A critical examination of Payments for Ecosystem Services (PES) as applied in a watershed management context

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Abstract

Interest has grown significantly in advocating market-based policies for water resources management in response to: a) increasing global deterioration of watershed ecosystem services, b) complex and uncertain socio-economic and ecological drivers of land-use change in a watershed setting, and c) the need to respond to the criticism of an 'implementation gap' in achieving 'Integrated Water Resource Management' (IWRM). Chief among these policy mechanisms has been 'Payments for Ecosystem Services' (PES) which are theoretically premised as a voluntary and conditional transfer of incentives from ecosystem service beneficiaries to land-users to either improve the delivery of the identified ecosystem service(s) or more commonly to implement land-use practices which encourage those services.

The present research contributes to the trend of critical analysis of PES by: a) developing a novel governance framework for ecosystem services in a watershed context; b) applying a methodological tool for assessing technical and social complexities for socio-ecological conflicts to a proposed PES scheme for watershed management; c) linking ecosystem services to discrete improvements in human well-being through a consideration of Amartya Sen's 'capabilities approach'; d) exposing the trade-offs implicit in targeting payment schemes for diverse objectives, and e) exploring the behavioural economics assumptions of PES in a context at the frontier of the free-market and exchange values for public goods and services. Few studies examining PES feasibility have taken an explicitly *justice*-oriented lens to considering fairness, equity for ensuring long-term negotiation outcomes. We argue that an emphasis on justice will not only place the often neglected socio-political and power asymmetries that mar IWRM efforts closer to the centre of scrutiny, but will also facilitate greater empowerment and ultimately longer-term success in achieving both ecological and human well-being aspects of water management.

This research recognises the imposition of power in defining PES premised as a market-based tool which prioritises one form of value (economic value) over others, making justice (or the lack thereof) a critical linchpin for the long-term success of PES in achieving an integrated and adaptive vision for watershed management. A conceptual framework is developed to identify the

misalignment between the characterisation of ecosystem services as economic goods and their treatment within PES as exchangeable commodities. In this framework, it is clearly identified that PES as a policy mechanism to foster integrated and adaptive water resource management will work best in settings where ecosystem services are characterised as non-rival yet potentially excludable and where ecosystem benefits from watershed stewardship accrue at the household and community scale. The framework clearly precludes PES feasibility for ecosystem services that are either non-rival and non-excludable or rival but not excludable. Moreover, the framework clearly nests the management of ecosystem services hierarchically according to their economic characteristics. Thus, excludable and rival goods and services are best managed according to market principles, but only after ecosystem services possessing other combinations of rivalry and excludability are effectively safeguarded. Current conservation efforts, as exemplified through traditional PES theory, have considered all ecosystem services as though they are rival and excludable and have thus failed to understand the inherent and relational characteristics of the well-being that is derived from ecological structure. The continual drive to develop creative new rival and excludable commodities for exchange is thus out of touch with biophysical reality and thus grossly underestimates the value of life-supporting goods and services which are not amenable to market exchange.

Two empirical studies are provided based upon a proposed PES scheme in the Shivapuri-Nagarjun National Park located in the Kathmandu Valley of Nepal. The first of these illustrates how market-based logic in spatially optimising payments fails to resonate with social institutions and values and inadvertently separates society from nature in a piecemeal fashion. Spatial interpolation according to household characteristics was employed using ArcGIS to map areas of low opportunity cost, ecological vulnerability and poverty. These maps were compared with the preferences, constraints, and informal institutions of upstream service providers to illustrate trade-offs between externally determined PES targeting designs and local legitimacy. It is argued that the mechanistic ‘puzzle-matching’ of targeting payments according to efficiency and effectiveness continues to treat society as separate from nature and risks reinforcing perceived inequities. The second study uses social multi-criteria evaluation as an alternative methodology for designing a number of PES and other management alternatives for Shivapuri-Nagarjun National Park. This method is based upon a deliberative negotiation of values, priorities and

opportunities for improving individual and collective well-being and which do not allow for certain value sets to compensate others, but to expose the trade-offs inherent between equally legitimate perspectives. The criteria to evaluate different management alternatives were chosen to reflect individual and collective deprivations of functioning for upstream communities in order to enhance overall capabilities. Despite strong concordance for alternatives which build upon buffer zone management and decentralised decision-making for devising resource management rules in the park, several PES payment alternatives were illustrated to be more socially acceptable once the risk of social conflict over a divergence of top-ranking alternatives among two distinct coalitions of social actors was revealed. This research is the first to use social multi-criteria evaluation as a transparent and deliberative valuation method to compare PES and non-PES management alternatives for addressing a socio-ecological conflict over water quality services. It is also the first to consider social justice in the design of payments to reflect improvements in capabilities.

A final empirical study carried out at a pilot watershed PES programme in the Kyrgyz Republic examines the behavioural aspects of rational self-interest which characterise the assumed motivations underlying participation in PES schemes. Scholarship in the field of behavioural economics has illustrated how decentralised norm enforcement, norms of reciprocity, framing and image motivation may explain behaviour in social settings that override assumptions of purely rational and self-interested actors. In this study, different sets of incentives and framings of incentives are introduced to examine the extent to which they motivate participation in the maintenance and repair of collectively owned irrigation canals. The activity forms a part of a cultural tradition in the semi-nomadic areas of Central Asia known as *ashar*. Four collective activities took place in each of four different incentive and non-incentive treatments; a series of post-activity interviews and focus groups were conducted to triangulate motivation for participating in the activities. The exploratory study illustrates that the prioritisation of self-interested norms over existing social institutions can result in unintentional outcomes depending on the framing and strength of different sets of norms in a given context. PES incentives may be more successful in motivating behaviour where they positively reinforce existing social norms, without replacing them.

The implications of this thesis suggest that incentive-provision for watershed services has greater potential to be integrated within collective resource management than as a market-based tool. Appreciating the effect of incentive provision to encourage greater collective action can offer few generalising heuristics with the exception of efforts which facilitate or reinforce existing social institutions and norms and place recognition and procedural justice as core components of negotiated outcomes.

Résumé

L'intérêt d'appuyer des politiques fondées sur le marché pour la gestion des ressources en eau survient en réponse à : a) une dégradation de plus en plus élevée qui touche les services écosystèmiques parvenant des bassins versants; b) l'incertitude et la complexité des pressions socio-économiques et écologiques, modifiant les pratiques d'exploitation des terres dans un contexte d'un bassin versant; et c) la nécessité de répondre aux critiques sur l'implantation de la Gestion Intégrée des Ressources en Eau (la GIRE). Un des plus importants outils politiques des dernières années sont les "paiements pour les services écosystèmiques" (PSE) basés sur la théorie d'échanges volontaires et conditionnels misant sur les incitatifs pour la conservation de la nature. Il s'agit d'une compensation via laquelle les paiements sont liés à l'élaboration de réglementations spécifiques (écologiques ou sociales) appelées "conditionnalités". Le transfert des incitatifs rendu par les bénéficiaires d'un service écosytèmique aux utilisateurs de la terre en aval doit être effectué pour : a) améliorer l'approvisionnement des services; b) assurer l'entretien écologique de la terre; ou c) transformer certains comportements pour la réalisation des services écosytèmiques désirés.

Le vif intérêt que suscitent actuellement les mécanismes de PSE pour la gestion des ressources naturelles est supporté par une argumentation fondamentalement économique sur laquelle l'importance accordée à l'efficacité supplante les politiques de préservation et les initiatives communautaires pour la conservation de la nature, car ces dernières manquent d'obligations conditionnelles à l'entretien de la terre. La logique économique qui sous-tend les PSE se réfère à l'internalisation des coûts sociaux liés à la détérioration des ressources naturelles. Les services écosystèmiques étant de nouveaux produits produits (ou objets d'échange) à l'échelle du marché incarnent de meilleures déterminants dans la prise de décisions.

La présente recherche propose une analyse conceptuelle critique et empirique dans le cadre des PSE fondés sur les principes du marché afin d'avancer la gestion de l'eau intégrée et adaptée. La complexité des caractéristiques économiques, spatiales et temporelles des services économiques de même que les valeurs diverses souvent conflictuelles, pousse la recherche actuelle à

reconnaître les jeux de pouvoirs définissant la base des PSE en tant qu'instrument économique. La base des PSE priorise la valeur (principalement économique) sur les autres, positionnant la justice (ou son manque de) à un point tournant pour le succès long terme des PSE dans l'atteinte d'une vision de la gestion de l'eau intégrée et adaptée.

