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Market instruments, ecosystem services, and property rights: Assumptions and conditions for sustained social and ecological benefits

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ABSTRACT

Market-based instruments (MBIs) are promoted as economically efficient, targeted solutions to otherwise intractable environmental policy problems with additional potential to improve the livelihood security of ecosystem service providers. This paper argues that the effectiveness of MBIs (and the likelihood therefore of sustained environmental and social outcomes) depends on a number of often unacknowledged assumptions about the distribution of benefits arising from ecosystem service provision, the rights and duties associated with resource access, and the fitness for purpose of various policy instruments. These assumptions are illustrated and discussed using The Benefit Flows and Property Rights Matrix. It is argued that the legitimacy of MBIs depends both on the demonstrability of distinct public benefit and of acceptance among the wider community that private resource users ought to be compensated in some way for the provision of that benefit. Effective provision of ecosystem services through a market-mechanism thereafter depends on a range of additional conditions including inter-changeability of supply, scalability, lack of corroboration, adequate information, financial capacity, clarity of property rights, clarity of resource access-related duties and institutional capacity. Meeting these conditions is not simply a matter of appropriate incentive design but of political decision-making, moral judgement and social learning. Failure to recognise these conditions potentially undermines the effectiveness not only of MBIs but of alternative policy measures taken contemporaneously with MBIs such as community-based natural resource management.

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Introduction

Market-based instruments (MBIs) are promoted as economically efficient and targeted solutions to otherwise intractable environmental policy problems (NMBIWG, 2005). Such instruments are diverse. They include payments for ecosystem service provision (PES), pollution taxes, cap-and-trade schemes, eco-certification and labelling, and certain capacity building measures. What they have in common is the use of market mechanisms such as trading schemes, auctions and price signals to influence peoples' behaviour in pursuit of specific policy objectives (Dargusch and Griffiths, 2008; Scott, 1998). The use of market mechanisms, it is argued, provides the least cost path to environmental outcomes by allowing flexibility for individuals in the allocation of resources and by providing continuing incentives for innovation (see Lockie, 2010). Increasingly, the use of MBIs is also seen as an opportunity to produce social and cultural co-benefits including improved livelihood security for ecosystem service providers (Engel et al., 2008; Muradian et al., 2010; Scherr et al., 2010). Eco-certification and labelling schemes attempt to achieve this by providing a basis for price premiums and more stable supply chain relationships for certified producers. Direct payments for the provision of ecosystem services de-couple payments from commodity production and provide service providers with an additional and potentially more stable income stream. Well known PES schemes designed explicitly to secure social co-benefits include the Kyoto Protocol's Clean Development Mechanism (Gong et al., 2010), the United Nations' Reduced Emissions from Deforestation and Forest Degradation scheme (Börner et al., 2010; Hoang, forthcoming) and many agri-environmental programs run under the European Union's Common Agricultural Policy (Courtney, forthcoming).

This paper offers a critical examination of the often unacknowledged assumptions underlying use of market-based instruments to pursue environmental policy objectives. To do this, it outlines a conceptual framework relevant to all arenas of environmental policy impacting on non-state resource users – *The Benefit Flows and Property Rights Matrix*. This matrix is used to highlight: first, the nature of ecosystem processes and the subsequent distribution of public and private benefits that arise from ecosystem management; second, the duty of care to the environment and other resource users implicit in resource-access rights and the extent to which provision of required ecosystem services may be accommodated within, or

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exceed, this duty; and third, the range of policy options that might be considered relevant in light of the aforementioned assumptions about ecosystem service distribution and property rights and duties. Failure to match policy instruments, it will be argued, with resource user and public beliefs regarding distributions of benefits and rights/duties is likely to undermine the legitimacy of programs and result in policy failure. Policy failure is also likely to arise from lack of fitness for purpose; that is, an inability of policy instruments to meet their stated objectives irrespective of additional and perhaps unintended social or environmental benefits. The paper will go on therefore to highlight a range of conditions particularly relevant to the effective operation of PES programs that allocate payments through the application of market instruments such as auctions and tenders.

The promise of 'the market'

There are two basic arguments fundamental to the case for MBIs in environmental policy. The first is that environmental degradation is an outcome of market failure that ought ideally, therefore, be resolved through market means (Muradian et al., 2010). This is based on the proposition that a properly functioning market will always take into account the protection and regeneration of those natural resources on which it depends. The costs of conservation, therefore, are a cost of production that ought to be internalized and passed on to consumers. However, it is argued, imperfect information, inadequately defined property rights, and/or pricing of natural resource inputs below their full economic and environmental cost all create incentives to over-utilise (Bardsely et al., 2002; Scott, 1998). Of course, market reform is not always possible in the short to medium term and a case may still be made for public expenditure in order to encourage structural adjustment or to purchase distinctly public goods. The second argument in favour of MBIs suggests that, under such circumstances, market mechanisms offer the most efficient and effective means for the allocation of resources as they create incentives to individuals which encourage competition, creativity and innovation (Bardsely et al., 2002; Dargusch and Griffiths, 2008). Regulation, conversely, is dismissed as cumbersome, blunt and ineffective. Utilising economic expertise it is now possible, MBI proponents argue, to 'design and create markets' in parts 'of the economy where this was previously impossible' (NMBIWG, 2005: 5).

