**
Boulder Creek, Colorado	Ammonia, temperature, pH	City of Boulder and various nonpoint sources	Point/Nonpoint (na)
Cherry Creek Basin, Colorado	Phosphorus	12 WWTP’s and various nonpoint sources	Point/Point and Point/Nonpoint (up to 3:1)
Lake Dillon, Colorado	Phosphorus	4 POTW, several WWTP’s and various nonpoint sources	Point/Nonpoint (2:1)
Kalamazoo River, Michigan	Phosphorus	Various nonpoint sources, 1 POTW’s and non-Govt environmental groups	Point/Nonpoint (2:1 and 4:1)
Tar-Pamlico, North Carolina	Nitrogen and phosphorus	Large – point and nonpoint	Point/Point and Point/Nonpoint (2:1)
Specialty Minerals, Inc. (Massachusetts)	Temperature	Specialty Minerals, Inc. and one town	Point/Nonpoint (2:1)
Southern Minnesota Beet Coop. Trading Program (Minnesota)	Phosphorus	Beet Coop. and multiple farmers	Point/Nonpoint (2.6:1)
Rahr Malting – Minnesota River	Phosphorus/CBOD	Rahr Malting Plant and numerous nonpoint sources	Point/Nonpoint (2:1) CBOD/Phosphorus (8:1)

Source: Environomics (1999). Note: POTW is a privately owned treatment works and WWTP is a wastewater treatment plant.

of information for potential participants about the benefits of participating in a trading scheme; strict rules regarding eligibility to participate in the trading scheme; and difficulties in developing emissions proxies for nonpoint source polluters.

Although far less widely used, offset schemes in Australia have also failed to deliver the expected level of economic and environmental gains. This is demonstrated by the performance of the ongoing Environmental Improvement Initiative (EII) offset scheme in Geographe Bay, Western Australia. Under the EII program the Busselton wastewater treatment plant has sought to fund reductions in nitrogen emissions on nearby farms rather than reduce their on-site emissions. Early estimates suggested that off-site emissions reductions could be made for $1 million, less than a quarter of the $4.5 million required on-site for a similar reduction in emissions (Humphries et al 2002). However, despite the selective targeting of high emission industries, few offset projects have been undertaken and in the first two years of the project just $24 000 has been spent on offset projects. Minimal improvements in water quality have resulted (Humphries et al 2002).[14] The reasons for the poor performance of the EII are similar to those experienced in the United States. Perhaps most importantly, it is clear that nonpoint source polluters are reluctant to participate in the program as they face no regulatory requirement to do so (Humphries et al 2002). [15]

Box 1 The Lake Dillon effluent trading program

The Lake Dillon effluent trading program is the second oldest effluent trading program in the United States and the first involving nonpoint source polluters. In 1982, in response to concerns about water quality in Lake Dillon a cap was placed on the total amount of phosphorus from four point source municipal wastewater treatment plants that could enter the lake. In 1984 it was decided that these point sources could increase their phosphorus emissions into the lake as long as they paid for reductions in an equivalent amount of phosphorus emissions into Lake Dillon from nonpoint sources elsewhere in the lake's drainage basin. The primary source of nonpoint source pollution came from privately owned septic tanks of sewage disposal units. It was estimated that switching a home over to the wastewater facility would reduce emissions by one pound. Point sources were prohibited from trading any surplus pollution allowances and trades with nonpoint sources were based on a trading ration of 2:1 (two pounds of estimated phosphorus reduction from a nonpoint source allowed point sources to increase their emissions by one pound).

While estimates made prior to the introduction of the scheme forecast strong demand for emission credits, the first trade did not occur until 1999 - fifteen years after the commencement of the program. To this point in time the lack of trade has been primarily attributed to low demand for emission reduction credits from the wastewater treatment facilities that chose to upgrade their facilities rather than offset increased emissions. A number of secondary reasons have also been suggested for the lack of trade including non cost-minimising behavior from wastewater treatment plants and the significant restrictions that were placed on trading by the committee responsible for accrediting trades reduced interest in trading (Environomics 1999).

The impetus for trading in 1999 was the proposed development of a ski resort and village in the lake's drainage basin at Copper Mountain. Once it was established that the development would not be allowed to increase net phosphorus emissions into Lake Dillon, significant demand for phosphorus credits was ensured. It was agreed that the developers of the Ski resort (Intrawest) would pay for eighty homes to shift from the septic systems to the wastewater treatment plant (an eighty pound reduction in phosphorus emissions). The cost of $500 000 to the developers was considerably lower than all other alternatives. As well as providing for a cost-effective solution to the ski resort the trade also has the potential to reduce net emissions into lake Dillon by forty pounds (as a result of the 2:1 ratio)

[14] During this time more than $300 000 has been spent on the administration of the program ((Humphries et al 2002).
[15] Other reasons for the poor performance of the EII scheme to date include: the requirement that nonpoint sources share half of the costs of offset projects; lack of knowledge, time and interest by farmers; and suspicion of 'outsiders' (Humphries et al 2002).

The nature of the problems faced in both the United States and Australia highlight the difficulties that confront the policy maker in applying offset schemes and more advanced forms of nonpoint source emissions trading. Moreover, they suggest the need for more considered preparation, design and application of this instrument if it is to deliver anywhere near the potential cost savings it offers in practice.[16]

Other instruments

One of the most innovative and successful approaches to the management of nonpoint source water pollution is the Grassland Area Farmers Tradable Loads Program (GAFTLP) in the San Joaquin Valley of California. Under this scheme nonpoint source selenium emissions from farms in six agricultural districts are collected in main drains, which are then measured and treated as point sources. Monthly and yearly selenium load allocations (SLAs) [17] are apportioned among the six districts. If the aggregate selenium emissions from the group of nonpoint source emitters in a district (as measured in the main drain) exceed the SLA then the district authority is required to pay a fee.[18] Conversely, if aggregate emissions are below the SLA then the district authority receives a rebate from the regional drainage entity (Environomics 1999). As fees and rebates are levied on the district authority it is up to them to provide incentives for individual farmers to control their nonpoint selenium loads (Woodward et al 2002).

In addition to the fee and rebate system, each district may also trade their SLAs. [19] If an irrigation district exceed their SLA they may purchase rights from another district to avoid paying the fee. Conversely an irrigation district with emissions below their SLA may sell part of their SLA to another district if they can receive a better price than that provided by the rebate (Woodward et al 2002). By converting a number of nonpoint sources into one entity this system closely resembles the traditional point source emission trading program.

As of February 2000 nine trades had been undertaken in the GAFTLP and selenium emissions had been substantially reduced (Woodward et al 2002; Young and Karkoski 2000). Three major factors have contributed to the relative success of the scheme. First, emissions have been able to be accurately monitored at relatively low cost. Second, it appears that there are sufficient economic drivers for trade (the marginal costs of abatement differ between agricultural districts). Finally, transaction costs have been kept low due to the small number of participants in the market.

These conditions are only likely to be found in a limited number of circumstances. There may be few instances where nonpoint sources can be converted into point sources in the manner adopted in the GAFTLP. Where a conversion is technically feasible such a system warrants detailed consideration.

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[16] A pilot offset scheme is currently being developed in the lower Hawkesbury-Nepean River in Sydney (the South Creek pilot). While the design of this scheme is still being undertaken it appears that many of the lessons from the United States experience have been taken on board. In particular, a lengthy public information and education program and the proposal to use offset funds (rather than have individual offset agreements) are likely to improve the economic and environmental effectiveness of the pilot (NSW EPA 2002).

[17] The SLA is determined by mutual agreement among affected parties. It is based on historical discharge for the first two years and then decreases by five per cent per year for the following three years (Young and Karkoski 2000).

[18] The fee is equivalent to around $3/acre and is unlikely to provide a strong economic incentive to reduce emissions (Young and Karkoski 2000).

[19] Trades in the Grassland Areas Farmers Tradable Loads Program are made ex post, based on selenium loads that were actually discharged (Woodward et al 2002).

4. Lessons from the real world approach to nonpoint source water pollution

While the economic literature continues to focus on the theoretical merits of various ambient and input based economic instruments, policy makers have failed to warm to such an approach. Concerns regarding the political and administrative difficulties associated with applying such instruments have tended to over-shadow any potential efficiency gains they may offer. Indeed, on the whole the policy landscape for addressing nonpoint source water pollution has generally been characterised by a reluctance to adopt any forms of economic instruments at all. The clear preference in most cases has been to rely on suasive measures such as community education and information programs.