Un cadre conceptuel est développé pour démontrer le décalage existant entre la caractérisation des services écosystèmiques en tant que biens économiques et leurs traitements dans le contexte des PSE comme produit d'échange. Deux études empiriques ont été effectuées basées sur un projet de PSE prévoyait à la Vallée de Katmandou au Népal. La première de ces études illustre le revers de l'application des instruments économiques en ciblant l'optimisation des paiements dans une logique de marché, car via une vision divisant la nature et la société, ces instruments faillissent à atteindre les institutions sociales et les valeurs. La deuxième étude utilise l'évaluation sociale multicritère comme approche délibérant pour conceptualiser des incitatifs à la gestion des ressources naturelles à travers un processus de négociations ouvert à la communication des valeurs, priorités et opportunités pour l'amélioration due bien-être collectif et individuel. Cette évaluation ne permet pas la compensation d'un groupe par l'usage d'une valeur, mais incite plutôt la réalisation des échanges nécessaires à l'élaboration d'un compromis entre les perspectives des groupes impliqués.

La dernière étude empirique à été réalisé sur un site pour une étude de cas pilote d'implantation des PSE à un bassin versant au Kirghizstan. Le but de cette étude est d'investiguer les aspects comportementaux de l'intérêt personnel-rationnel motivant la participation des individus à s'inscrire à un projet de PSE. Cette étude exploratoire à démontré la prépondérance accordée à l'intérêt personnel-rationnel par rapport aux règles sociales collectives, ce qui peut produire des résultats non-désirés en fonction du contexte, de l'encadrement et des normes sociales déjà en place dans le lieu d'implantation.

Les implications de cette recherche indiquent que la promotion des incitatifs pour la gestion de l'eau sont mieux positionné sous la théorie de l'action collective et la gestion des biens communs qu'en tant qu'outil de marché. En effet, l'encadrement des paiements en tant qu'internalisation des externalités du marché restent trop à l'écart des réalités des asymétries hors de contacte de

pouvoir scalaires et des incertitudes temporelles qui exacerbent les inégalités ce qui mène à des impacts écologiques négatifs. Par contre, bien que les heuristiques de bonnes pratiques ne peuvent être généralisés, la justice procédurale et la reconnaissance des valeurs diverses sont des éléments fondamentaux des résultats négociés.

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Dedication

I dedicate this research to the plural voices of the masses implicated in complex and dynamic social and ecological conflicts around the world.

Contributions of the Authors

The first manuscript of the thesis included herein is by: V. Kolinjivadi, J. Adamowski, and N. Kosoy. The second manuscript is by: V. Kolinjivadi, A. Grant, J. Adamowski, and N. Kosoy. The third manuscript is by: V. Kolinjivadi, G. Gamboa, J. Adamowski, and N. Kosoy. The fourth manuscript is by: V. Kolinjivadi, S. Charré, J. Adamowski, and N. Kosoy. These manuscripts are presented as chapters of the thesis and have been presented at scientific conferences and have been already published or have been submitted for publication in peer reviewed journals.

The author of this thesis was responsible for conceptual development, sampling methodology, data collection, analysis, and preparation of manuscripts for publication. Dr. Jan Adamowski is the thesis supervisor and was responsible for reviewing and editing each manuscript. Dr. Nicolás Kosoy is the co-supervisor and contributed critical input in conceptual development of each manuscript as well as valuable technical guidance in analysing the collected data. Angela Grant was a graduate student in the Department of Bioresource Engineering at McGill University and assisted in carrying out spatial analysis of the collected data in the Sundarijal catchment of Nepal. Gonzalo Gamboa of the Autonomous University of Barcelona, Spain provided valuable technical guidance in carrying out the social multi-criteria evaluation. Lastly, Simon Charré of The Regional Environmental Center for Central Asia (CAREC) contributed local supervision and technical advice on the particulars of the case study description as well as in the final stages of manuscript submission.

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List of Abbreviations

ADB- Asian Development Bank

AM- Adaptive Management

CAREC- The Regional Environmental Center for Central Asia

CBS- Central Bureau of Statistics (Nepal)

DDC- District Development Council

EIA- Environmental Impact Assessment

FAO- Food and Agricultural Organization

HPCIDBC – High Powered Commission for Integrated Development of Bagmati River Civilization (Nepal)

IAWRM – Integrated and Adaptive Water Resource Management

ICIMOD- International Centre for Integrated Mountain Development

IUCN- International Union for the Conservation of Nature

IWRM – Integrated Water Resource Management

JICA- Japan International Cooperation Agency

KUKL- Kathmandu Upatyaka Khanepani Limited (Nepal)

KVWSMB- Kathmandu Valley Water Supply Management Board

NAIADE- Novel Approach to Imprecise Assessment and Decision Environments

NEA – Nepal Electricity Authority

NETIF- Nepal Environment and Tourism Initiative Foundation

NGO- Non-governmental organization

NPV- Net Present Value

NWSC- The Nepal Water Supply Corporation

PES- Payments for Ecosystem Services

PSAH- Pagos por servicios ambientales hidrológicos (Mexico)

REDD- Reducing Emissions from Deforestation and Forest Degradation

SD- Standard Deviation

SMCE – Social Multi-Criteria Evaluation

SNNP / ShNP- Shivapuri-Nagarjun National Park

VDC- Village Development Council

WAFED- Water and Energy Users' Federation (Nepal)

WUA- Water User Association

Chapter 1

GENERAL INTRODUCTION

Solving environmental problems in the world today represents one of humanity's greatest challenges. The environment is a site of conflict between multiple lenses, competing values and perspectives of different communities and interest groups across spatial scales and between present and future generations. Different groups of people have legitimate values which conflict with each other and result in the differential distribution of goods and bads across the landscape and over time (Martínez-Alier et al., 1998). The sheer complexity of the global environmental problems that humanity is facing requires a transdisciplinary perspective which does not rest at simply bringing together disciplines, but instead seeks to transcend a single discipline in order to develop a coherent effort in addressing these problems. The interaction of ecological, economic, legal, political, psychological, technological and social perspectives in explaining our present predicament indicates that any proposed solution which appears to be 'objective' may be grossly underestimating the complexity of the true problem. Farley's (2010) metaphor of the compound eye of a dragonfly serves to illustrate how each of the thousand individual lenses contributes a particular understanding of the world around us. However, being aware of the compound lens yet relying on a single lens will result in a failure to arrive at a sustainable future.

In response to alarming and rapidly increasing environmental degradation across the world, substantial efforts have been made over the last two decades to better understand society's dependence on ecological support systems (Daily, 1997; de Groot et al., 2002). These efforts have gained substantial traction in the conceptualisation of 'ecosystem services' as benefits that humans derive from nature. These include *provisioning* services such as timber, food, medicine and water; *regulating* services, such as protection from climate disasters and disease; critical *supporting* services such as nutrient cycling and soil formation; and *cultural* services referring to spiritual, recreational and other non-material benefits (Millennium Ecosystem Assessment, 2005). The Economics of Ecosystems and Biodiversity (TEEB) expands upon this definition by identifying 'ecosystem functions' as the assemblages of ecological structure and processes that underlie the capacity of an ecosystem to provide goods and services (Abson et al., 2014). The

term has evolved from being originally conceived as an awareness-raising concept to garner greater political support for ecological stewardship to the commodification of ecosystem functions as packaged services to be sold in markets (Gómez-Baggethun et al., 2010; Peterson et al., 2010). Market-based approaches are underpinned by the economic theory of Robert Coase (1960) who argued that if property rights are established and transaction costs are low, the voluntary exchange of public goods will lead to the most efficient outcomes. Of these market trades for ecosystem services, payments for ecosystem service (PES) schemes have rapidly become a buzzword within natural resource management as a means to incentivize ecological stewardship and simultaneously improve the welfare of negotiating agents in an economically efficient manner (Wunder, 2005, Pagiola and Platais, 2007; Engel et al., 2008). These schemes are premised upon an assumption that the undersupply of ecosystem services is the result of market failures and hence aligning values with market prices will solve environmental degradation problems (Muradian et al., 2013). PES schemes are often heralded as being the most cost-effective means of achieving environment and development goals, particularly in developing countries (Ferraro and Kiss, 2002).