The idea that environmental degradation is an outcome of market failure makes intuitive sense. Yet it is not always the case. For producers, it is *economically* rational to consider the internalization only of those environmental costs that support production. Ecological values unrelated to production are more rationally ignored. Further, standard welfare economics suggests that environmental costs should be considered examples of market failure only if the future benefits of addressing them exceed current costs (Bromley, 2007). Calculation of these benefits relies on the application of discount rates since future values are generally estimated to be worth less than current values in real terms. Proponents of environmental reform argue for lower discount rates that make investment to reduce future costs more attractive and to reflect a more precautionary approach to uncertainty over future ecosystem changes (e.g. the UK's Stern Report). Critics counter that such uncertainty should, if anything, be cause to adopt even higher discount rates. Complicating this, the costs and benefits of both action and inaction frequently fall on different actors due to the often extensive spatial and temporal scales of ecosystem processes (Bromley, 2007).

Of course, even if we accept that market failure is not always the best explanation for resource degradation, various forms of MBI may still offer the most effective policy options to deal with it. MBIs generally are classified into three broad groups (see Whitten and Shelton, 2005; NMBIWG, 2005). First, market friction mechanisms such as eco-labelling or education are used to improve the efficiency of existing markets by removing obstacles to the recognition of ecosystem services. Second, price-based mechanisms such as auctions, tenders and taxes are used to set or modify prices in order to force markets to incorporate the cost of ecosystem services. Third, quantity-based mechanisms such as cap-and-trade and offset schemes are used to set targets to achieve or maintain environmental services. This three-fold classification fails, however, to distinguish between those price-based instruments which seek explicitly to encourage *existing* markets to internalize environmental costs and those that seek, alternatively, to use price-based instruments as a means to create *new* markets through which to allocate payments for the provision of ecosystem services. The three-fold classification subsequently confuses those policy objectives that price-based instruments are inherently best suited to. It is proposed here that a four-fold classification is more useful which separates the use of price-based MBIs as tools of market reform from the use of price-based MBIs to create new markets for ecosystem services (see Table 1).

Whether evaluating the causes of degradation or designing policy interventions to address it, the most glaring weakness of many arguments in favour of MBIs is their circularity and consequent blindness to alternatives. Since failure is conceptualised as the outcome of factors external to the market, markets are seen never as the problem and always as the solution (Muradian et al., 2010). This is a non-refutable and non-verifiable article of faith. It is not a basis for sound policy. Nor does it resolve what are fundamentally moral and political questions over the internalization of costs that do not directly support production (Van Hecken and Bastiaensen, 2010). Bromley (1997: 1383) thus argues that while market mechanisms may be used to pursue well-defined policy goals they cannot be used to decide what those goals should be; for example, how clean air and water should be or how much biodiversity should be conserved.

Table 1
Typology of market-based instruments.[a]

Classification	Market intervention	Examples	Suited to:
Market friction	Improving efficiency of existing markets by removing obstacles to recognition of ecosystem services	Standards, certification, eco-labelling, ethical investment schemes, capacity building	Outcomes that can be improved through reduced transaction costs or increased information
Price-based I (market reform)	Setting or modifying prices to incorporate the cost of ecosystem services	Eco-taxes	Measurable point source activities such as carbon emissions, water extraction etc.
Price-based II (market design)	Utilising market-mechanisms to allocate payments for ecosystem services	Auctions, tenders	Diffuse source environmental outcomes such as biodiversity, salinity mitigation etc.
Quantity-based	Setting targets to achieve or maintain ecosystem services	Cap and trade mechanisms, tradable offsets	Measurable point source activities such as carbon emissions, water extraction etc.

[a] Adapted from Dargusch and Griffiths (2008), Whitten and Shelton (2005), and NMBIWG (2005).

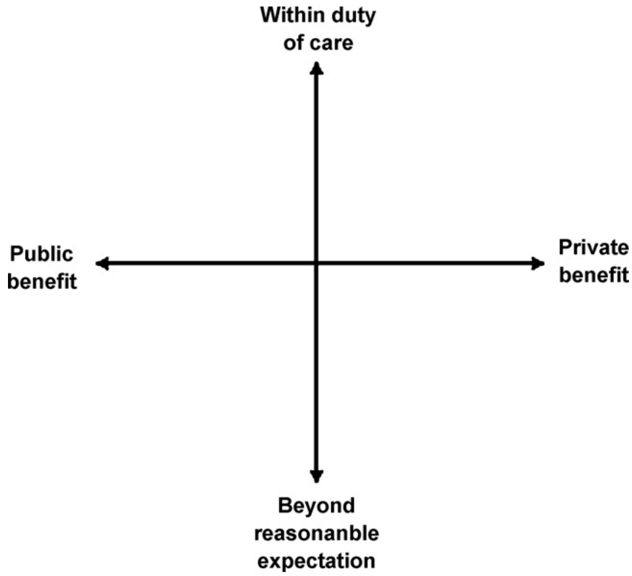

Fig. 1. The Benefit Flows and Property Rights Matrix I: distribution of public and private benefits by property rights and duties.

Ecosystem processes, benefits and property rights

Underlying all environmental policies relevant to privately owned or utilised resources are a bundle of assumptions about the nature of ecosystem processes and the distribution of services they provide, on the one hand (Bardsely et al., 2002; Engel et al., 2008), and about the rights and duties associated with access to resources, on the other (Reeve, 2001). These assumptions can be mapped on a simple matrix (see Fig. 1). The horizontal axis of Fig. 1 reflects the relative distribution of public and private benefits derived from actions to protect or enhance ecosystem processes and services. The vertical axis reflects the relative degree to which provision of desired services is seen as part of the duty of care that resource users are expected to exercise as a condition of resource access, or conversely, as beyond the capacity and/or moral responsibility of resource users.