On the surface, this outcome may be seen to have arisen for two main reasons. First, it is only in the last twenty years or so that the scope and magnitude of nonpoint source polluters to water pollution has been recognised. That is, it is a relatively new political issue and developing efficient solutions will take time. Second, the inherent characteristics of nonpoint source pollution do not lend themselves to easy solutions (either theoretically or in practice) and as such have often been placed in the 'too hard' basket (Dosi and Tomasi 1994). Digging a little deeper, however, suggests that underlying beliefs about the property rights to pollute, the associated interest group politics and the high transaction costs of designing and implementing efficient and environmentally effective economic instruments have also played a critical role in shaping the way in which policy makers and governments address the problem of nonpoint source water pollution. These two issues are discussed in greater detail below.

Underlying beliefs about property rights and interest group politics

Underlying beliefs about property rights have undoubtedly played a major role in shaping the way in which governments around the world have approached the problem of nonpoint source water pollution. Although emissions from large point source polluters such as sewage works and factories are regulated in most developed countries, there has been great reluctance to extend the polluter pays principle over to nonpoint source emitters.

While most governments would not admit that nonpoint sources have the right to pollute waterways with their emissions, the reluctance to regulate or tax the management actions of these polluters has inadvertently provided the perception that these rights do exist. Changing these perceptions is not an easy task, and it is made even more difficult by the magnitude of the required change in management practices by nonpoint source polluters. By their nature, nonpoint source polluters are often relatively small enterprises unable to make all of the changes that may be required of them to reduce their emissions. Not surprisingly, calls to do so raise strong objections from farm interest groups. Where the decline in water quality is not sufficient to become a significant political issue in its own right, the temptation for government to bow to pressures from these politically powerful groups may be overwhelming.

The power of interest group politics is not just confined to influencing which pollution control measures are adopted. In those instances where economic instruments based on the polluter pays principle are implemented, interest groups are often able to bring considerable pressure to bear on the policy makers responsible for instrument design (Oates and Portney 2001). This has been clearly demonstrated in the design of emission charges for point source water pollution throughout Europe.[20] In France for instance, calls from environmental groups for

[20] The almost universal preference to 'grandfather' permits in tradable emission permit schemes is another example of how industry groups have influenced instrument design. This issue is discussed in Keohane et al (1997).

increases in emission charges during the 1980's were rejected by President Mitterand's Ministry of Finance on the grounds that industry should not be subjected to any further tax burden (Andersen 2001). Similarly, in other European nations, the implementation of emission charges (and environmental taxes more generally) has included a wide array of significant exemptions and tax relief for certain sectors – often to placate vocal industry lobby groups (Ekins and Speck 1999). At its worst this impact has been so great that the charge levied on high emission sectors (the focus of such schemes) has actually ended up lower than the charge on less significant small-scale emitters or households (Ekins 1999).[21]

The influence of interest groups in the policy process provides one piece of the puzzle as to why suasive measures and subsidies are often chosen by governments for the management of water pollution over charge based economic instruments. It also presents a clear challenge to economists to try and take instruments based on the polluter pays principle and package them in such a way as to make them politically acceptable while still providing for economic efficiency and effectiveness gains. At first blush, it may appear that there isn't a lot that could be done about this other than the approaches already adopted – lowering charges or providing concessions. There are two further practical options, however.

The first relates to the timing of the introduction of the instrument. It is no surprise that sudden calls for the imposition of charges and taxes on polluting industries or mandatory requirements for increased pollution abatement generate considerable opposition. Who would think otherwise? Alternatively, charges or management requirements could be phased in progressively over a number of years, with any future increases transparently presented in a clear schedule. This approach, while reducing the environmental effectiveness of the instrument (at least in the first few years) allows the polluting firms to plan their investments efficiently in relation to either an increasing charge or management requirement and their own potential abatement costs (Paras 1997).

The second option relates specifically to charge based economic instruments, whether they be based on emissions, inputs or management actions. Recycling revenue raised from the charge back to the affected parties can foster increased support for these instruments (Paras 1997). Revenue could flow back to affected industries in the form of a subsidy for pollution abatement [22] or to reward exceptional environmental performance. Moreover, it is technically possible, albeit extremely complex, for the emission charge to be revenue neutral if other Government taxes or charges are reduced by an amount equal to each affected parties total emissions charge (Ekins 1999). [23]

So, it appears that there is more that could be done with the design of economic instruments based on the polluter pays principle to support their increased acceptance and use. In particular, it appears that there are some relatively unexplored options available for reducing industry opposition with regards to the impact of the charge on both their tax burden and competitiveness. But the potential for these options to bring about widespread change should not be overstated. Such instruments (particularly taxes) are likely to involve the transparent imposition of a cost on

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[21] The example, which Ekins provides, is the Swedish CO_2 tax where industry and horticulture received reductions of up to 75 per cent from the base rate.

[22] This approach has been adopted in the Netherlands system of effluent charges. It is estimated that the revenue raised through Dutch effluent charge has been responsible for the majority of industrial investments in water pollution control in that country (Andersen 2001).

[23] This was successfully achieved with the Dutch small energy users tax (Ekins 1999).

potentially powerful members of society no matter how they are designed (putting the unlikely case of revenue neutral design aside).

Transaction costs

Transaction costs have also played a critical role in instrument choice and help to explain both the relatively low adoption of economic instruments in natural resource and environmental management and the deviation between their theoretical promise and practical performance.

Transaction costs include both policy transaction costs and market transaction costs (McCann and Easter 1998; Thompson 1996; Challen 2001; Falconer and Saunders 2002).[24] Policy transaction costs are predominantly borne by the policy maker (i.e. the government) and include the costs associated with: research, information gathering and analysis; enactment of enabling legislation (including lobbying and public participation costs); design and implementation of the policy (including costs of regulatory delay); support and administration of the on-going program; contracting costs; monitoring/detection costs (of both the efficiency of the outcome and the level of compliance); and prosecution/inducement costs (McCann and Easter (1998; 2002). [25] Market transaction costs include costs associated with search and information, bargaining and decision making and the enforcement of the negotiated trade (Stavins 1995). Market transaction costs are predominantly borne by the polluter, but as many economic instruments are essentially administrative tools they are also partially borne by the policy maker (Herath and Alfons 1999).

The textbook neoclassical explanation of the pollution problem assumes that the range of costs outlined above are equal to zero. Yet it is obvious that these costs not only exist in the real world but in some cases can be considerable (Stavins 1995; Saleth and Dinar 1999). It should come as no surprise then that economic instruments based on the neoclassical model fail to achieve their theoretical promise once these costs are considered. Indeed, high policy and transaction costs have played a major role in the failure of many water pollution trading schemes in the United States to deliver expected cost-savings (Hahn and Hester 1989; Environomics 1999).

High policy transaction costs have also shaped nonpoint source water pollution policy design. Suasive measures, including information and education programs are easier to design, face less stakeholder opposition and do not require new legislation. Moreover, by not mandating any pollution abatement they do not require potentially expensive monitoring, enforcement or prosecution. In short, these measures have relatively low transaction costs. Cost-share programs are likely to have higher policy transaction costs, requiring more considered design and on-going administration. However, they are unlikely to face significant negotiation and legislative costs. Furthermore, being based on the voluntary adoption of improved management practices they do not require an overly onerous monitoring regime. Harder-edged economic instruments, such as emission taxes and emissions trading would be expected to face far higher transaction costs. To

[24] The nomenclature of transaction costs has become complicated and confused since Ronald Coase first brought the concept to popular attention in his article ‘The Problem of Social Cost’ in 1960. [24]. In a narrow sense transaction costs are often viewed as being the actual costs of making a transaction in a market – for instance, the costs of bringing buyers and sellers together in a tradable emissions permits market. The broader definition of transaction costs, which includes policy transaction costs, has arisen out of the New Institutional Economics Paradigm, and through the work of Williamson (1985) and North (1990) in particular.

[25] McCann and Easter (1998) acknowledge that their framework builds upon the institutional cost framework developed by Thompson (1996).

operate successfully they demand far more certainty regarding the source and magnitude of emissions and their environmental impact. In turn, this requires greater research and information gathering. Enactment of legislation is likely to become more protracted due to the influence of farm lobby groups and underlying beliefs about property rights (particularly for instruments based on the polluter pays principle). [26] Contracting, monitoring, detection and administration costs are also likely to be high, particularly where there are a large number of nonpoint source polluters.

The significant differential in transaction costs between suasive measures and economic instruments has surely played some role in pushing the former to the forefront of nonpoint source policy. With limited funds and resources available, it is no surprise that governments may be unable (and sometimes unwilling) to bear the higher policy transaction costs associated with more sophisticated measures. This presents a considerable challenge to economists. As Stavins (1995) suggests:

> '..with transaction costs as with other departures from frictionless markets, greater attention should be paid to the details of design of specific systems, in order to lessen the risk of overselling these policy ideas and in order to create systems that stand a chance of being implemented successfully' p146.