PES schemes have become particularly prolific in relation to watershed management due to the convenient upstream-downstream conflicts associated with river basins corresponding to service beneficiaries (downstream) compensating potential service providers (upstream) for altering land-use practices in specific ways (Landell-Mills and Porras, 2002). It is not hard to understand why PES is so conceptually attractive. The often divergent but politically salient goals of environmental conservation and poverty alleviation, brought together within the arena of PES, is of great interest amongst conservation practitioners, water managers, governments, donors, and social scientists alike. According to Wunder (2007), PES schemes are characterised as voluntary transactions where a well-defined ecosystem service is "bought" by at least one ecosystem service buyer from at least one ecosystem service provider if and only if the ecosystem service provider(s) secures the delivery of the service. This latter point is a key distinguishing element of PES and is referred to as conditionality. Typically, PES schemes that adhere to the theoretical definition offered by Wunder (2007) apply strict conditionality, indicating that payment to a land-user is dependent upon evidence that changes in land-use behaviour have resulted in the delivery of identified ecosystem services. Given the time lags and difficulty in procuring

ecosystem services within an expected time period, more often conditionality is assessed in a less strict form by which payment is made if the land-user has adopted land-use practices that are believed to provide the ecosystem services of value (Muradian et al., 2010). In order to determine who can be considered a potential service provider or recipient of payment and a potential service beneficiary or 'buyer', property rights are required to be established prior to the arrangement of a PES scheme (Rawlins and Westby, 2013).

While the adoption of PES arrangements according to this definition has expanded at great speed over the last decade, there have been limited examples of success of these arrangements. Indeed, only 11 PES were clearly identified as user-financed (i.e. direct beneficiary pays) or government-financed (i.e. government finances on behalf of beneficiaries) initiatives which more or less conformed to the theoretical definition (Wunder et al., 2008). Meanwhile a recent synthesis of 9 PES initiatives primarily in South and Southeast Asia determined that the preconditions for the Coasean conceptualisation of PES were hardly met. This was due to the lack of capacity to measure ecosystem service delivery, unstable funding, unclear property rights, and unfair burdens on more marginalised community groups (particularly since risks of default and biophysical uncertainties are largely shouldered by poorer service providers than those providing the payments) (Leimona et al., 2015). PES schemes which have most closely adhered to the theoretical definition do exist, but are few and far between. A PES scheme in which Nestlé-Vittel Water, a private company, paid 27 upstream dairy farmers according to their opportunity costs as well as additional management costs for reducing grazing pressure represents a highly publicized example of PES (Wunder et al., 2008). In order to arrive at 'acceptable' compensation, a process of negotiation between the company and service providers lasted 10 years in order to negotiate terms and conditions respective to individual farmers. However, the non-agricultural sector (upstream golf courses) also responsible for impacts on downstream water quality, have not been included in the transactions. Moreover, the total cost of the scheme has spiraled beyond USD25 million without any indication of whether the PES initiative represents a more 'efficient' tool than alternative environmental policies would have (Perrot-Maître, 2006). As Leimona et al. (2015) have argued, the few PES schemes that have been successful in conforming to the theoretical definition have been characterized by: a) minimal service providers; b) clear tangibility or measurability of localized ecosystem services, and c) strong social capital or

relationships of trust between actors. Given, the complexity of interactions between social and ecological systems at multiple scales combined with systemic and growing uncertainties associated with climate change, the ability for the actions of a few discrete individuals to influence even localized ecosystem services is becoming increasingly less predictable. Furthermore, a growing body of literature has raised critical inquiry on the imposition of a single value ethic and the failure to incorporate social and political realities to PES contexts in order to better separate theoretical attractiveness from short-term reality and long-term consequences (Matulis, 2014; Kallis et al., 2013; McAfee and Shapiro, 2012; Van Hecken et al. 2012; Gómez-Baggethun et al., 2010; Muradian et al., 2010; Pascual et al., 2010; Vatn, 2010).

In relation to integrated water resources management (IWRM), the concept of ecosystem services and the role of PES in achieving an IWRM vision have come to dominate recent research for governing water resources (Liu et al., 2013; Jax, 2010). As defined by the Global Water Partnership, IWRM refers to:

> "a process which promotes the coordinated development and management of water, land and related resources in order to maximise the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (GWP, 2000: 22).

This definition has been enriched by Medema et al. (2008) who argued that IWRM is a continuous balancing process in making trade-offs in an informed way through the consideration of diverse goals and values. Essentially, this recognition emphasizes an adaptive and co-evolving dynamic intrinsic to complex socio-ecological relationships, in which nonlinear relationships are the rule rather than the exception. Accordingly, the IWRM framework has faced growing criticism in relation to an empirical implementation gap (Mitchell, 2005; Biswas, 2004), high uncertainty, subjectivity, and dynamic social, political and ecological interactions. Moreover, the IWRM scholarship has tended to perceive water management from a purely technocratic challenge of efficient supply-side techniques or upstream-downstream analyses and has been less able to consider the inherently political influences of water management, particularly unequal power relations in the promulgation of knowledge and technology (Norman and Bakker, 2009;

Rahaman and Varis, 2005). While the IWRM framework does not explicitly recognise the concept of ecosystem services as they pertain to watershed areas, the emphasis on linking water resources to social welfare firmly inherent in the IWRM mandate identifies the close alignment that a focus on ecosystem service maintenance and delivery would have within broader water resource management. Further, the lack of concrete policy mechanisms to actualise IWRM gives additional credence to PES as a mechanism to achieve this end. As Cook and Spray (2012) argue, both IWRM and the ES concept have emerged arising from a preoccupation towards determining pure objective and rational ideas within a context of substantial uncertainty and inherently political conflicts of interest.

1.1. Problem Statement

There is a highly contested debate in the current academic literature regarding so-called "market-based" instruments for environmental policy and natural resource management (Pirard, 2012). These instruments refer to a range of policies including cap-and-trade permits, biodiversity offsets, certification schemes, water pricing and PES among others. Lockie (2013) has argued that such instruments are effective only under particular institutional contexts in which ecosystem metrics and financial exchange values have been developed and are considered culturally acceptable. Others, such as McAfee and Shapiro (2012), strongly argue that "selling nature to save it" through market commodification has served to drive apart nature and society by falsely assuming preordained financial values for nature based on individual gain and simple cause-and-effect relationships between ecosystem processes and the delivery of ecosystem services. The practices of land privatisation, commodification of ecosystem services and an institutional apparatus (often involving government support) for market exchange represents what Castree (2008) calls the 'neoliberalisation of nature'. Predicated according to pure markets, such policy tools risk ignoring the power dynamics entrenched within the broader political economy and which affects and perhaps reinforces inequities in local resource access and control (McAfee and Shapiro, 2012). Ultimately, market-based approaches fall under a paradoxical logic of 'green capitalism' by which capital accumulation, causing environmental degradation associated with the global economic system, is proposed once again as a solution to address the problems caused by inherent capital accumulation (Muradian et al., 2013; McAfee, 2012). Given

that predicted financial flows of nearly US$30 million a year are being invested in environmental market-based mechanisms, (Yanez-Pagans, 2013) there is much to be gained in identifying potential risks of PES implementation, to propose alternative methodologies for PES design, and to critically examine the theoretical and empirical assumptions underpinning the concept. It is the purpose of this research to illuminate upon these issues.

With specific reference to PES, a number of authors have argued how the justification of payments according to a single-metric exchange value may confound social relations over inherently public and intangible benefits and even result in the deterioration of ecosystem services in contexts inappropriate for environmental commodification (Kosoy and Corbera, 2010; Spash, 2011). An assumption is made that all forms of value that society holds towards ecosystem benefits, however they are conceived, are amenable to quantification through monetary flows. Accordingly, human beings as rational actors will always respond in their self-interest and hence behavioural change can be effectuated by creating the correct price signals (Engel et al., 2008). However, few studies have actually proven the assumed premise that the diversity of behavioural responses will always respond to price signals as predicted (Fehr and Falk, 2002; Bowles, 2008). Moreover, there have been minimal attempts to develop heterodox methodologies in the design of PES arrangements which consider the plurality of values that communities may have in relation to their own conceptualisations of nature's benefits (Vignola et al., 2010). It is thus imperative to determine the institutional and social circumstances in which incentive provision can either strengthen or weaken intrinsic motivation to be effective stewards of ecosystem services.