Protecting and/or enhancing ecosystem processes provides services and benefits to private resource users as well as to the wider community. Soil conservation measures, for example, help to maintain soil fertility and structure in situ (thereby reducing fertiliser and other input costs) at the same time that they help to maintain downstream water quality, waterway function and infrastructure (thereby reducing rehabilitation and maintenance costs) (Zhang et al., 2007). Similarly, biodiversity conservation measures such as the retention of relatively 'natural' landscape features within farms help to reduce input costs through the provision of services such as pollination and pest control (Omer et al., 2010) at the same time that they help to protect both the intrinsic value and the adaptive capacity of ecosystems in the face of environmental change (Lockie, 2009).

The right to intervene in ecosystem processes is, according to Reeve (1997) intrinsic to property rights in natural resource-based industries. Property rights can be defined as a collection of entitlements and obligations associated with access to a specific resource (Reeve, 1997). Such entitlements and obligations may take a variety of institutional forms; they may be assumed as well as formalised or legislated; they may be contested; and they may vary over time. A critical feature of entitlements as they relate to ecosystem processes is the potential for resource users to transmit harms to other resource users as organisms, air, water, nutrients, sediments etc. move through landscapes in ways that transgress property boundaries (Reeve, 1997) and thus encourage their treatment as open access resources. Although rights to intervene often have a strong cultural and ideological basis (Bromley and Hodge, 1990), the transmission of harm is not inevitable. Resource users may accept an obligation, or duty of care, not to transmit harm via ecosystem processes; they may transmit benefits to other resource users; and institutional frameworks may explicitly de-couple property rights in a resource from rights to intervene in specified ecosystem process.

At face value, soil and biodiversity conserving management actions offer win–win solutions to environmental challenges for both individual resource users and the wider community. However, the ecosystem processes that link management actions with such benefits have temporal as well as spatial and social dimensions. They are complex and characterised often by high degrees of uncertainty and conflict (see Earl et al., 2010). Whether or not resource users will decide (or be compelled) to implement conservation measures will be based on (often intuitive) expectations of future benefit, preferences for high or low discount rates, capacities for investment (of capital, labour, expertise etc.), risk aversion and, critically, judgement regarding the extent to which resource users ought to be responsible for the provision of benefits either inter-spatially (i.e. to other resource users and/or the wider community) or inter-temporally (i.e. to future users) (see Bromley, 1989). This judgement will also be influenced by degrees of certainty or uncertainty surrounding expected benefits and their distribution. Explicit actions to conserve or enhance biodiversity, for example, may have no short-term private benefits provided any species lost from the ecosystem belong to functional groups represented by sufficient remaining species (i.e. other pollinators, other decomposers etc.) (see Swift et al., 2004). Under such circumstances, the primary service provided by conservation is maintenance of what might be called functional redundancy, the value of which lies in postponement of threshold effects and rapid systemic change from future species losses. In other words, the value of which is exceptionally difficult to calculate either in magnitude or in timing but which is almost certainly dispersed spatially, temporally and socially.

Ecosystem processes, property rights and policy options

Fig. 2 begins to locate broad environmental policy options within The Benefit Flows and Property Rights Matrix developed in Fig. 1. It begins with policy options that fit reasonably comfortably within each of the quadrants established by the intersection of the two axes. However, it is important to remember that benefits and responsibilities associated with ecosystem services may be shared and each axis is therefore continuous. The matrix provides a heuristic device with which to identify and illustrate important characteristics of conservation policy. Its purpose is not to develop a rigid typology of policy options or to imply an authoritative categorisation or characterisation of specific policy options, but to provide a basis from which to ask theoretically informed questions of these options. Empirical exceptions to all categorisations of policy made in this paper using the matrix are to be expected due to differences in their interpretation and operationalization across different jurisdictions and across time.

In the upper left quadrant of Fig. 2, the services provided by ecosystem management are seen to be primarily public in nature. Despite this, their provision is seen as a reasonable expectation of resource users. Under such circumstances, one of the major policy options is to regulate. Legal restrictions may be placed on particular resource uses or, conversely, specific management actions may be

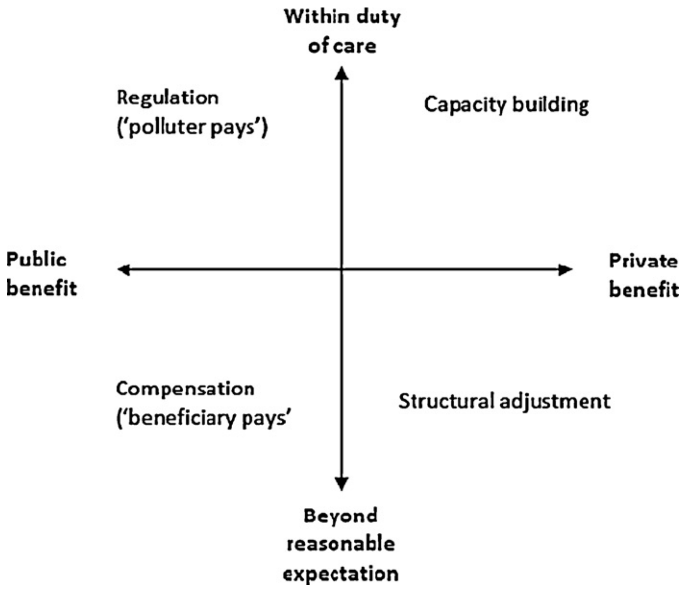

Fig. 2. The Benefit Flows and Property Rights Matrix II: distribution of benefits by property rights and duties and major policy options.