So what can be done to reduce transaction costs? McCann and Easter (1998) suggest the policy measures aimed at addressing nonpoint source pollution are affected by a number of factors, including: the number and diversity of agents affected by the policy; resistance to the policy; the desired amount of pollution control; the time frame involved; whether the policy was voluntary or not; uncertainty; the available technology for best management practices; monitoring requirements; and the existing institutional arrangements. [27] Many of these factors relate to the way in which a policy is designed and implemented. This suggests that economists and policy makers can influence the level of transaction costs.

The first point where policy makers could encourage a reduction in transaction costs relates to the level of pollution control sought. The ideal of achieving zero or 'socially optimal' levels of nonpoint source pollution control is unrealistic in the current technological and political environment and small incremental gains or even maintenance of the status quo may have to be accepted as the policy objective.

Reducing uncertainty and resistance to the introduction of economic instruments also appears to be of great importance. Adopting policies which place little or no financial burden on current nonpoint source polluters (such as subsidies) is one way in which these costs could be reduced. Where taxes are preferred, a number of steps may be taken. As discussed above, regulatory requirements could be phased in rather than imposed in a single hit. Tax rates could be gradually increased to the desired level. Finally, revenue raised from the imposition of the tax could be recycled back to the affected industry. Transaction costs generated due to uncertainty could be reduced through a strong public education and information program prior to the

[26] Woerdman (2001) suggests that policy transaction costs are likely to increase the more that the distribution of property rights deviates from the status quo as rent seeking activities from lobby groups increased.

[27] Prior to McCann and Easter (1998) a number of analysts had outlined factors influencing transaction costs. These factors included: the degree of pollution control being sort by the policy maker (Stavins 1995); the type of policy being considered (Coase 1960); the number and diversity of agents (Williamson 1985, Oates 1986); available technology (North 1990); and the institutional environment (Coase 1960, North 1990).

implementation of the instrument [28] and the adoption of a transparent and unambiguous approach, backed by legislation where appropriate.

Market transaction costs may be reduced through the use of a brokerage service to supply information about potential buyers and sellers and help them identify one another (Stavins 1995). Governments can also reduce market transaction costs by reducing regulatory barriers, such as lengthy pre-approval for trades.

Making the required changes to reduce policy and market transaction costs will not always come easily. More often than not they involve making important trade-offs with the economic efficiency and environmental effectiveness of the economic instrument. Moreover, they may reduce one component of transaction costs while increasing another (e.g. phasing in projects reduces resistance but may increase administration costs). These trade-offs have to be carefully considered. Situations where transaction costs exceed the economic efficiency gains provided by economic instruments should be avoided. Similarly, the environmental performance of the instrument should not be sacrificed simply to generate greater reductions in transaction costs.

Conclusion

While the challenges for the successful design and implementation of economic instruments to address nonpoint source water pollution are considerable, the demand for economic solutions to this and other environmental problems appear to be increasing.[29] This growth is largely in response to two factors. First, alternative policy approaches have often failed to bring about the desired reductions in nonpoint source pollution. Concerns about the future environmental impacts of maintaining this path have grown. Second, the pressure on governments to deliver more cost-effective environmental management has increased as environment budgets have failed to keep pace with the required level of environmental management.

The ball is now firmly in the economists' and policy makers' court and the challenges are clearly presented. The question remains - can economic instruments be designed so that they are politically acceptable, reduce unnecessary transaction costs, provide for efficiency gains in the allocation of resources and provide incentives for improved environmental performance? These questions are considered in the following section in the context of a practical case study of the Swan-Canning River system in Perth, Western Australia.

5. Case study: Nonpoint water pollution in the Swan-Canning River system

The Swan-Canning river system consists of the rivers, watercourses, drains and tidally affected estuaries on the coastal plain around metropolitan Perth. It covers an area of more than 2000 square kilometers and is made up of thirty-one catchments spread over urban, industrial, semi-rural and rural land uses (SRT 1999).

The system is of great economic, recreational and environmental importance for the people of Western Australia, and has been since the establishment of the Swan River colony in 1829 (SRT 1999). Over the last one hundred years a combination of broad scale clearing for agriculture and rapid urbanisation and industrialisation has placed the system under significant environmental pressure. Public and government attention was drawn to this issue in 1994 as a

[28] The costs of the public education and information program would also have to be considered in ex-ante analysis of the instrument, however.

[29] This may be evidenced by the $10 million portion of the 2001 National Action Plan for Salinity and Water Quality earmarked for pilots of various economic instruments around Australia. Further evidence is provided by the growing numbers of economists in Environmental and Natural Resource Government Departments and organisations.

severe toxic blue-green algal bloom forced the closure of the river to recreational use for several months (SRT 1999). Since that event the incidence and severity of algal blooms in the Swan-Canning River system has increased dramatically (SRT 2002a).[30] In February 2000, a massive toxic blue-green algal bloom resulted in concentrations of algal cells in the system reaching as much as 6500 times the limit considered safe for recreational use as nitrogen and phosphorus levels in the Swan-Canning river system exceeded acceptable limits by 700 per cent and 300 per cent respectively (SRT 2000b).

In response to mounting community concern regarding the impacts of these algal blooms and fears that their incidence and severity would continue to increase in the future, the Western Australian Government launched the Swan-Canning Cleanup Program (SCCP). The initial stages of the SCCP focused on collecting scientific data on the condition of the river and determining the likely causes of the algal blooms. A number of key findings, with great importance for the management of the environmental health of the system, resulted. These findings are briefly outlined below.

First, the Swan-Canning River system is naturally predisposed to algal blooms. The open, sunny, slow-moving and shallow conditions of the Swan River combined with sandy soils with poor nutrient binding properties are ideal for algal growth.

Second, algal blooms are being fuelled by high concentrations of nitrogen and phosphorus entering the river system from the Swan-Canning catchment (SRT 1999). While nutrients from large point sources have been effectively managed, nutrients continue to enter the system from a large range of urban and rural nonpoint sources, small to medium size industrial and commercial properties and intensive agricultural enterprises such as piggeries, poultry farms and market gardens (SRT 1999). [31] Nonpoint sources are believed to provide the majority of nutrients delivered to the Swan-Canning river system (SRT 2000c).

Third, the level of nitrogen and phosphorus entering the system varies considerably from one catchment to the next, suggesting the need for a targeted management approach (see Table 3). Two catchments in particular, Ellen Brook and the Southern River, have a particularly strong influence on the estuarine ecology of the Swan-Canning River system. Located in the rural fringe around the Perth metropolitan area, the Ellen Brook and Southern River catchments supply the second and third largest volume of water to the Swan-Canning estuarine system, respectively. Both have moderate to very high nutrient concentrations.[32] Nonpoint agricultural sources are believed to account for the majority of nutrients entering the river system from both the Ellen Brook and Southern River catchments (SRT 2000c). A number of urban catchments, including those of the Mills Street Main Drain, Bannister Creek and Bayswater Main Drain are also responsible for lesser, but nonetheless significant export of nutrients into the Swan-Canning system. These nutrients come from a number of nonpoint sources, including urban parks and gardens, golf courses, unsewered residential areas as well as from small commercial and

[30] Between 1994 and 2002 twenty-nine separate blooms have been recorded in the Canning River system alone (SRT 2002a).

[31] Government policy from the 1930's onwards encouraged the relocation of heavy industry and sewage treatment plants and by 1980 large point sources of nutrients had been mostly eliminated (SRT 1999).

[32] The Ellen Brook catchment alone accounts for around one third of the total phosphorus load entering the Swan-Canning system (Agriculture Western Australia 2001).