It is worthy to mention that the above criticisms regarding a market-based logic underpinning PES implementation draws a parallel with many of the concerns underlying IWRM in recent years. That PES is increasingly being considered as a means by which an IWRM vision of water management can be achieved is then particularly worrying if both are aligned from a purely technocratic, positivist and utilitarian approach for considering natural and human dimensions of water (Bidaud et al., 2013; Cook and Spray, 2012). As McDonnell (2008) has argued, the use of spatial analysis linked through mathematical modelling and combined using weighting procedures and matrix-based procedures are far from an objective means for capturing the

complexities of ecological, economic, cultural and political interactions. Research critically examining the theoretical, design and methodology of PES in light of the criticisms of IWRM is therefore essential if we are to avoid making poor investments in strategies which could undermine the long-term delivery of watershed ecosystem services and result in greater social disparities.

Recent scholarship critically analysing PES schemes have contributed to the theoretical development of the concept by establishing the institutional parameters in which PES conceived either as markets or incentive-based negotiations can most favourably improve resource management (Fisher et al., 2010; Muradian et al., 2010; Vatn, 2010). A number of studies have sought to either reveal the inherently political nature of PES implementation (Van Hecken et al., 2012; Sommerville et al., 2010; Martinez Tuna and Kosoy, 2007; Corbera et al., 2007; Kosoy et al., 2007) through empirical investigations. Others have developed alternative methodologies for clarifying the bargaining space in negotiating incentives beyond a market-framing (Purushothaman et al., 2013; Vignola et al., 2012). Finally, a number of studies have emerged which take a step away from the design and implementation components of PES and examine the more foundational assumptions underpinning the interaction of incentives and human behavioural change and integrating this insight into potential PES feasibility in a given context (Bowles, 2008; Kerr et al., 2012; Narloch et al., 2012; Yanez-Pagans, 2013).

The body of research presented in this thesis essentially argues that PES is an attractive and potentially successful mechanism for water resources management where the ecosystem services being managed are non-rival but potentially excludable. These include: a) groundwater *quality*, where aquifers might exist on private land, but whose benefits are not diminished by an individual's benefit of the service; b) surface water quality improvement, and c) flood protection. However, PES as defined as a tool to correct for 'market failures' is fundamentally flawed to the extent that: a) ecosystem services are treated as marketable commodities or standardised units of trade assumed to be rival and excludable; b) individuals only behave according to an assumption of *homo economicus* when it comes to natural resource management; c) the management objective lies in determining the utilitarian values for ecosystem services which serve as a conduit between otherwise separate and distinct 'society' and 'nature', and d) the tool was

designed to fit reality into a construct rather than designing the tool to reflect social reality. To the latter point, it is emphasized here that both PES and IWRM should be envisioned as *practice-derived concepts*, rather than theory-defined imposition.

This thesis contributes to the trend of critical analysis of PES by: a) developing a new governance framework for ecosystem services in a watershed context; b) applying a methodological tool for assessing technical and social complexities for socio-ecological conflicts to a proposed PES scheme for watershed management; c) exposing the trade-offs implicit in targeting payment schemes for diverse objectives, and d) exploring the behavioural economics assumptions of PES in a context at the frontier of the free-market and exchange values for public goods and services. Finally, few studies examining PES feasibility have taken an explicitly *justice*-oriented lens to considering fairness, for ensuring long-term negotiation outcomes. We argue that an emphasis on justice will not only place the often neglected socio-political and power asymmetries that mar IWRM efforts closer to the centre of scrutiny, but will also facilitate greater empowerment and ultimately longer-term success in achieving both ecological and human well-being aspects of water management. In this manner, IWRM and the consideration of PES are both viewed as evolutionary processes of landscape management to reflect an assemblage of ecosystem components and services conforming to the values of diverse social actors over time.

1.2. Research Objectives

1.2.1. General Objectives

1. To critically analyse the theoretical, behavioural, and methodological aspects of Payment for Ecosystem Service (PES) design and implementation using case studies with relevance for water resources management, and

2. To effectively argue, through empirical evidence, that PES, as applied in watershed management, is more appropriately reframed or redefined as an incentive-based negotiation for

collective action rather than as a pure market exchange.

1.2.2. Specific Objectives

1. To specify the institutional parameters through which PES can effectively consider socio-ecological complexity in watershed management;

2. To broadly situate PES feasibility according to a conceptual framework which considers the spatial and economic characteristics of ecosystem services;

3. Construct a targeting matrix to spatially visualise the trade-offs in PES optimisation for prioritising economic efficiency, ecological effectiveness, and poverty alleviation;

4. Reveal the extent to which external efforts to optimise PES confront specific historical and pre-existing political relationships, local institutions and community preferences associated with a design strategy in a given context;

5. To apply a valuation and decision-analysis technique known as social multi-criteria evaluation (SMCE) to make trade-offs explicit in the design of PES schemes;

6. Design incentives in a particular context to reflect the capabilities 'to do and be' as a means of achieving human well-being through management interventions in a particular context;

7. Determine how different behavioural signals which comprise of both social and market norms influence watershed management in a particular context, and

8. Offer recommendations for improving the design of incentives in a specific watershed basin according to the set of incentives which best aligns social norms and local institutions with individual gain.

Chapter 2

GENERAL REVIEW OF LITERATURE

2.1. Foreword

PES schemes have been very popular as a policy tool for dealing with environmental problems, and have been touted as the new wave of implementation tools for environmental management (Muradian et al., 2013). The PES arrangement refers to a contractual transaction between a buyer and a seller for an ecosystem service or more commonly as a land-use practice that is likely to secure that service (UNECE, 2007). In recent years, academics have been raising the call that the market-driven approach which underlies the definition of PES makes gross underestimations of ecosystem service values, simplifications of ecosystem dynamics and puts a blinder to the institutions and cooperative social relationships of exchange and reciprocity in society, thus posing significant barriers to implementation of PES in practice (Martinez-Alier et al., 1998; Kosoy and Corbera, 2010; Muradian et al, 2010, Vatn, 2010; Tacconi et al. 2012). Claims that PES can provide 'double dividends' in terms of improved ecosystem stewardship and poverty alleviation require greater validation through empirical research and the application of specific methodologies which attempt to design PES to meet these objectives (Pascual et al., 2010). While the debate continues over the viability of PES, it is firstly necessary to understand how ecosystem services are conceptualised before evaluating the potential of incentive-based negotiation for maintaining or enhancing their delivery.

2.2. Ecosystem Services

2.2.1. Stocks and Funds

A number of ecosystem processes and services such as nutrient cycling, biodiversity, soil retention, carbon sequestration, pollination, and water purification among others are not only supported by agricultural land but are recognised as 'services' or benefits to human societies in terms of the positive contribution they provide to agricultural productivity (Dale and Polasky,

2007). Thus, the suite of largely invisible or less tangible regulating and supporting ecosystem services not only enhances the ability of the land to offer humans more tangible provisioning services, but the management of provisioning services feeds back to enhance or diminish the capacity of the land to provide regulating and supporting services. On a global scale, the intensification of agricultural production has vastly enhanced the provisioning services of crop production while negatively impacting regulating and supporting services such as nutrient cycling, soil retention, carbon sequestration, and biodiversity (species loss) through increases in chemical fertilisers pesticides, sediments and dissolved salts (Tilman et al. 2002).

The distinction between provisioning services on the one hand and regulating and supporting services on the other is an important one for the development of PES schemes. Farley and Costanza (2010) distinguish between ecosystem *goods* as possessing stock-flow physical properties (i.e. transforming forests into houses or medicinal plants into capsules) or as ecosystem *services* possessing fund-service physical properties (a particular and often complex configuration of ecological structure that generates a particular service). While stock-flow (or what we term 'provisioning services') can be stock-piled, fund-service type ecosystem services (or what we term 'regulating and supporting services') cannot be. The latter cannot be viewed as discrete resources to be exploited, but as complex biological assemblages and require ongoing management for the services they provide. The physical configuration of ecological structure in the 'fund-service' type is not quantitatively altered to become the service, but instead produces *qualitative* changes that are less amenable to measurement or modelling than provisioning resources are. Thus, it is argued that managing the physical characteristics of ecosystems as diverse 'funds' may be a wiser approach to ensuring the flow of services than trying to quantitatively model ecosystem services as if they were all resources to be stock-piled (Farley and Costanza, 2010). Ultimately, provisioning services (stock-flow goods) and regulating/supporting (fund-services) are highly intertwined and will trade-off with each other, as it is the ecological structure which not only provides direct resources for consumption or use (i.e. medicines, timber, fruit) but also interacts through complex and dynamic relationships to serve as a fund for promoting regulating and supporting services at larger scales.