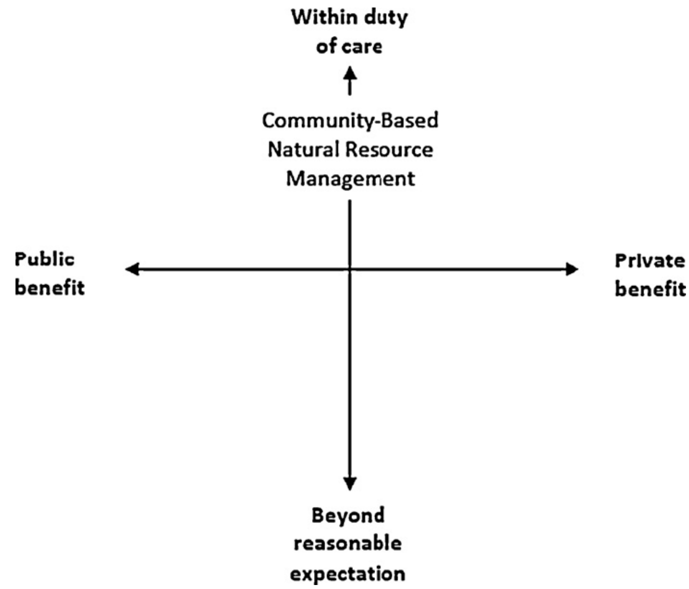

Fig. 3. Community-based natural resource management within The Benefit Flows and Property Rights Matrix.

mandated. Point source pollution control offers a classic example of this form of regulatory control as summed up by the notion of the 'polluter pays' principle.

In the lower left quadrant, services are still primarily public in nature but their provision is seen to be beyond what may reasonably be expected of resource users. Under these circumstances, a major option becomes compensating resource users for opportunity costs incurred in providing the required service (e.g. lost production) and/or subsidising the direct cost of such actions (e.g. capital investments). In this case, the 'beneficiary pays' principle is invoked. A number of arguments can be used to support invocation of this principle. On moral or legal grounds, provision of the required service could be seen to transgress the property rights of resource users or, at least, to be beyond what might be considered fair. More pragmatically, resource users could be seen to lack the financial and/or managerial capacity to provide the required service. Alternatively, state agencies may lack the political resources to introduce or enforce measures requiring private resource users to provide public benefits (see Bromley, 1997).

In the upper right quadrant, benefits are seen to be primarily private and within the reasonable expectations of resource users. Of course, regulatory controls may still mandate that resource users provide these benefits. However, as it is in resource users' self-interest to internalize these particular environmental costs education and capacity building are often preferred as a strategies to ensure self-interest is recognised and acted upon without being seen to infringe private rights. Capacity building measures may go beyond the provision of information and technical assistance to include training in business and resource planning and management, support to form self-help resource user groups and associations, and so on (Lockie, 2006). Measures such as these are sometimes though about as a policy alternative to regulatory or market-based approaches (see Whitten and Shelton, 2005). However, where capacity building initiatives are explicitly oriented towards addressing market failure either by reducing market friction or by linking economic and environmental objectives in resource management it is appropriate to consider them a form of market-based strategy (Lockie and Higgins, 2007).

In the lower right quadrant, benefits are seen as primarily private but beyond reasonable expectations of resource users. Subsidising service provision would be an option here but one that is likely to lack political legitimacy (see also Bardsely et al., 2002). Structural adjustment measures that aim to ease resource users without the capacity to provide for their own self-interest out of the industry (or which aim to provide just enough short-term assistance to boost this capacity) are a better fit with this quadrant.

Policy options do not always sit neatly within one of the four quadrants. Community-Based Natural Resource Management (CBNRM), for example, sits more-or-less in the middle of the horizontal axis and towards the top of the vertical axis (Fig. 3). CBNRM programs generally seek to encourage cooperation among resource users at a relatively local scale in order to capitalise on indigenous knowledge, encourage social learning, coordinate resource use and conservation activities, make more efficient use of community and state resources, and avoid some of the pitfalls of centralized planning, management and regulation. CBNRM programs largely accept, therefore, that resource conservation activities have both private and public benefits. Since some of these benefits accrue to other resource users, peer support (and pressure) are useful in encouraging and rewarding efforts to provide public benefits. Ultimately though, CBNRM programs do treat sustainable resource management as a responsibility that lies within the duty of care inhering in resource access rights. Cooperation between resource users is seen as an opportunity to enhance their capacity to act on this responsibility and to improve livelihoods in the process. It does not absolve them of responsibility.