Table 3 Nutrient emissions from catchments in the Swan-Canning River system

Catchment	Land Use	Nitrogen emissions (1997-99)	Phosphorus emissions (1997-99)	Average annual discharge
Ellen Brook	Rural – broad acre grazing, animal agistment, horticulture and viticulture Urban – townships are common	Moderate	Very High	High
Southern River	Semi rural – broad acre agriculture, horticulture and intensive livestock agistments Urban – rapidly developing residential areas	Moderate	Moderate	High
Mills Street Main Drain	Urban – medium density residential Light industry	High	High	Low
Bannister Creek	Urban – medium density residential Industry – light to heavy	Moderate	Moderate	Low
Bayswater Main Drain	Urban – high density residential Light to medium industry	Moderate	Low	Moderate
Bickley Brook	Urban – medium density residential Semi-rural – broad acre agriculture and horticulture Remnant vegetation – forested areas upstream	Moderate	Low	Low
Blackadder Creek	Urban – medium density residential Light industry	Moderate	Low	Low
Helena River	Urban – medium density residential Rural - broad acre grazing, horticulture and viticulture Light industry	Moderate	Low	Moderate
South Belmont Main Drain	Urban – high density residential Industry – light to medium	Low	Moderate	Low
Avon River	Rural - broad acre grazing, animal agistment, cereal crops Urban – townships are common	Low	Low	Very High
Bennett Brook	Urban – low density residential Rural – livestock agistment, viticulture and horticulture	Low	Low	Moderate
Jane Brook	Remnant vegetation – large tracts of native forest Semi-rural – broad acre grazing, viticulture and horticulture Urban – some pockets of low density residential	Low	Low	Low
Susannah Brook	Rural – broad acre agriculture, viticulture and grazing Remnant Vegetation – forested areas upstream Urban – some pockets of low density residential	Low	Low	Low
Canning River	Urban – medium density residential Remnant Vegetation – forested areas upstream Semi-rural – animal agistment, horticulture, turf farms	Low	Low	Moderate
Yule Brook	Rural – horticulture, intensive animal agistment Urban – rapidly developing residential areas Industry – areas of light to medium industry	Low	Low	Moderate

Source: SRT (2000c); SRT (2002).

industrial point sources that continue to discharge their wastewater directly into the drainage system (SRT 2000c).[33]

Fourth, because of lags in transport time through the catchment, linking nutrient concentrations to a change in catchment condition is difficult and uncertain, particularly for nonpoint sources of nutrients (SRT 2000d). Trends in nutrient concentrations from nonpoint sources of nutrients in catchments are likely to be characterised by a steady slow change in nutrient levels over long periods (SRT 2000d).

Fifth, a large amount of phosphorus enters the Swan-River system as surface runoff during storm events (Jakowyna 2002). The export of nutrients in this fashion has been exacerbated by the degradation of riparian zones and extensive land clearing (SRT 2000c).[34] Around one sixth of nutrients are also estimated to enter the system via movement from the shallow groundwater table that lies beneath Perth (SRT 1999).

Finally, nonpoint source nitrogen and phosphorus emissions from Perth's rapidly increasing urban residential development [35] pose a significant long term threat to the Swan-Canning river system. New urban development would be expected to increase the level of nitrogen and phosphorus emissions into the Swan-Canning river system as: nutrient runoff from urban areas increases in proportion to the area of impervious surfaces such as roads and roofs (SRT 1999); the clearing of remnant vegetation for urban development makes land more susceptible to erosion and provides direct paths for nutrients into waterways [36]; fertiliser application rates increase as housing density increases (SRT 1999); and as the number of vehicles and domestic animals in a given area increase (SRT1999). In the case of Perth, the environmental impact of urban development is further exacerbated as it is expected to be forced into areas which are both difficult to drain and likely to have the greatest potential impact on water quality in the Swan-Canning river system (Robinson 2001).

In response to their increased knowledge regarding current and potential sources of nitrogen and phosphorus into the Swan-Canning river system, the Western Australian Government released the Swan-Canning Cleanup Program Action Plan in 1999 (the Action Plan). The 10 recommendations made in the Action Plan may be summarised into four key Actions: 1) support Integrated Catchment Management to reduce nutrient inputs; 2) improve planning and land use management to reduce nutrient inputs; 3) modify river conditions to reduce algal blooms; and 4) monitor river health, fill critical gaps in knowledge and report progress to the community.

Of greatest interest for the purposes of this paper is the second of the key actions. More specifically Action 2 outlines the need to modify land-use practices and prevent or relocate polluting activities, develop and adopt best management practices to reduce nutrient inputs in current land management practices and in all future developments, and use economic and

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[33] These include former landfills, industrial sites and stockyards.

[34] Satellite imagery indicates that in some portions of the Swan-Canning river system, such as the western margin of the Darling Plateau in the Ellen Brook catchment, nearly all remnant vegetation has been cleared (Agriculture Western Australia 2001).

[35] Perth is one of the fastest growing cities in Australia. Its population is expected to increase from its current level of around 1.4 million to between 1.8 and 2.3 million people by 2031 (Western Australian Planning Commission 2000).

[36] It has been estimated that around 6000 hectares of remnant vegetation was cleared in the Perth metropolitan area over a two year period in the mid 1990's, with a high proportion of this being for approved urban development (SRT 1999).

regulatory mechanisms to encourage catchment, wetland and river foreshore management (SRT 1999).

To date the major actions taken to address water quality in the Swan-Canning river system as part of the Action Plan have centred on a combination of on-ground works by community and catchment groups [37], public education, ongoing research and monitoring and a number of scientific and engineering projects [38] (2002b). These actions reflect the way in which the Western Australian Government treats responsibility for addressing nonpoint source water pollution. The Action Plan states that it is 'impractical and unreasonable' (SRT 1999 p89) for costs of environmental improvements to be borne by nonpoint sources that contribute to nutrients in the Swan-Canning river system. Rather, the Action Plan states that the costs of implementing actions to reduce nutrient emissions should '…. be considered a general community cost and be funded, principally, from State Government revenue and revenue of relevant local governments on a basis to be determined by consultation between the parties' (SRT 1999 p89).

The prevailing attitude to who should pay for the costs of reducing emissions may partly explain why very little attention has yet to be paid to the potential role that economic instruments could play alongside the current suite of activities adopted through the Action Plan, particularly for addressing nonpoint source pollution. Moreover, while the strong political will for improved water quality in the Swan-Canning river system may be evinced by the substantial effort currently being made through the various activities of the Swan Canning Cleanup Program, the focus of the program remains centred around measures based on the beneficiary (i.e. the community) pays principle. At no point in the Swan Action Plan is there any indication that nonpoint sources are likely to be regulated or licensed, with most effort being aimed at education and information provision and cost-sharing assistance for voluntary on-ground works (SRT 1999; SRT 2000a; SRT 2001; SRT 2002b). Furthermore, there is no evidence that these views will change in the near future and economic instruments must be designed with this consideration firmly in mind.

6. Potential economic instruments for nonpoint water pollution control in the Swan-Canning river system

After consideration of the political, regulatory, institutional, environmental and economic landscape of the Swan-Canning river system it appears that there may be a significant role for economic instruments, alongside regulatory and suasive measures, as part of the ongoing management of water quality. An initial evaluation of the merits of four such instruments is provided in this section.

The first two of these instruments are designed to provide greater incentives for the adoption of best management practices (BMP) by agricultural nonpoint source polluters. The first, a system of auctioned grants for the voluntary improvement of management practices is based on the current underlying beliefs regarding the property rights for nonpoint source emissions. The second, a system of BMP incentive charges, is based on the principle that the nonpoint source polluters should pay for their emissions. While it is clear that such an approach sits uneasily in the current political environment, the design of this instrument is aimed at mimimising this

[37] In 2001-02, $415 000 was provided to eight catchment groups, with the Ellen Brook catchment group being the largest recipient (2002b)

[38] These include the building of an artificial wetland, drain retrofitting, oxygenation of the river and the Phoslock modified clay treatment trial.

opposition from nonpoint source polluters and their lobby groups through a system of gradual phasing in of BMP requirements and charges for non-compliance as well as a system of revenue recycling to reward positive environmental action.

The third economic instrument to be examined is a system of emissions offsets targeted at managing future increases in urban residential nonpoint emissions. In recognition of the potential for high transaction costs to erode the benefits from such a scheme, a regionally managed offset bank system is considered.

A fourth instrument, similar to that introduced in the San Joaquin Valley in California (as outlined above) is also examined. While not being suitable as a short term solution (due to the current regulatory and institutional arrangements) this instrument is considered due to its ability to address both urban and agricultural nonpoint source pollution at some point in the future.

As will become apparent, the relative strengths and weaknesses of each instrument vary considerably. Furthermore, it is important to recognise that instrument design and the choice of where the instrument is applied are critical determinants of the likely economic and environmental performance of each of these instruments. This point should not be understated – economic instruments have performed well in those instances where concerns regarding administrative complexities, distributive impacts and stakeholder acceptability have been explicitly addressed upfront. In cases where factors such as monitoring and enforcement costs, stakeholder opposition and inappropriate economic conditions have been ignored, economic instruments for pollution control have generally performed very poorly. With these considerations in mind each of these instruments is discussed below.

Auctioned BMP payments

The development and implementation of BMPs is regarded as one of the key ways in which nutrient emissions from nonpoint sources can be reduced. BMPs complement industry standards and objectives, taking account of the state of technology, local physical and environmental conditions and the financial implications for the party undertaking management change (SRT 1999).