Understanding the relationships between ecosystem services and their distinctions as stocks or funds is crucial for PES practitioners since funds and stocks possess unique and irreconcilable economic characteristics which render specific policy tools such as PES better suited for the management of certain ecosystem services over others. However, thus far, the vast majority of PES initiatives have failed to consider this nuanced understanding of ecosystem services and have tended to conceive of all services as resources to be stock-piled, commodified and exchanged. It is particularly salient to identify how ecosystem services can be assessed as a means of determining how successful society is in managing ecological endowments over time as well as better design policy such as PES to better reflect social and ecological realities.

2.2.2. Ecosystem services as indicators of global funds

Bockstaller et al. (1997) sought to measure ecosystem services ranging from ground water quality, air quality, soil quality, surface water quality, biodiversity and landscape quality using a number of indicators which included nitrogen and phosphorus readings (leached), soil structure, cover and organic matter content which relate to one or a combination of these services. Measuring the services that derive from ecological structure requires close attention to the components and inter-dependencies of ecosystem characteristics. In addition to spatial scale, ecological systems can be characterised according to biological composition across and between communities; populations and species; canopy gap dynamics; degree of connectivity; size or shape in the landscape; substrate including soil conditions, and disturbance regimes. Ecosystem functioning refers to patterns of net primary productivity, decomposition, erosion and nutrient cycling, trophic relationships (e.g. food webs), and succession.

Often, determining patterns of structure and composition of ecological systems is easier than determining functional patterns (which are related to the delivery of regulating and supporting ecosystem services). Thus, indicators of functional patterns are often represented as structural and compositional components including age, species composition and size. For example, remote sensing of these characteristics can be used to estimate carbon storage by the landscape (Dale and Polasky, 2007). Typically, landscape-scale characteristics that can be captured through remote sensing and geographic information systems are less expensive and more accessible than

intensive field-based observations. For example, land-cover changes can be used to predict loss or degradation of habitats of concern. It can also be used to measure erosion which can be determined by the proportion of exposed land, particularly on slopes of greater than 5% (Maloney et al. 2005). The measurement of chemical fertilizer application can also be used as a proxy of a number of ecosystem services (such as soil quality, water quality, and biodiversity). In particular, nitrogen inputs to agricultural soils, ground and surface waters from mineral fertilisers and animal-production systems has a significant impact on the trade-offs of provisioning services to regulating, supporting and cultural services.

Attention to scale is also important to inform indicators that are simple enough to capture the complexity of the system yet effective and simple enough to be monitored and conveyed. Indicators or proxies to measure ecosystem services must also be sensitive to historical or anticipated change in the ecological system, given a range of scenarios of uncertainty. Considerations of land-use practice: crop mix, rotation and tillage strategy, type of farm equipment used, fertiliser and pesticide applications over time, as well as future changes associated with altered rainfall patterns, temperature regimes and species migrations may have important interactions which indicators should be sensitive to in terms of the alignment of change between indicator and broader ecosystem change (de Groot et al. 2002). Lastly, the more closely indicators are aligned to management actions, the easier it will be to link land-use practices with ecological processes that may enhance the delivery of desired ecosystem services. For example, a typical strategy for receiving payment in a PES scheme is based on the maintenance or enhancement of natural vegetation along defined waterway buffers thereby linking land-use management with water quality (and perhaps other services such as biodiversity protection or carbon sequestration). However, in reality, the highly interdependent nature of ecosystem services and uncertainty of dynamic social and ecological drivers makes the identification of such ‘low hanging fruit’ in the identification of robust indicators hard to come by. Given resource constraints to optimize indicators to be as robust as possible will itself involve a process of trade-offs according to the set of ecosystem services considered in a given context. This process itself may underestimate reliability of the indicator.

As mentioned by Duarte et al. (2002), water quality is directly impacted by agricultural activities through loss of soil and by inputs of nutrients from fertilisers and pesticide usage which impacts biotic life, and changes water properties (e.g. turbidity, pH, and temperature). Vulnerability to erosion on exposed sloping land from sedimentation is measured by total suspended sediment concentrations and concentrations of nitrate and phosphate during baseflow and stormflow; coarse woody debris, streambed particle size; dissolved oxygen concentrations, and benthic macro-invertebrate richness (abundance and diversity) (Houser et al. 2006; Maloney et al. 2005). Understanding the ecological relationships associated with water quality improvements, including the aspect of slopes, vegetation type and structure, soil and streambed characteristics, climatic and hydrological patterns will all aid in ensuring PES schemes are effective and ultimately socially acceptable.

It is worthy to mention that indicators or proxies to measure ecosystem service flows should not be measured as though they are independent of each other. In any given context or location, the degree to which indicators may be complementary or integrative requires critical consideration. The suite of indicators chosen should examine various land-use management regimes for agricultural land to determine which regime or scenario provides maximal ecosystem service flows with minimal trade-offs. Value should be measured by heterogeneity of ecosystem service flows rather than which management regimes commands the highest price according to a singling-out approach of ecosystem services (typically provisioning services). For example, buffer strips along waterways provide a bundle of ecosystem services ranging from soil retention, nurseries of biodiversity, habitat protection, aesthetic value, carbon sequestration value, and improvement in water quality. We must also consider maximising ecosystem service flows from a broader mosaic perspective in which natural habitat complements and supports agricultural land over time and across the landscape (Santelmann et al. 2004; Boody et al. 2005). When measuring ecosystem service flows, the issue of scale (geographic and temporal) is of utmost importance. Landscape-scale evaluation of changes can be useful to understand overall impacts, while farm or site-specific measures are useful for understanding how specific land management practices may affect specific service flows. Understanding how ecosystem service measurements relate to each other at different resolutions of scale remains a challenge (Dale and Polasky, 2007).

Finally, despite our good efforts to measure ecological functions in meaningful ways for human management systems and societal institutions and policies such as PES, the field of ecological science must not redress into complacency within the arena of ecosystem services. The metaphor of natural capital as a stock of ecosystem goods providing a flow or fund of less tangible ecosystem services is a useful management framework, but may be 'blinding' us to the realities of ecological relationships and trade-offs that exist across spatial and temporal scales (Norgaard, 2010). As climate change and socio-economic and population pressures continue to unfold, it will be increasingly more difficult and indeed unfair to expect ecologists to confidently predict how ecological processes will be affected, let alone expect ecologists to 'black box' such processes into packets of services for humanity. This realisation has serious implications for PES development and emphasizes the urgent need to consider adaptive management approaches which respond appropriately to social and ecological learning.

2.2.3. Ecosystem Services and human well-being

The impact that watershed management has on well-being is measured in terms of changes to indicators attributed to multiple dimensions of experiences representing human well-being (Figure 2.1). A household can be identified as 'poor' if it is either deprived to a certain threshold of these functionings (weak sustainable well-being perspective) or deprived in even one dimension (strong sustainable well-being perspective) (Reddy et al., 2004). In the former classification, improvement or performance in some functionings can at least compensate for declines in others; whereas this compensation is not permissible in the latter perspective which places a requisite role for land and water resources as forming the basis of livelihoods of people within a watershed basin (Rennie and Singh, 1996). A holistic wealth indicator developed by Arrow and Dasgupta (2004) illustrated that wealth (measured as inputs of utility value, health and education, freedoms and choices) has reduced in most countries and is linked to the decline of natural capital.