Market-based instruments occupy a number of locations in the matrix. In the upper left quadrant, quantity-based measures (e.g. cap and trade mechanisms and offsets) and some price-based measures (i.e. eco-taxes) reflect the 'polluter pays' principle and its expectation that private resource users will provide public goods as a condition of resource access. Both quantity-based measures and eco-taxes attempt to institute market-reform by forcing resource users to internalize environmental costs. Eco-taxes do this most simply by requiring resource users to pay for pollution emissions or other environmental harms. But they cannot guarantee net harm reduction since resource users and consumers may simply accept some or all of this additional price (Henderson and Norris, 2008). Quantity-based measures address this issue through emission caps and the establishment of new markets in pollution and pollution offsets. The purpose of these pollution markets is to facilitate the internalization of environmental costs in a socially optimal manner

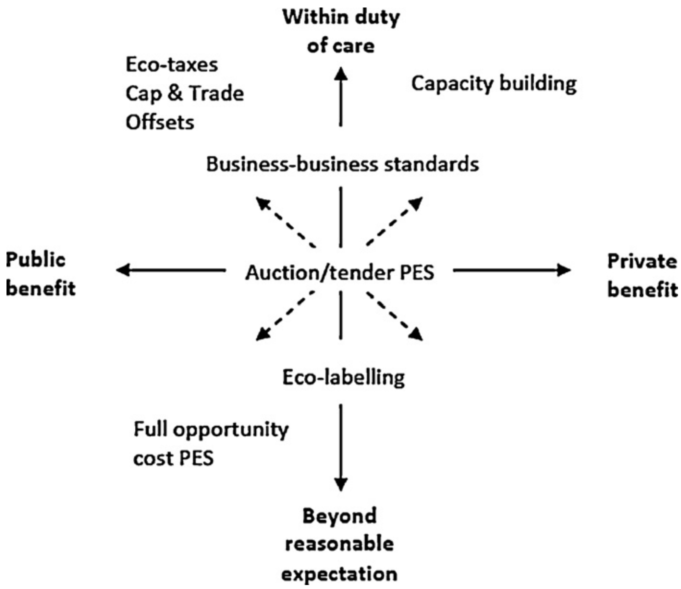

Fig. 4. Market-based instruments within The Benefit Flows and Property Rights Matrix.

across the economy. It is a means, rather than a substitute for, more far-reaching market reform.

In the lower left quadrant, full opportunity cost payment for ecosystem service programs absolve resource users of any responsibility to provide public goods through compensation for all conservation costs and all foregone production income (Engel et al., 2008; Muradian et al., 2010). Such programs do not attempt to reform existing markets but instead define public good environmental benefits as lying outside the mainstream market and the property right duties of market-oriented producers. Participation in full opportunity cost PES programs must be voluntary (Engel et al., 2008). A market mechanism such as auctions may be used to identify resource users who can provide the required service at least cost but fixed incentive payments may also be offered which users can then evaluate relative to other resource use options. In either case, a new property right is created (e.g. for biodiversity protection or lack of emissions) which resource users may then decide to trade or not (Gómez-Baggethun et al., 2010; Kosoy and Corbera, 2010).

PES schemes that utilise a market-based allocation mechanism may, alternatively, reflect the assumption that target ecosystem services provide both public and private benefits (Engel et al., 2008). They may also reflect the assumptions that resource users have at least some responsibility to provide these services even if they lack capacity or require some incentives to do so (Muradian et al., 2010; Vatn, 2010). For these reasons, auction and tender-based PES have been placed at the intersection of the horizontal and vertical axes of Fig. 4. The dotted lines reflect the possibility that individual resource users will associate varying degrees of benefit and responsibility with the required service. Application of market mechanisms such as auctions and tenders to allocate PES incentives are not just about finding those resource users who can implement desired management actions at lowest financial or personal cost. It is about finding those resource users who are willing to undertake actions at below cost on the basis that they have calculated significant private benefits from the target service, who believe they have a responsibility to provide public benefits but need a small amount of assistance to do so, who have a personal interest in relevant environmental values, and so on.

Standards and certification schemes may similarly occupy multiple positions within the matrix. In Fig. 4, eco-labelling schemes have been placed in the middle of the horizontal axis and towards the bottom of the vertical axis. Labels such as certified organic, fair trade, Forest Stewardship Council etc. communicate to consumers that producers have taken verifiable measures to internalize social and/or environmental costs (Mutersbaugh and Klooster, 2010). Consumers share responsibility for addressing social and environmental costs by paying price premiums that reward producers for their internalization. Business-to-business standards have also been placed in the middle of the horizontal axis but towards the top of the vertical axis. Such standards require resource users to demonstrate to other value chain participants that they have produced commodities in a manner that satisfies a range of quality, safety, environmental and social criteria. However, compliance with business-to-business standards is not advertised to consumers and results in no price premium for producers. It is simply an increasingly common price of doing business that protects retailers, in particular, from the reputational damage that may accrue should controversy erupt over unsustainable or irresponsible production practices (Hatanaka and Busch, 2008).

In the upper right quadrant, capacity building, education, research and other mechanisms to resolve market failures caused by inadequate or poorly understood information remain an important option for governments and resource users.

Research and education is, of course, important to efficient and effective environmental policy across all quadrants of The Benefit Flows and Property Rights Matrix. An additional feature of MBIs that bears noting, therefore, is the diversity of ways in which they may be used to manage the uncertainties, risks and controversies which confront environmental policy; that is, the conflicting data, views and interests typical of both axes of the matrix. Capacity building measures, for example, are oriented towards the substitution of environmental uncertainties with calculable risks that resource users and existing markets can then address more rationally using standard business planning and investment techniques (Matthews, 2009). They rarely confront institutionalized or assumed property rights and are seldom, therefore, controversial (except, of course, that there is 'never enough' invested in them). Capacity building measures may also need to precede or accompany other policy options.