The process of developing BMPs is a statutory requirement in the Swan-Canning Environmental Protection Policy. As part of this process, management guidelines for a number of industries responsible for nonpoint source pollution have been developed. For instance, a range of broad recommended management practices for the grazing industry in the Ellen Brook are provided in the Agriculture Western Australia publication *Sustainable Land Management in the Ellen Brook Catchment* (Agriculture Western Australia 2001). As the Ellen Brook is a major contributor of both phosphorus and nitrogen into the Swan-Canning River system this example is used to explain how a system of BMP payments might operate.

The grazing of beef and horses on legume based pastures are the most common land uses in the Ellen Brook catchment and are a significant source of phosphorus emissions into the Swan-Canning River system (Agriculture Western Australia 2001). Emissions of nitrogen and phosphorus from the grazing industry are derived from commercial fertilisers applied to pastures in rates that exceed plant requirements and effluent from animal faeces in grazing areas (Agriculture Western Australia 2001). Subsequently, the control of water erosion and nutrient and fertiliser application are seen as the two key elements of managing the export of nutrients in this catchment. Within these two elements there are a complex range of specific management actions that could be considered as BMPs.

Management measures to reduce the risk of erosion by water include: construction of grade banks; fencing off creeks and waterways; re-establishing vegetation (such as sedge and rush beds) along the edges of creeks and waterways; maintaining grassed waterways in paddocks; cultivating on the contour of the land; and increasing the infiltration of water by maintaining healthy and well structured soils (Agriculture Western Australia 2001).

Nutrient and fertiliser management measures include: the use of soil testing to determine the nutrient requirements of the pasture; [39] carefully timed fertiliser application to enhance plant uptake and avoid leaching of nutrients; [40] the use of slow release fertilisers (where possible); the establishment of setbacks beyond or within which no fertiliser is applied; vegetated buffer areas which break up and reduce the severity of runoff sheet flows; and the fencing of creeks and waterways to reduce the level of stock faeces directly entering the waterway (SRT 1999; Agriculture Western Australia 2001).

Strategic replanting along fence lines and streamlines, windbreaks and revegetated foreshores and floodplains are also considered to be important for reducing the level of nutrients entering waterways and improving water quality more generally (SRT 1999). Re-vegetation of the riparian strip provides a number of benefits including erosion control, filtration of nutrients, habitat for wildlife as well as aesthetic and recreational benefits for humans.

Together, these actions represent a ‘toolbox’ of BMPs that may be employed to reduce emissions. Which particular practice is most effective will vary considerably from one site to the next (SRT 1999). That is, there is no single BMP or group of BMPs that may be considered as the most suitable management alternative for all farms in the Ellen Brook catchment. BMP on one farm may involve soil testing and the construction of grade banks while for others the most important management action may be the fencing off of waterways. In the Ellen Brook catchment, a wide variety of landforms, soil types and microclimates require different land management styles and practices to manage the impact of the agricultural activity on water quality (Agriculture Western Australia 2001). The heterogeneous nature of nonpoint source polluters and the impacts that various land management activities have on the level of emissions from one site to the next are important considerations in choosing and designing the appropriate policy response.

The current approach for encouraging the adoption of these management practices in Ellen Brook is based on a public information campaign (including farm planning days). Funding for undertaking some of the activities suggested above is available through Commonwealth programs such as Landcare. A dedicated program aimed at providing grants to encourage the uptake of BMPs in the Ellen Brook catchment (and in the Swan-Canning River system more generally) does not currently exist. The possible design of such a scheme is discussed below.

In their most simple form BMP payments are no more than a fixed-rate payment from the government to a landholder for the landholder to undertake an agreed management action. For instance, a farmer may receive $x for each metre of fencing they place around a waterway. Adopting this form of grant delivery is most efficient where the potential improvement in water

[39] Soil tests in the Ellen Brook area have suggested that too much phosphorus is being applied to pastures (Agriculture Western Australia 2001).

[40] Agriculture Western Australia suggest the best approach to involve a split application of phosphorus, potassium and sulphur being top dressed three to four weeks after germination, followed by a later application of mainly potassium and sulphur in August. Decisions on the appropriate timing and amounts will depend on soil types and soil testing should be carried out prior to application (Agriculture Western Australia 2001).

quality and the opportunity cost of delivering these improvements are homogenous (Falconer and Saunders 2002). These conditions rarely apply in practice, however, and as discussed above Ellen Brook is no exception.

Where sites are heterogeneous, targeted grant delivery is likely to deliver a more cost-effective outcome (Falconer and Saunders 2002). Such a system takes into account the differences between farm types and differentiates payments according to the opportunity cost of undertaking a management action and the expected improvements in water quality that it will provide. Targeting grant delivery comes at a cost, however. This occurs due to the clear presence of information asymmetry (Latacz-Lohmann and Van der Hamsvoot 1997). Nonpoint source polluters (in this case farmers in the Ellen Brook catchment) have a far better idea than the government as to how various management changes will affect their production plans and profits. Under these conditions the government needs to expend resources to have landholders reveal information about these impacts.

Auctions (or competitive tendering) are one mechanism that can elicit the required information from farmers. In addition, auctions may be designed to accommodate variability in water quality improvements from one site to the next (Latacz-Lohmann and Van der Hamsvoot (1997). Competition between bidders for BMP payments and the ability of the government to compare the environmental and cost-effectiveness of each bid also provides for a more cost-effective solution.

For these reasons, competitive tendering has been used in a number of environmental programs around the world. The Conservation Reserve Program (CRP) in the United States is the largest and best known of these. [41] More recently, this approach has been used as part of the BushTender trial in Victoria to deliver payments to farmers for management actions that increase the biodiversity value of their land. Results from early trials of BushTender indicated that cost savings of around $2.3 million (or eighty-five per cent) were achieved by using a competitive tendering system rather than a fixed-rate approach (Stoneham et al 2002).

Despite the obvious advantages of using such an approach, the use of competitive tendering in addressing nonpoint source water pollution has been limited. Many of the issues faced in the provision of BMP payments to address nonpoint source water pollution mirror those faced in biodiversity conservation, however, and it appears that a similar scheme to that adopted in the BushTender trial could be used to improve water quality in the Ellen Brook catchment.

The first step in developing such a scheme involves linking the impact that various management practices would be expected to have on emissions of nitrogen and phosphorus. This is necessary so that the relative improvements in water quality from actions undertaken from one site to the next can be compared. In the case of Ellen Brook it appears that these links are uncertain and cannot be made with a great degree of confidence. This uncertainty is exemplified by the findings of a detailed study undertaken in Ellen Brook to examine the role of constructed wetlands in reducing nutrient emissions from farms. Despite intensive research this study was unable to provide any quantitative guidance on what emissions benefits would result (Agriculture Western Australia 2000). While the difficulties in establishing the links between management actions and water quality should not be understated, for the purposes of examining how a system of auctioned BMP payments could operate it is assumed that these links are known.

[41] Land retirement contracts under the CRP have been delivered through a competitive bidding mechanism since 1986.

The next stage of a system of auctioned BMP payments would involve an assessment of individual sites within the Ellen Brook catchment to determine which management actions would be most suited to each site and what the likely impact on water quality of undertaking these actions would be. Based on the model adopted for BushTender, landholders would then identify the actions they would be prepared to undertake and prepare an agreed management plan as the basis of their bid (Stoneham et al 2002).

Following the preparation of the management plan, landholders would be asked to submit a sealed tender for the level of payment they require from the government if they are to undertake the proposed management action. It is envisaged that cost-sharing provisions could also be included in the scheme whereby individuals may propose to undertake some of the costs of changing their management practices. This would not only alleviate the public cost of undertaking the management change but would also improve the competitiveness of their bid. Furthermore, tenders could also be made collaboratively with neighboring properties for actions that may cross farm boundaries. For instance it may make sense for a contour bank to run into a neighbor's dam, while stabilisation of creek lines is easier and more cost-effective when large lengths are done at the same time (Agriculture Western Australia 2001).

Once all tenders have been received, the government would then access each tender in terms of its cost and the anticipated impact on pollution levels. Funds would be provided for those activities that are considered to provide the greatest per dollar improvement in water quality. Successful landholders would then be notified and required to sign legally enforceable management agreements outlining what is expected from the landholder and how much they will receive if they undertake these actions. The BMP payments could be either provided upfront (placing the risk of non-compliance with the government) or paid once the management action has been completed (placing the risk of non-payment with the landholder). To share the risks the payments could be spread over the life of the project.