Understanding how the maintenance of ecosystem services directly results in well-being improvements requires acknowledgement of the significance of value plurality within human society. If we assume that all the things we have reason to value can be measured through one

homogenous dimension, such as utility, then we must also assume that certain criteria of value, such as the opportunity to choose whichever value system is socially applicable, can be compensated for by maximising the utility score. However, our value systems are diverse and irreducible to each other (Sen, 2010). For example, some cultures may possess stronger values for cultural and spiritual values associated with trees in a sacred grove than they do for provisioning values of forests like timber (Duraiappah, 2011). The non-commensurability of bringing the spiritual value of sacred groves and the provisioning value of timber into a single monetary unit is clearly not without difficulty or perhaps is even in the realm of impossibility.

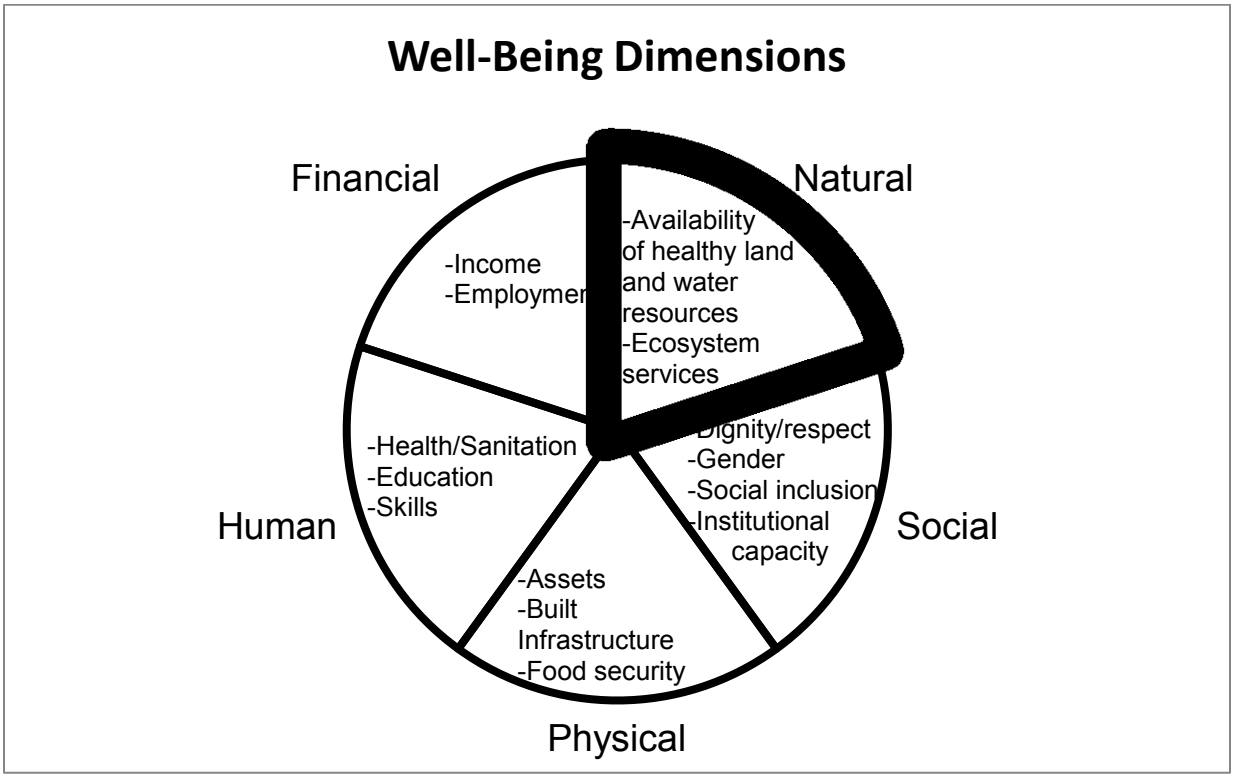

Figure 2.1: The above illustrates a multidimensional perspective on poverty classification through five dimensions of well-being, each with a tentative set of measurable indicators. The natural dimension is highlighted to reiterate the foundation of human well-being on natural resources comprising the substrate for physical developments, influencing social and financial arrangements, and provisioning requirements for nutrition and health (adapted from Reddy et al., 2004; Alkine and Santos, 2010).

2.2.4. PES and well-being

Within this dilemma, the trade-off between achieving the ecological objectives and improving the well-being of the poorest becomes increasingly evident. The role of PES for watershed services in improving social welfare is tempered by recent observations indicating that benefits from PES are substantially lower if measured in terms of local benefits (watershed protection) than they are for global benefits (carbon sequestration or foreign tourists paying for biodiversity conservation). Hence, capturing the demand from often poor local beneficiaries of water related services to pay for improved well-being upstream will require innovative combinations of public-private arrangements in order to achieve objectives (Rodriguez et al., 2011). The small and fragmented nature of land parcels managed by the poor is also a concern for realising benefits through PES. In addition to higher costs of transaction in targeting piecemeal plots of land into individual PES negotiations, the biogeography of targeting discontinuous plots for more sustainable land-management is less likely to be adequate for improving the ecological health of watersheds (Wunder, 2007). Indeed, where the number of poor people living in the upstream regions of a particular catchment is high, the value of ecosystem service incentive payments will tend to be lower (Kosoy and Corbera, 2010). This is likely as a result of higher costs of transaction and greater insecurity of ecosystem service delivery associated with more numerous influences on land that require consistent and sustainable management in order to have an impact on a particular ecosystem service, such as water quality. The trade-off can be de-linked if we can overcome our preoccupation with ensuring an efficient PES transaction, which is more the exception than the rule in terms of empirical implementation for watershed-service payments (Ghazoul et al., 2011). The conditionality criteria for monitoring identifiable improvements in ecosystem service delivery is rarely adhered to and is often achieved only through costly technical studies linking land-management activities, land characteristics and impacts on a given ecological service. Dynamic land management regimes by diverse human communities, and a suite of spatially and temporally diverse biophysical relationships do not provide a ‘one-size fits all’ relationship between a given land activity and its impact on say, water quality (Wunder, 2007).

As mentioned, benefits are often socially constructed during periodic re-negotiations and are thus context-specific to any socio-ecological system (Fisher et al., 2010). A question then arises. If payments can then be designed to reflect socially acceptable conditions even if the value (however defined) of providing that service does not match its cost of provisioning, is it not a beneficial way to promote marginally sustainable watershed stewardship if the arrangement is perceived as positive by upstream and downstream stakeholders alike? As Muradian et al. (2010) have argued, it is the stakeholders' perceptions of costs and benefits that matter more than documenting 'fictitious' efficiency. Indeed, given the conceptual difficulty, uncertainty, lack of experience, and high transaction costs associated with defining 'market prices' for service provision, it seems logical to proceed with ecosystem service management through awareness-raising and negotiation (Wunscher et al., 2008). It is at this juncture that PES can be designed both to marginally improve benefits for downstream beneficiaries on an ongoing basis while focusing payments on alleviating the deprivations of well-being affecting upstream populations and equally engaging in a committed participatory approach for watershed management.

Several PES schemes have sought to target more holistic improvements to well-being and provide examples extending beyond simply monetary wealth criteria for poverty alleviation. A recently implemented PES scheme in the Dong Nai River Basin in Vietnam has targeted involvement of women-headed subsistence households as service providers through the development of new skills and off-farm employment opportunities for sustainable bamboo and cacao products, handicrafts and essential oils. In addition to payments for forest protection and management, these largely forest-dependent farmers were provided market knowledge, production training and advice to develop business partnerships with new markets located in downstream Ho Chi Minh City (Nguyen et al, 2011, Kolinjivadi and Sunderland, 2012). In other cases, diverse locally-empowered institutional arrangements develop increased internal motivation for continuity of payments and sustainability of land-use. Clements et al. (2010) found that greater long-term well-being enhancement to a larger number of stakeholders was more promising for the village-managed payments than direct payments to specific service providers. As a result of local development and approval of the terms of trade, village representatives were able to clarify property rights and provide wider community-level benefits

ranging from new schools and roads and maintenance of existing water extraction infrastructure (Clements et al, 2010).