Quantity-based measures, by contrast, are ill-suited to high levels of uncertainty. Their effectiveness and legitimacy rely on a comparatively clear understanding of the ecosystem services in question – including the relationship between management actions and target environmental outcomes – as well as a transparent, robust and broadly accepted institutional and regulatory framework for monitoring, trading etc. Establishing such certainty and legitimacy has been a major challenge for proponents of national and global cap and trade regimes to control greenhouse gas emissions. In other cases – such as the development of tradable water rights within watersheds – resource users often have embraced new market-based regimes with comparable enthusiasm despite the intent inherent in these to in fact limit access to the resource (e.g. Bjornlund, 2003). Tradable water rights and similar schemes put measurable values on resources that users may realize in a variety of ways and incorporate more overtly, again, in their regular business planning and risk management.

Standards schemes vary considerably in how they deal with uncertainty, risk and controversy. Some prohibit specific production practices with perceived potential to cause environmental or social harm regardless of whether a significant cause and effect relationship has yet been established (e.g. the prohibition of genetically engineered organisms within organic certification systems). Other standards schemes focus on quality systems designed to identify risks and goals and thereafter to demonstrate continuous improvement against these (e.g. environmental management systems). Operating at the business level on an often voluntary basis, quality standards would not generally be expected to confront resource users' perceived property rights. However, with few buyers due to the highly concentrated nature of the retail sector, many producers have little choice but to certify against retailer-preferred business-to-business standards and thus do interpret these as a constraint on property rights (Hatanaka and Busch, 2008; Lockie and Tennent, 2010).

All MBIs, therefore, regardless of type, attempt to influence the behaviour of resource users by providing them with a rational basis on which to make decisions that are in both their own and the public interest. MBIs do not ask resource users to provide public goods altruistically (at least not to any great extent) but to act as self-interested entrepreneurs (Kosoy and Corbera, 2010). It is assumed that provided instruments have been well designed, the flexible and diverse exercise of economic rationality by each resource user over both space and time will result in sufficient internalization of environmental costs and/or production of additional public goods to meet policy goals (Muradian et al., 2010). Leaving decisions to individuals in this manner has considerable potential to reduce conflict over the infringement of perceived property rights that almost inevitably follows centralized attempts to manage uncertainty and controversy, particularly through regulatory means. Some MBIs improve understanding and reduce transaction costs. Others create new opportunities to realize the value of natural resources. It does not follow, however, that MBIs are a panacea for all policy challenges. This is particularly the case where conflict is focused on the goals of natural resource policy, as opposed to the means, and/or where conflict is focused on how much responsibility private resource users ought to bear for meeting these goals. Nonetheless, the promise of reduced conflict renders MBIs extremely politically attractive. The following section will thus consider the conditions under which auction and tender-based PES ought to be considered viable policy instruments. These are of particular interest here due to the manner in which such PES largely devolve questions over the distribution of benefit flows and responsibilities to individual resource users.

Auction-based PES and fitness for purpose

The efficiency of auction and tender-based PES schemes is seen to derive from their ability to allow individuals to form their own views regarding the distribution of benefit flows from, and responsibilities to provide, targeted ecosystem services. Individual bids to provide management actions that support ecosystem processes and services through an auction or tender process subsequently will incorporate these considerations along with the estimated financial costs (direct and opportunity) of undertaking those actions. In theory, those resource users who perceive significant private benefit or responsibility will submit lower bids, all other things being equal, than resource users who perceive limited private benefit or responsibility (Commonwealth Treasury, no date). Resource users who estimate higher costs of provision or with limited financial capacity to absorb these costs themselves will, again, all other things being equal, tend to submit less competitive bids (Engel et al., 2008). Government agencies or other institutions with responsibility for managing PES schemes then evaluate bids against both the payment requested to assist in service provision and the quality of the measures proposed to provide that service (NMBIWG, 2005). Together, these calculations and processes should lead to higher overall social welfare by providing target ecosystem services at least cost, providing incentives to resource users to find innovative ways of lowering the cost of service provision further, and allowing flexibility to resource users in their business and natural resource management.

Proponents largely agree that the efficiency and fitness for purpose of MBIs when applied to particular applications rests on several criteria including their 'environmental effectiveness, cost effectiveness, wider economic effects, equity and acceptability' (Henderson and Norris, 2008: 114–115). Yet reviews suggest that few existing PES schemes are actually able to demonstrate their efficiency relative to other policy instruments (Muradian et al., 2010; Pascual et al., 2010). Common pitfalls in PES implementation have been identified as poor targeting, offering insufficient incentive to encourage participation, displacing undesirable practices to alternative locations, paying for activities that would have been undertaken anyway (or, alternatively, undermining voluntary action), failing to guarantee long-term environmental improvements, and dilution through multiple competing co-objectives such as poverty alleviation (Engel et al., 2008; Lockie and Tennent, 2010; Muradian et al., 2010; Wunder et al., 2008). Muradian et al. (2010), however, usefully observe that biophysical and social complexity is the norm for contexts in which PES may be applied, not the exception. With this in mind, it is argued here that in order to realize the potential benefits of auction and tender-based PES the following conditions must be satisfied:

Inter-changeability. There has been some discussion in the literature regarding whether PES constitute genuine market-based instruments in circumstances where there is only one buyer – often a government agency (Engel et al., 2008). The more critical issue though is whether there are sufficient sellers to ensure voluntary and competitive participation (Kroeger and Casey, 2007). Efficient and effective operation of markets in ecosystem services requires a degree of inter-changeability, or redundancy, among service providers; that is, more resource users ought to be capable of undertaking management actions in support of the desired ecosystem service than strictly speaking are required. Unless willing potential sellers can be excluded from the market, there is no incentive to submit competitive bids or to innovate to lower the cost of provision. Further, use of a market mechanism to allocate PES is more likely to generate cost savings and drive innovation where the marginal cost of service provision varies among potential sellers (Henderson and Norris, 2008).