The benefits of auctioned BMP payment over fixed-rate payments are clear. As evidenced by the results of the BushTender trial in Victoria significant cost-savings and economic efficiency gains may be made by opening up competition between farmers for payments and incorporating cost-minimisation as the basis for ranking tenders, (Stoneham et al 2002). These gains enable either more to be achieved with a limited budget or the freeing up of government resources for other programs.

It is important to recognise that these cost-savings do not take into account the total transaction costs associated with the design, delivery and upkeep of the approach, however. The full cost of a system of BMP payments includes not only the costs of the payments to landholders but also the administrative and organisational costs incurred by both parties. These would include the cost of acquiring the necessary level of information to rank tenders, negotiating contracts with landholders and monitoring and enforcing management agreements. Whether or not a system of BMP payments would be more cost-effective than fixed-rate payments once transaction costs are taken into account is uncertain. Moreover, the reduction in emissions of phosphorus and nitrogen from properties in the Ellen Brook catchment generated by a system of auctioned BMP payments is also unclear due to the voluntary nature of the program. Both of these issues demand further and careful consideration.

BMP incentive charges

While auctioned BMP payments are based on the current underlying beliefs regarding the property rights for nonpoint source emissions, a system of BMP incentive charges is based on the polluter pays principle. In this sense, auctioned BMP payments may be viewed as a form of subsidy and BMP incentive charges as a tax.

As discussed earlier, although it is not currently possible to use a system of emission charges for nonpoint source polluters, taxes may be based on observable input use and management actions. Where one input is the major contributor to the pollution problem a single input tax may be considered.[42] However, in the case of Ellen Brook it is clear that there are a combination of inputs and management actions that are important determinants of the level of emissions from farms in the Ellen Brook catchment. This suggests that a more integrated approach, that takes account all of these actions is required. BMP incentive charges are one such approach.

Under a system of BMP incentive charges, nonpoint source polluters face variable charges based on their management practices. Actions that result in high levels of emissions face the highest charges while farms demonstrating BMPs would be exempt. Assuming that incremental improvements in management practices may be made, a series of charge levels could be placed between these two extremes. Ideally the level of charges would be closely related to the level of environmental damage caused by various management actions. This would improve the economic efficiency of the instrument and enable the internalisation of externalities associated with excessive emissions. If the level of environmental damage caused by emissions, and hence the charge faced by the nonpoint source polluter, was high enough, an incentive to implement BMPs to avoid paying the charge may exist. In this way the system of BMP incentive charges would encourage the uptake of BMPs, reduce nitrogen and phosphorus emissions and improve water quality.

Unfortunately, the level of science needed to link potentially costly management actions with emissions and environmental damage is not currently available. As discussed earlier, any attempt to base charges on environmental damage under conditions of scientific uncertainty is likely to generate significant opposition from affected stakeholders. Moreover, the legal validity of adopting such an approach is questionable.

Whilst significantly reducing the appeal of BMP incentive charges, this does not necessarily spell their demise. Indeed, many practical applications of emission charges for point source pollution (particularly in Europe) have faced similar problems, with charges not being linked to environmental damage and the level of incentive for pollution abatement remaining too low to facilitate pollution abatement (OECD 1997; Andersen 2001; Stavins 2001). Despite this, some of these schemes are still considered to have been relatively successful and have resulted in improved water quality (Andersen 2001). These improvements have arisen as the revenue raised from emission charges has either been recycled back to polluters so that they undertake pollution abatement or has been used for public environmental programs.

The European experience with point source emission charges suggests that even without the required scientific and economic information a system of BMP incentive charges could reduce emissions from nonpoint source polluters in catchments such as Ellen Brook. The first stage in establishing these charges would be to determine what management actions are going to be required under the system. Using BMPs as an end point, a schedule of what may constitute a

[42] A fertiliser tax may warrant further consideration for catchments such as Ellen Brook. Two factors work against this instrument, however. First, fertiliser tends to be highly inelastic meaning that a large tax would be needed to induce management changes and reduce emissions (Ecotec 2001). Second, introducing a large tax on fertiliser is generally politically unpalatable and is unlikely to be seriously considered (Vatn et al 2002).

reasonable time-path from current to best management practices could be prepared. This could be done by a regional or catchment authority or by the State Government in consultation with farmer and environmental groups.

The chosen path from current to best management practices may end up involving several broad progressive management changes.[43] For instance, the first step may be that all farms undertake soil testing. An interim step may involve the provision of fertiliser setback areas, while a final step could entail the revegetation and fencing of all waterways.

Once these steps have been agreed to a series of charges may be attached. For instance, the charge for farms who have undertaken none of these steps could be $x/year. Farms who have undertaken step one may pay a lower charge equal to $y/year. Charges would get progressively lower as more of the BMP steps were completed until farms that are deemed to be undertaking BMPs would pay no charge at all.[44]

The imposition of these charges would be expected to face significant opposition from farms and farm groups within Ellen Brook. Lengthy negotiations between these groups and the government have the potential to rapidly increase policy transaction costs. As alluded to earlier, there are a number of options available to reduce both the opposition from farm groups and the resultant transaction costs.

Firstly, the introduction of charges can be phased in gradually in recognition of the potential magnitude of management changes that may be required. It is clear that some of the recommended management changes will be relatively costly and require significant resources from the landholder. It would be clearly unreasonable for the government to expect farms to undertake these actions at short notice. Once the schedule of required management actions and charges was determined farmers may be given a period to adjust to the new system. For instance, it may be decided that farms have five years to adjust before the charges were to be introduced. In this period they can determine the impact that the charges will have on their operation and make management changes so as to reduce the charge they will receive once the program comes into operation.

Opposition could also be reduced by gradually increasing charges over a period of time. For instance, a charge schedule for each of the BMP steps over a period of ten years from the commencement of the program could be presented to farmers. Charges in year one of the program (five years after its initial announcement) could start low and gradually increase to the desired level by year ten (fifteen years after the commencement of the program). Fees could also be capped at a maximum amount so as not to endanger financial viability in the short term. While these may appear to be extreme concessions, they may be required to engender the necessary stakeholder support for the introduction of the charge system. Where there is the serious overhanging threat of direct regulation (i.e. a mandatory requirement to adopt BMPs) concessions may be watered down, as stakeholder support for the charge system would be more forthcoming.

Finally, once administrative costs have been covered, revenue raised through the collection of charges could be recycled back to affected industries to reward those farms that have made the

[43] Broad management changes that would be likely to reduce emissions from all farms should be chosen. Whether or not this is possible when farms are heterogeneous is uncertain.

[44] Any BMPs, which were found to be not as effective as initially intended, could be removed from the schedule. It is important that farmers who have undertaken these actions in the current round of BMP incentive charges are not penalised.

greatest management changes. These farms could have their charges returned and also receive additional payments to offset the costs of the actions they have undertaken. Revenue could also be used to provide rebates to farms that can demonstrate exceptional financial circumstances.

Alternatively, part of the revenue could be recycled back to the affected industry (to reduce opposition) and part could be used to fund public environmental works in the Ellen Brook catchment. Public works in the Ellen Brook could include large engineering works, such as the construction of wetlands, as well as the foreshore management of public waterways (SRT 1999).

Although the actual environmental effectiveness of a system of BMP incentive charges would depend heavily on the way it was designed, such an approach holds considerable appeal. It provides greater flexibility to farmers than a regulatory requirement to implement BMPs while providing for greater incentives to do so than a voluntary requirement. Moreover, the funds raised through the charge can be used to fund greater improvements in water quality. Furthermore, the measurement and compliance costs associated with this system are likely to be significantly lower than more complex instruments (especially market based instruments).

Such a scheme cannot guarantee significant environmental improvements, however. It is clear from the discussion above that the level of economic and environmental efficiency offered by BMP incentive charges is dependant on the level of incentive charges and the way in which revenue is recycled. Setting charges so that they are both politically acceptable (in terms of current beliefs about property rights) and high enough to provide an incentive for management change is difficult. Allowing concessions, such as lower charges and phase-in provisions, while reducing opposition to the imposition of a system of BMP incentive charges, may also reduce the environmental effectiveness of the program. If charges are set too low, no incentive for improved environmental performance by farmers may result. In this case, the only improvements in water quality will be generated through the recycling of revenue for pollution abatement or public works. Alternatively, too higher charges or too fewer concessions may mean that the program generates such opposition that it never sees the light of day. These issues highlight the difficult trade-offs that are faced by the policy maker in designing such a program and once again demand careful consideration prior to implementation.