2.3. Integrated and Adaptive Water Resource Management

2.3.1. From end-of-pipe solutions to governance priorities

Global water challenges continue to expand across the world as increasing numbers of people continue to struggle with access to safe water and sanitation, as direct and indirect drivers associated with population growth, expansion of economic activities, and climate change continue to influence effective and equitable water management. The inertia of both the physical infrastructure (lock-in) and growth-driven governance systems add a further obstacle to fostering systemic change (Unruh, 2002; Grafton and Hussey, 2011). Governments and local communities around the world are seeking interventions that will ensure healthy watershed ecosystems, reliable and high quality delivery of watershed goods and services, and mitigation strategies for impending climate change (Ahmad, 2003; Rahaman and Varis, 2005). In order to address these dynamic challenges under a context of uncertainty and ever-increasing urgency to be proactive in effectuating change not just in the water management sector but for natural resource management as a whole, it will be necessary to fully embrace the connectivity and complexity of socio-ecological systems by incorporating diverse knowledge systems and collective negotiation techniques in order to move towards a common understanding or new social norms for human development (Brondizio et al. 2009). Indeed, as Pahl-Wostl et al. (2011) suggests, an increased debate has emerged in the field of water management urging a paradigm shift in transitioning from purely technical end-of pipe solutions for well-defined water problems (primarily point-source style problems) towards understanding social relations, norms and organisation for dealing with multi-scalar and highly dynamic and complex water problems (largely governance related).

It has been argued that the root causes of natural resource management crises are less influenced by the realities of biophysical thresholds, knowledge systems and technology, but are more closely reflected in poor governance systems (Ostrom, 2007; Pahl-Wostl et al., 2011).

Recognition of the dynamic nature of water management challenges and the multiple demands for conflicting use of water resources led to the evolution of the concept of Integrated Water Resource Management at the UNESCO International Conference on Water (1977). Since this time, IWRM has been the priority of political agendas in the field of water resources (Rahaman and Varis, 2005). The objectives commonly stated for IWRM include 'optimising' and 'sustainably' using water resources for socio-economic development while protecting and enhancing the endowment of the ecological resources which provide water resource flows (Grafton and Hussey, 2011). It is centred on the co-ordinated management of land and water resources in balancing both resource protection while meeting both social and ecological objectives. IWRM is characterised by incorporating the following elements (IWA/UNEP, 2002):

a) fully integrate management at the nexus of land and water systems;
b) adopt a systems approach to problem structuring and intervention planning;
c) involve fully inclusive participation to the maximum extent possible;
d) provide an enabling environment on the part of local government systems and supported by central governments;
e) make best use of existing technologies and knowledge systems;
f) involve capacity-building, and
g) involve innovative policy interventions based on both regulation, incentives and moral obligations according to locally understood norms and culture

These elements recognise the emphasis required on social robustness for management of common pool resources, such as water. Designing highly participatory and integrated interventions that best reflect these characteristics can make a positive contribution to the achievement of IWRM. However, taking a systems perspective across the landscape which embraces the complexity and uncertainty of interactions across geographic and temporal horizons, diverse knowledge systems, and multiple values through deliberative participatory approaches remains missing within an IWRM framework. It is worth mentioning that there are tangible management benefits of bridging the conceptual link between watershed goods and services and overcoming deprivations to critical human assets or fundamental human needs (Duraiappah, 2011). Placing a priority on both ecological and socio-economic baselines for

watershed resources (a 'floor of basic human well-being standards and a ceiling of ecological appropriation) is imperative for the improvement of IWRM and is itself a function of fair social deliberation. However, embracing this opportunity does not mean we can be overly confident or complacent in our decision systems.

Different localities and contexts will require diverse configurations of stakeholders and culturally-specific institutions to achieve a sustainable balance of ecological limitations and socio-economic needs of watershed systems. Thus, adapting IWRM theory to local contexts limits the prescriptive potential of the vision in terms of specific techniques or strategies for implementation and has been lambasted by some critics as a limitation of the concept (Biswas, 2004; Jeffrey and Gearey, 2006; Medema et al. 2008). Secondly, overcoming uncertainty in the management of common-pool resources is a naive assumption. Integrative management attempts should not only embrace limited information but accept it as an inherent property of complex socio-ecological systems. Past attempts at water management that have hinged on technocratic solutions to narrowly defined problems will continuously be aiming for a moving target. Indeed, IWRM has been recognised as failing to account for variability and uncertainty of the broader 'problemshed' in which it exists. A combination of ecological and socio-economic uncertainty characterised by altered climatic regimes, population pressures, evolving social needs, regional and global resource pressures, and other stochastic events can never be accurately quantified (Pahl-Wostl et al., 2011). Accordingly, a broad concept known as 'adaptive management' was introduced to emphasize systematic learning over time from the outcomes of implemented interventions for natural resources in order to continuously improve or at least fine-tune management practices to better reflect socio-economic realities (Pahl-Wostl et al., 2007).

2.3.2. Adaptive management

Intrinsic to the adaptive management (AM) process is the importance of differing perspectives or framings of the problem. This inclusionary element is similar in some respects to the participatory focus of IWRM. Confronting realities of the complexity of natural resource management requires complex adaptive systems that use mixed qualitative (deliberative participatory techniques) and quantitative methods to assess the future success of policy

interventions in meeting stated objectives and/or to model potential interactions (synergies and trade-offs) of interventions across ecological and socio-economic systems at multiple scales (Pahl-Wostl, 2007; Engle et al., 2011). Moreover, innovative partnerships of social actors that involve a mix of decentralised and enabling institutions at higher levels form the governance basis for effectuating adaptive management. These adaptive *co-management* strategies have been the cutting edge of institutional theory and greater research is needed to understand what configuration of partnerships best reflects the governance needs of water resources in particular contexts (Olsson et al. 2004; Brondizio et al. 2009; Ostrom, 2005). Unlike IWRM, an adaptive management strategy has no 'achievement' phase. It is a continuous and iterative process as new knowledge feeds back into re-defining the problem, improving the understanding of the socio-ecological system and better refining management interventions in light of this knowledge. In essence, this 'preparedness' for external shocks combined with common understandings of the demands and constraints on the resource base through participatory processes can help achieve continuous and long-term socio-ecological resilience to stochastic events (Folke et al. 2005). Specific techniques that have been identified that emphasize social learning and adaptive principles for water (or other natural resource) management strategies which embrace both technical and socially conceived perceptions of the system include participatory modelling and social multi-criteria evaluation (Munda, 2004; Halbe et al., 2013). As Pahl-Wostl et al. (2011) has suggested, generating processes of social learning and adaptive management strategies for water resource management requires capturing the interdependence between the processing of technical information about a problem and engaging in processes of social exchange or deliberation.

2.3.3. Striving for a paradigm shift

Constructing an integrated and adaptive management framework such as IAWRM requires a profound shift in the structural paradigm which currently encompasses water resource management. Watershed management cannot be considered as an optimisation strategy for addressing multiple objectives efficiently (for which IWRM has been criticised for). At the same time, adaptive management strategies cannot forever be adjusting to a set of socio-economic institutions which are beset by structural power dynamics and therefore do not result in

improvements over time. The approach we take towards appropriate management strategies will directly influence the process and outcomes of decisions and subsequent consequences for socio-economic and ecological sustainability. It is therefore crucial that we approach management of natural resources with foresight, precaution, and recognition of the *social* fabric (i.e. socially-constructed narratives) which underpin our perspectives towards meeting individual and collective needs, developing common understandings or norms, and maintaining a healthy and resilient resource base.

More critical research on the assumed structural tenets of globalised society needs to be confronted with regards to the realities of biophysical thresholds and fundamental assets for human well-being (Rockström et al. 2009; Kallis, 2011). Indeed, theoretical development has greatly outpaced implementation and is additionally inflated in relation to capacities (integrated knowledge systems and shared understandings or norms of behaviour) that are in fact required for integrated and adaptive systems (Pahl-Wostl et al., 2011). Accordingly, structural change cannot be achieved unless social learning processes are sufficiently empowered to question underlying narratives in order to re-structure institutional arrangements to enhance individual and collectively perceived capabilities (Pahl-Wostl, 2009). The realisation of integrated and adaptive approaches for water resource management (and natural resource management in general that follow an IAWRM vision) requires a paradigm shift that places much higher emphasis on continuous reflection, diverse capacities, knowledge and social institutions than on mechanistic or technical strategies that assume existing institutions as static or adequate for the sustainable futures that are desired.