Scalability. The combined effects of individual resource users undertaking agreed management actions under the PES scheme must provide sufficient, and appropriately connected, activity to support ecosystem processes and services that operate at larger temporal and spatial scales (Drechsler and Hartig, 2011; Stone and Wu, 2010; Wünscher et al., 2008). At the same time, the combined effects of these agreed actions should not undermine critical non-target ecosystem processes and services including those that are currently poorly understood (Kosoy and Corbera, 2010; Norgaard, 2010). Similarly, lack of conservation activity by those resource users who do not undertake to provide agreed management actions – or who drop out of PES schemes over time – must not undermine the scaling up of actions undertaken by those who maintain agreed actions (e.g. by disrupting spatial continuity or by increasing emissions) (Hartig and Drechsler, 2009; Wunder et al., 2008).

Lack of corroboration. Resource users must calculate for themselves the value and cost of providing ecosystem services in order to avoid cartel behaviour that would undermine the efficient functioning of markets. Individualising resource management in this manner must be more effective and efficient in providing target services than collaborative planning and learning activities of the sort traditionally associated with CBNRM, integrated watershed management, etc.

Adequate information. No environmental policy is implemented in a situation of perfect information (Muradian et al., 2010). Nevertheless, resource users and administering agencies need sufficient understanding of the benefits resource users and the wider community derive from target ecosystem services to submit and evaluate bids. Agencies must be capable of assessing the relative value of competing bids (Ferraro, 2008), of monitoring compliance with agreed management actions (Engel et al., 2008; Henderson and Norris, 2008), of assessing the contribution of these actions over time to target ecosystem processes and services (Wunder et al., 2008), of estimating and monitoring the social impacts of PES schemes (Muradian et al., 2010), and of remaining alert to unintended ecological and social consequences (Norgaard, 2010). In turn, resource users must have the capacity to incorporate this information and understanding in their own business and natural resource planning. In other words, while auction and tender-based PES allow individual resource users to calculate for themselves the benefits they derive from target ecosystem services, these calculations should incorporate scientific and other knowledge and they should evolve through participation in the PES scheme. Incentives will not be allocated in an environmentally and socially optimal manner if bidders otherwise capable of providing desired management actions are excluded by ignorance.

Financial capacity. Although PES offer financial incentives, their effectiveness as market-based instruments is premised in part on resource users' recognition of the private benefits they derive from ecosystem services. If resource users face substantial short-term financial pressures they may discount the long-term financial benefits of taking immediate action to support ecosystem processes and services more heavily in order to secure short-term cash flow from resource exploitation (see Bromley, 2007), and thus submit 'high' and uncompetitive bids. Under these circumstances, PES payments will tend to accrue to comparatively wealthy resource users who are able to supply more environmental service and thus optimise the efficiency of the scheme (Muradian et al., 2010). Conversely, poor resource users may not discount the long-term benefits of immediate action heavily enough and submit 'low' bids in order to secure cash flow from PES payments (Kosoy and Corbera, 2010). Either way, comparatively wealthy resource users are in a much stronger position to decide for themselves the circumstances under which they will supply ecosystem services.

Clarity and acceptability of property rights. PES create new property rights in service provision designed to encourage the internalization of environmental costs. This allocation depends, in turn, on the distribution of existing property rights (Muradian et al., 2010) and the institutional and ideational frameworks on which these rights rest (Gómez-Baggethun et al., 2010). The effectiveness of PES, therefore, depends both on the ability of administering agencies to direct payments to the appropriate resource users (that is, to those capable of providing or undermining the required services) (Vatn, 2010) and on the perceived fairness of these payments (Pascual et al., 2010; Sommerville et al., 2010). Where existing degradation or exploitation is related to a lack of clarity over who holds rights to access or manage a resource, or to a lack of equity in resource access, this may prove difficult (Engel et al., 2008; Engel and Palmer, 2008; Gong et al., 2010). It may also provide conditions for the appropriation of traditional access rights by elites (Scherr et al., 2010). Addressing questions of distribution and equity is not a distraction from the efficiency goals of PES but a prerequisite (Muradian et al., 2010; Pascual et al., 2010).

Clarity of resource access-related duties. PES should target public monies towards activities that are above and beyond what might reasonably be expected of resource users. While auction and tender-based PES allow individual resource users to decide for themselves the amount of assistance they will request to provide public benefits, the political legitimacy of PES schemes ultimately will depend on a reasonable degree of congruence between the 'duty of care' implicit in resource users' bids and the expectations of peer networks and the wider community. This is likely to be complicated by at least two factors. First, the availability of economic incentives may change resource users' perceptions of their own duty of care and lead to what has been termed a 'crowding out' of altruistic behaviour (Vatn, 2010). In other words, resource users may come to expect financial assistance to undertake activities they otherwise would have seen as their own responsibility. Second, the acceptability of implicit assessments of duty of care will be complicated by existing distributional inequities among resource users. What is considered a fair and reasonable expectation of comparatively poor resource users may not be the same as what is considered fair and reasonable to expect of comparatively wealthy resource users. Significant conflict, where it exists, over the duty of care that ought to inhere in property rights will not be resolved through devolution of decision-making through PES to one group of stakeholders.