Emission offset bank

Under a system of emission offsets any emissions increase occurring as a result of new development or changed management practices must be offset by an equivalent reduction in emissions elsewhere in the same river system. Emission offset schemes are most commonly designed to maintain the total level of emissions in the river system at a constant level (otherwise known as no net increase) and are typically targeted at large point sources of emissions.

As discussed earlier in this section, Perth has already significantly reduced nitrogen and phosphorus emissions from large point sources and the problem of water quality in the Swan-Canning river system is mainly due to nonpoint source emissions. This may suggest that offset schemes are of little practical interest. This is not necessarily true. Offset schemes could also be designed so that increases in one set of nonpoint source emissions are offset by reductions in another set of nonpoint source emissions.[45] For instance, in Perth an offset scheme could be designed so that increases in nonpoint source emissions from urban residential development were

[45] An emissions offset scheme targeted at small point sources in Perth also warrants further consideration. Small point sources include small to medium size industrial and commercial premises and intensive agricultural activities such as piggeries, poultry farms and market gardens (SRT 1999).

offset by reductions in agricultural nonpoint source emissions. The design of such a scheme, including the use of an offset bank, is discussed below.

Urban Nonpoint Source Emissions Offset bank

A number of management options are available to address the potential impact of increasing nonpoint source emissions from urban residential development in Perth. One such option is to prohibit any new developments that contribute increased emissions into the Swan-Canning river system. This appears to be the option favoured in the Swan-Canning Cleanup Program Action Plan (1999), which states that local government town planning schemes should 'specifically eliminate uses that can contribute nutrients and sediments to watercourses and wetlands' (SRT 1999 p73).

Eliminating emissions from urban residential development is not an easy task however. Indeed, even following a BMP approach that includes the incorporation of water-sensitive design principles in the design of stormwater drainage systems, the construction of detention basins, vegetated swales and artificial wetlands for nutrient stripping and programs to encourage more efficient fertiliser and water use, may not eliminate all nitrogen and phosphorus emissions (SRT 1999). Moreover, undertaking all of these actions would be costly, particularly for developments in areas with poor drainage, steep slopes or less permeable soils (SRT 1999).

Achieving similar reductions in nitrogen and phosphorus emissions elsewhere in the Swan-Canning river system could potentially be achieved at a lower cost. In particular, simple management changes on farms in catchments such as Ellen Brook and the Southern River could potentially result in equivalent emissions reductions at a much lower cost. It is this cost differential which provides the platform for a nonpoint source emissions offset scheme in the Swan-Canning river system.

The first stage in establishing an offset scheme is the development of legislation that would prohibit new urban residential development unless either: a) all nonpoint source emissions are eliminated on site; or b) any increase in emissions from the development is offset through equivalent emissions reductions elsewhere in the Swan-Canning river system. The decision on how to reduce emissions would be made by the developer based on comparing the costs of doing so on site with the costs of arranging for other emitters in the system, such as agricultural nonpoint sources, to undertake equivalent reductions. The legislation could also establish the rules of the offset program, including the types of offset actions that can be used, criteria for determining whether these actions have been successful, the areas where offset actions can be undertaken and the trading ratios that may apply (Morrison 2002).

Offsets are typically undertaken on a bilateral basis, whereby the developer conducts their own search for the appropriate off site action and negotiates the terms of the offset contract with the other party. The costs associated with making bilateral offset transactions can be high, however, and threaten to erode much of the potential cost-savings associated with the offset. In extreme cases, the transaction costs may even outweigh these cost-savings.

A system of offset banking, based on the wetland mitigation banking [46] model used in the United States, offers a pragmatic solution. [47] With an offset banking system, government or

[46] In 2001, there were estimated to be over 200 operational wetland mitigation banks in the United States (Morrison 2002).

approved private organisations undertake actions to reduce emissions. Based on the outcomes of these actions, a 'bank' of emission reduction credits is built up. Similar to the operation of the bilateral offset scheme the developer is presented with two options if they are to comply with the offset legislation. First, they can eliminate the emissions from the development by implementing on site management actions. Alternatively, they can allow the emissions from their development to increase unabated and purchase an equivalent amount of emission reduction credits from the offset bank.

Offset banks offer a number of advantages over bilateral offset schemes. First, they significantly reduce transaction costs per unit of emissions reduction. This is achieved through the increased efficiency with which large offset organisations, whether public or private, are able to organise offset actions, compared to individual developers. Large organisations can bring together financial, planning and scientific expertise that is simply not available to developers (US Corps of Engineers et al 1995). Second, they may achieve reductions in emissions more effectively. Large projects, which may offer greater per dollar reductions in emissions than a number of small projects, may be undertaken by the owner of the offset bank to generate emission reduction credits. Third, they provide for greater environmental certainty without slowing down applications for urban residential development (US Corps of Engineers et al 1995). Credits generated in offset banks are based on the outcomes of activities that have already been completed meaning that development can proceed as soon as credits are purchased by the developer.

The introduction of an urban residential development offset bank is not without its problems, however. One such problem relates to the way in which an offset scheme can contribute to localised impacts on water quality (otherwise known as 'hotspots'). Hotspots would occur if nitrogen and phosphorus emissions increased beyond acceptable limits in certain catchments despite the total level of emissions in the system remaining unchanged. This would be a real concern in the Swan-Canning if the majority of emissions reductions were to be undertaken in semi-rural and rural catchments such as the Ellen Brook and Southern River while urban residential development was focused in other catchments. The risks of hotspots can be reduced through introducing distance ratios in to the offset scheme or through government review of each offset trade (Morrison 2002)[48]. Distance ratios require greater reductions in emissions the further the offset project is away from the urban residential development. By doing so they encourage offset projects to be developed in closer proximity to the urban development.[49]

A second and far more important problem facing the successful implementation of an offset bank relates to how emissions from nonpoint sources can be reliably measured. The efficient operation of the offset bank is predicated on the ability of the government (or an approved private business) to estimate both the emissions loadings from new urban residential developments (the 'debits') and the 'credits' generated by emissions reductions from projects conducted by the owner of the offset bank (Morrison 2002). While there are no easy answers

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[47] The use of a brokerage service, provided by either the Government or private sector, is another option that could be introduced to reduce the costs associated with searching for an appropriate offset.

[48] The introduction of distance ratios would be expected to increase the market transaction costs. These increased costs would need to be traded-off against the reduced environmental risk of allowing free trade.

[49] While, reducing the risk of hotspots this would also limit the cost-savings generated by the offset bank. The impact of the distance ratio would need to be weighed up against the risk of 'hotspots' in the design of the offset bank.

here, it appears the best way forward involves the use of emission proxies. As discussed earlier, such proxies are currently not available in the Swan-Canning river system. Progress elsewhere, suggests that developing emission proxies for nonpoint source emissions is possible, however.[50] Once these estimation tools have been refined and adapted to the conditions of the Swan-Canning river system the practical application of a nonpoint source emissions offset bank would appear to warrant serious consideration.

Conversion of nonpoint sources into point sources

The economic instruments discussed in this section have all followed the conventional approach to addressing nonpoint source water pollution. That is, they have been based on either an emission proxy or on inputs and management actions that are correlated with the level of nonpoint source emissions. The success of the Grassland Area Farmers Tradable Loads Program (GAFTLP) in the San Joaquin Valley of California indicates that there may be an alternative management solution. The possibility of implementing a similar program in the Swan-Canning river system is discussed below.

The consideration of a program similar to the GAFTLP is predicated on the physical ability to convert groups of nonpoint sources into point sources. For this to be achieved three key conditions must be met. First, that there is a system of main waterways that may be treated as point sources. Second, that all (or at least a very high proportion) of emissions in the main waterways come from nonpoint sources. Third, that these nonpoint sources can be identified.

The system of waterways in the Perth metropolitan area appears to satisfy the first of these conditions. A network of small local drains in urban catchments flow into a system of main drains.[51] These main drains 'act as conduits which act to concentrate the nutrients from the drainage catchment into a single point source' (Department of Environmental Protection 2001 p26). In semi-rural and rural catchments a system of man-made and natural waterways flow into large brooks or rivers. These natural waterways act like the main drains in collecting the nutrients from the drainage catchment and effectively convert emissions from a large group of individual sources into a single point source. Based on the ability to measure catchment emissions in main drains and waterways, phosphorus and nitrogen load targets have been set for each of the thirty-one catchments within the Swan-Canning River system (Robinson 2002). At this point in time, the nutrient concentrations in fifteen of these catchments (see Table 3 for full listing) are being monitored on a regular basis (SRT 1999).