Institutional capacity. PES are not, as mentioned above, an alternative to planning and governance (Ring et al., 2010; Vatn, 2010). They depend on the capacity of state or other agencies to define and enforce appropriate property rights, identify and mobilise sellers, solicit trust, act on behalf of absent buyers, monitor PES implementation and outcomes, and reduce information deficits and associated transaction costs (see also Börner et al., 2010; Clements et al., 2010; Scherr et al., 2010).

Where any of the above criteria cannot be met, serious consideration should be given to alternative policy instruments. This is particularly critical given that auction and tender-based PES schemes come with their own opportunity costs. For example, the principle of non-corroboration has the potential to close off or crowd out alternative approaches such as community-based natural resource management that encourage resource users to collaborate at larger scales (Muradian et al., 2010). At the same time, introduction of the 'beneficiary pays' principle has the potential to undermine resource users' perceived duty of care (Clements et al., 2010). Where this occurs, provision of ecosystem services may increasingly be restricted to payment recipients and only for the duration of those payments.

In the absence of inter-changeability or redundancy among potential service sellers, fixed price PES, transparent sector-wide negotiation over compensation payments, and/or non market-based policy instruments such as CBNRM are likely to be more efficient as they avoid the incentive to submit unrealistically high (i.e. rent seeking) bids due to lack of competition in the bidding process. Such lack of inter-changeability may be a problem either in situations where only a small number of resource users are capable of providing the required service or in situations where most, if not all, potential sellers need to be involved in a program in order to scale management actions sufficiently to secure the target ecosystem service (Kroeger and Casey, 2007). Importantly, scaling up is not always about getting more resource users involved in a program. It may also be about getting the right resource users involved; e.g. farmers with contiguous landholdings in order to provide spatial continuity to revegetation programs and thus facilitate wildlife movement and adaptability (Hartig and Drechsler, 2009). Alternative instruments including fixed payment PES are also likely to be more efficient where sellers lack the financial capacity to apply a modest discount rate when calculating the cost of service provision. To be attractive to potential sellers in these circumstances, fixed payments may need to approach the full short-term opportunity cost of service provision or, alternatively, be accompanied by additional policy measures either compelling resource users to act or providing additional non-monetary assistance (Clements et al., 2010).

Conclusions

Market-based instruments offer a potentially useful addition to the suite of policy options available to resource management agencies. However, they are not a panacea for all hitherto intractable policy problems and, like all other policy options, come with their own range of opportunity costs. Unfortunately, many arguments in favour of MBIs are based on circular logics which fail to acknowledge that market failure is not always the cause of resource degradation or that more markets are not always the solution. Where MBIs fail, such logics risk rationalising failure as symptomatic of 'design flaws' to be addressed through the application of more technical expertise rather than as the potential consequence of inappropriate instrument choice and the need to embed instrument choices in effective planning and governance regimes (Norgaard, 2010). As the discussion of auction and tender-based PES sought to demonstrate, MBIs can be fit for purpose when appropriate conditions are met (in this case, inter-changeability, scalability, lack of corroboration, adequate information, financial capacity, clarity and acceptability of property rights, clarity of resource access-related duties and institutional capacity). When such conditions cannot be met, alternative measures are likely to be more effective and efficient.

As The Benefit Flows and Property Rights Matrix sought to demonstrate, all environmental policy embodies a range of assumptions regarding the dynamics of ecosystem processes, the distribution of benefits arising from these, and the relative responsibilities of resource users to protect or provide valued ecosystem services. One of the appeals of market instruments is the flexibility they allow individuals in valuing and managing ecosystem services. Yet, it has been argued, even though MBIs such as auction and tender-based PES may occupy a variety of locations within the matrix, they are unlikely to operate effectively or maintain their political legitimacy if significant conflict among interested stakeholders is not addressed through more deliberative means (Kosoy and Corbera, 2010). As measures that invoke the 'beneficiary pays' principle, PES are particularly likely to suffer a deficit of legitimacy when they are seen as reflections of the political incapacity of government agencies to enforce environmental regulations, or as means to channel funds to larger and wealthier interests, rather than as fair and reasonable responses to what may otherwise be beyond the capacity of resource users to deliver. PES are also unlikely to operate effectively if potential bidders or the agency staff evaluating these bids understand little about the distribution of public and private benefits from target ecosystem services, or if they are insensitive to issues of distribution.

Asserting the technical expertise of instrument designers or the flexibility offered by MBIs to potential participants does not obviate the need to underpin incentive design and operation with political decision-making, moral judgement and social learning (Gómez-Baggethun et al., 2010). Failure to recognise these conditions potentially undermines the effectiveness not only of MBIs but of alternative policy measures taken contemporaneously with MBIs such as community-based natural resource management. This is particularly likely when MBIs and other policy measures embody, and therefore communicate, fundamentally different assumptions about the duty of care inhering in resource access and management rights.

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