A recent study into the nature of nutrients in the fifteen monitored catchments indicates that the main source of nutrients varies considerably from one catchment to the next and that nutrients in just six of the catchments were found to come predominantly from nonpoint sources (Jakowyna 2002). [52] Of these six catchments, the Ellen Brook and Southern River are major contributors of emissions to the Swan-Canning river system, while the Avon River, Jane Brook, Upper Canning River and Yule Brook are all minor contributors (see Table 3). The study also indicated that matching emissions from individual sources (both point and nonpoint) with nutrient concentrations in distinct waterways and main drains is still an uncertain exercise.

[50] Spreadsheet based tools have been used in Georgia in the United States to estimate emissions from new urban developments (Morrison 2002).

[51] Over 2500 kilometres of small surface and sub-surface drains are connected to a 300 kilometres long main drain system (SRT 1999).

[52] Nutrients in other catchments were found to come predominantly from a single point source, a mix of sources or from sub-surface and groundwater flows (Jakowyna 2002).

Indeed, ‘with the exception of Ellen Brook, our understanding about specific nutrient transport processes in catchments is limited’ (Jakowyna 2002 p 72).

So it appears that although groups of individual nutrient sources in the Swan-Canning river system can be converted into ‘point sources’, in few instances are the sources predominantly nonpoint and, perhaps more importantly, the links between individual sources and measurable nutrient concentrations in main drains and waterways remains uncertain (Ellen Brook excepted). This indicates that a system based on the GAFTLP is not currently suitable in the Swan-Canning River system.

Research into the movement of emissions from individual sources into main drains and waterways is constantly developing, however, and it is feasible that in a number of years these links will be much clearer. In preparation for that day, a number of authorities including the Department of Environmental Protection have already began investigating the potential for licensing nutrient loads from main drains in the Swan-Canning river system. Licensing could also potentially be extended to natural waterways, such as Ellen Brook and the Southern River.

By introducing licensed emissions from main drains and waterways in the Swan-Canning river system each catchment is essentially treated as a single point source. By doing so, economic instruments typically used to manage point source water emissions may be considered. Emission charges could be applied to the authority responsible for water quality in each catchment based on the nutrient loads in main drains and waterways.[53] Alternatively, a system of tradable emission permits could be introduced. While it is emissions from each of the fifteen catchments being capped, rather than individual point sources, the operation of the instrument would remain much the same. Whether there would be sufficient depth in the market [54] or the required differential in marginal abatement costs between catchments for the instrument to deliver efficiency gains is uncertain. These economic considerations, whilst being important, are not the limiting factor in the applicability of this approach to nonpoint source emissions in the Swan-Canning river system, however. Indeed, they are dwarfed by a number of institutional challenges.

The greatest barriers to the introduction of catchment based emission charges or tradable emission permits are institutional. For either emission charges or a tradable emissions permit scheme to operate effectively, the point source polluter (in this case the authority responsible for catchment wide emissions) has to be able to control their level of emissions. In terms of the Swan-Canning river system this could, for example, be achieved if a catchment authority was responsible both for the level of emissions in the main drain or waterway and is able to undertake catchment level pollution abatement (such as engineering works or BMP incentive payments). The system that currently exists in the Swan-Canning river system is far removed from this model.

Under current institutional and legislative arrangements the Western Australian Water Corporation is responsible for the management of the main drain (‘point source’) system. Under their current operating license the Water Corporation faces no quality requirements for the water in the main drain system (Robinson 2002). Local governments are responsible for the extensive network of smaller drains that flow into the Water Corporations main drains.[55] Meanwhile the

[53] This approach has been suggested by the Department of Environmental Protection (2001).

[54] There are just fifteen potential participants and most easily meeting current water quality requirements.

[55] 28 local Government areas in the Swan-Canning river system are responsible for around 80 per cent of the total drainage infrastructure in Perth (Robinson 2002). Main drains, operated by the Water Corporation account for the remaining 20 per cent.

responsibility for strategic drainage planning and water quality in the Swan-Canning river system rests with the Department of Environment and Water (DEW). [56]

For any economic instrument based on the level of emissions from ‘point source’ main drains and waterways to work, considerable institutional change would be required. At the extreme, new independent catchment based authorities responsible for drainage management and water quality could be created. Stripping the Water Corporation, DEW and local governments of their current responsibilities would face massive resistance, however. Moreover the costs of making such a significant institutional and legislative change (including policy transaction costs) would be extremely large. Alternatively, overall responsibility for coordinating drainage management in Perth could be placed with either the Water Corporation or local governments (Gunningham et al 2002). The Water Corporation is well placed from a statutory and administrative viewpoint to undertake this role and the organisational model could be closely based on those used in Melbourne and Sydney (Gunningham et al 2002). Local Governments would face far greater administrative and coordination challenges in managing the entire drainage system (Gunningham et al 2002). Moreover, it is understood that local government is be resistant to taking on any additional responsibility for drainage services (Robinson 2002).

While there is now widespread recognition that changes are required to enable the proper planning and management of drainage at the catchment level, it is clear that there is very little agreement between key stakeholders as to what these changes should be and how they should proceed (Robinson 2002). Ultimately it must be conceded that even if it were physically possible to convert groups of individual polluters into point sources, the current institutional settings simply do not lend themselves to such an approach. Furthermore, any change in institutional and legislative settings associated with the creation of a more favourable drainage model are unlikely to come easily. Catchment authorities in the Swan-Canning river system, with the power and finances to respond to economic incentives provided by either emission charges or tradable emission permits, remain a pipe dream.

7. Conclusion

The management of nonpoint source water pollution presents an immense challenge to economists and policy makers alike. A complex array of physical, economic, political and institutional barriers currently lie between theoretically appealing textbook economic prescriptions and their transition into successful real-world solutions. At the same time current policy approaches to the management of nonpoint water pollution are failing to deliver the desired improvements in water quality and calls for more cost-effective economic solutions are increasing.

This places economists in a precarious position. There is undoubtedly a strong urge to replace existing measures with economic solutions that, at least on paper, look to be both more economically efficient and environmentally effective. These urges need to be tempered. Far greater attention needs to be given to where and when it is appropriate to apply specific economic instruments. It is clear that no single instrument provides a universal solution to all environmental problems – each case must be decided on its own merit. As Stavins (2001) puts it:

[56] DEW is an amalgamation of the Department of Environmental Protection, the Waters and Rivers Commission and the Swan River Trust. The amalgamation of these agencies was announced in October 2001 but is yet to be completed.

‘No particular form of government intervention, no individual policy instrument – whether market based or conventional – is appropriate for all environmental problems. Which instrument is best in any given situation depends on a variety of characteristics of the environmental problem, and the social, political, and economic context in which it is being regulated. There is clearly no policy panacea’ p 15

Greater attention also has to be paid to designing economic instruments in a way that retains their theoretical essence whilst also managing to gain the stakeholder and political support necessary for their implementation. Past experience suggests that it is simply unrealistic to try and apply theoretically ideal economic instruments to real-world water pollution problems. The obstacles are simply too great. Equally however, there seems little point in applying instruments that bear virtually no resemblance to their ideal state. Such instruments often fail to deliver significant environmental improvement and in some cases may even end up costing more than alternative measures, including regulation.

Instrument design also needs to recognise underlying beliefs about property rights and the importance of political considerations up-front. Innovative solutions, which aim to satisfy the demands of our elected officials whilst also providing for environmental improvement at a lower cost than alternative measures are required. Some degree of concession will of course be necessary. And while these may lead to more gradual environmental improvement or less than optimal economic outcomes, it will represent a positive step forward. Surely some environmental gain and some cost-savings are better than none at all!

Finally, transaction costs need to be considered in any comparison of alternative measures. For far too long analysis of the potential cost-savings offered by economic instruments compared to regulatory or suasive approaches has failed to consider the policy and market transaction costs associated with the design, implementation and ongoing operation of the economic instrument.

It is apparent that once these considerations are taken into account, many of the economic instruments that have arisen out of the economic literature are destined to never see the light of day. They are simply too complex or impractical and in some cases face insurmountable political barriers. Moreover, the benefits that result from their introduction may be insufficient to justify the associated costs.

This is not to say that economic instruments should not be considered for the management of nonpoint source pollution. In many cases the introduction of economic instruments, such as auctioned BMP payments, BMP incentive charges and emission offset banks may be both more environmentally effective and more efficient that existing suasive measures or proposed regulatory regimes. Indeed, their appeal may be even greater for water quality problems in systems other than the Swan-Canning, including those associated with salinity.

Despite their promise however, these instruments still remain unlikely to offer an entirely satisfactory economic, political or environmental solution. While offering more than most measures, they are certainly no magic bullet and the search for the holy grail of an economic instrument that can guarantee environmental and economic efficiency gains whilst also being politically palatable to key stakeholders must continue.

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