2.4. The extent to which IWRM[1] can be achieved through PES

2.4.1. IWRM and social welfare

The Global Water Partnership places emphasis on equitable social welfare as a core objective of integrated water resource management. However, various definitions of IWRM have been utilised in government documents and even within the research literature suggesting different

[1] Henceforth, when IWRM is mentioned, "integrated" and "adaptive" water resource management is referred to.

perspectives by which watershed management can aid in improving well-being. The Bonn International Conference on Freshwater in 2001 specified a number of basic principles for guiding water policy and management practices within the broad IWRM approach in addressing poverty. It was agreed that providing access to water to the poorest was the primary objective of any poverty reduction measure in watershed management since water security forms a basic human need for health, production, food security and general reduction in vulnerability (Ahmad, 2003). Secondly, it was recognised that sustained productive uses of water requires high integrity watershed ecosystems, minimum flow of water and application of locally-specific conservation practices to ensure healthy ecosystems. Thirdly, optimum interventions would need to be designed to reflect local-level realities, be built upon transparent regulatory and financial arrangements, and be ideally aligned to the water needs of the poorest communities through close consultation (Ahmad, 2003). These recommendations are consistent with a negotiated incentive mechanism (such as PES) designed according to prevailing circumstances and local needs. Such policy tools can be geared towards poverty alleviation, especially if management activities reflect the needs of the most marginalised groups. The 'integrated' aspect of IWRM is perhaps the framework's greatest strength and its biggest curse. It is important to consider the appropriate mix of integration sought, whether coordination of watershed objectives across a geographic area, over a desired time period, over a set of specific objectives, or across policy mandates, programmes and governing agencies (Cardwell et al, 2006). Understanding the diverse management objectives and demands on water resources in an otherwise fragmented institutional network is certainly of merit, but unifying institutions over the common cause of water resources management lacks a clear roadmap for spanning the multitude of contexts and governance circumstances that may exist and change over time.

In order to assess the potential of IWRM to improve well-being, it is necessary to pay close attention to what its definition implies in practice. 'Integrative' refers to bringing together multiple parts or enabling unity through horizontal and vertical processes of integration. 'Water' of course is the essential solvent for life and makes up seas, lakes and rivers. 'Resources' refer to a substance or material that provides benefit or support for human endeavours or states of being. Finally 'management' is the process of controlling or handling something so as to achieve an objective or goal (Cardwell et al, 2006). Thus from its semantic derivation, IWRM is the

unification of activities to control water resources in order to obtain a certain objective (ibid.) Various international agencies have adopted their own versions of this basic conceptualisation while considering a well-being perspective to various degrees (World Bank, 2003). However, few definitions place improving human well-being as a primary objective and most fail to extend beyond the watershed boundary in considering root causes of water resource mismanagement (Cardwell et al, 2006).

From the perspective of the poor, well-being improvement requires a set of ecosystem services associated with a healthy functioning ecosystem and not just the provision of high quality water inputs and predictable flows. For instance, rural communities depend critically on forest resources, local biodiversity which can be enhanced more broadly through natural resource management (Moench et al, 2003). Moreover, limiting water management to the watershed basin is not always the most appropriate unit for addressing localised problems that are influenced by broader political issues such as land tenure or the perspective of rural communities in relation to their neighbours. Thus, IWRM as it stands does not target well-being improvement directly despite the clear linkage between poverty and water security. Shifting the objective of well-being enhancement as a core target of IWRM requires a greater emphasis on both the productive use value of water in addition to prioritizing ecological functioning. Empowering communities to decide the fate of their own water and land resources for maximising community well-being will require institutional support to both invest in resource management and also to ensure that negative externalities (particularly to downstream interests) are minimised or managed. Such support could include improving land access and ownership arrangements, supporting allocations and enforcement procedures developed through collective action by local organisations, acknowledging local cultural principles and traditional knowledge, market development and information supply for sustainable low-cost water technologies (Wester et al., 2003; Kiersch et al., 2005).

As Parkes et al (2010) claim, a well-being framework for IWRM requires an institutional support system that considers how watershed management impacts each dimension of well-being, ranging from health and social equity to quality of life while minimizing the trade-offs. For example, Reddy et al (2004) found that landless individuals benefitted more in terms of financial

capital and employment than medium or richer households in proportion to other forms of capital (social, human, or physical). However, in most developing countries, finding the appropriate mix of institutional strategies is a struggle given competing resource demands, high levels of corruption, and significant inequities of interests. It may also be naïve to believe that integrated water governance can proceed without any trade-offs. Thus, it is recognised that while integrated water resource management does have the potential to improve well-being by identifying resource options through collaborative decision-making, the framework intrinsically limits the focus to water resources instead of wider and highly linked natural resource governance (Campbell et al., 2001). It also appears to confront the same barrier of poor governance at higher levels of government where IWRM is typically touted.

2.4.2. Overcoming the implementation gap of IWRM

As Merrey et al. (2005) argue, the most feasible role of IWRM for enhancing well-being is best left to providing the discussion space for stakeholder interests to be represented in a transparent manner so that new action strategies and management scenarios can be developed. A discussion space of this nature could be fostered through existing local institutions and could be encouraged as a platform for negotiating agreements between upstream or downstream interests for the sake of attaining benefits and minimizing trade-offs. PES negotiation can serve as the basis for such a forum in arranging the incentive structure for socially responsible watershed management upstream and downstream of a particular transaction. Further, given the long-term uncertainties of sustainable water management, at minimum a PES agreement can aid in fostering stronger social relations between downstream and upstream entities and serve as a basis for conflict resolution.

As an incentive-based tool, PES provides compensation of some kind to upstream land users by tapping into resources deriving from downstream beneficiaries. In order to evaluate the extent to which PES can improve well-being in a watershed context, it is necessary to consider how well designed the transaction is for actually sustaining and enhancing watershed services as well as considerations for subsequent negotiation for maintaining land-use stewardship beyond the initial payment period. The restoration of ecosystem functionality in the watershed will often exceed

that of a payment negotiation period resulting in temporal lags for positive impacts on water resources. Moreover, even when cause and effect relationships between land-use management and ecosystem service improvement are clear, ecological functioning requires specific yet highly uncertain biophysical thresholds in order for tangible water quality or quantity improvements to be achieved (Ferraro, 2002). Ultimately, PES application, much like recommendations offered for IWRM, will be well-being enhancing to the extent that local stakeholders are empowered to tailor the transaction to raise levels of identified categories of well-being. Notions of conditionality and additionality which form the backbone of PES success may need to be re-considered as 'works in progress' rather than specific deliverables required by a set date. PES for well-being improvement would seek to reward households and communities that establish or maintain sustainable land-management but not necessarily those who have proven that service delivery has improved as a result of the payment. Furthermore, PES should seek to establish the interest, eligibility, opportunity and training for the poorest and/or most marginalised households to achieve sustained and equitable improvements in well-being by engaging in wise land-use management (Pagiola et al., 2005). As specified in Leimona and de Groot (2010), this would mean entrusting local resource management plans, rewarding community-scale management actions, and supporting payments which benefit multiple stakeholders rather than just to individuals.

Well-being enhancement within IWRM can also be conceived as the empowerment of justice-oriented institutions to govern the fair distribution of natural resources. It is important to consider that equitable access to resources is purely a means to achieve an end-state of well-being. Moreover, the ability for individuals to achieve well-being enhancement from a given distribution of resources can vary from heterogeneities in skill sets to variations in physical environment and social climate (Sen, 2010). Thus, the capacity to utilise resources and have opportunities for making decisions on resource use provide a critical step for maximising well-being and will be discussed further in the next section.

2.5. Environmental and social justice through the Capabilities Approach

2.5.1 Conceptualising equity

Some academics have considered equity for ecosystem services from an efficiency perspective in terms of income or assets, in which allocations to lower income individuals provides more 'equity bang' for the buck (Gauvin et al., 2011; Alix-Garcia et al., 2008). From a well-being perspective, others have envisioned equity as the allocation of resources under a veil of ignorance (i.e. Rawlsian) or in terms of a cost-benefit analysis of various dimensions of well-being with externally imposed judgements on how different dimensions can trade-off with each other (Sen, 2010; Alkire and Santos, 2010). In the case of the efficiency-oriented and Rawlsian views of equity, the emphasis lies in the potential equity that could be achieved with a fair distribution of resources. However, this view fails to distinguish between equity as the core objective to be sought by society versus the institutional mechanisms and means by which equity could be achieved (Sen, 2010). Individuals may have a fair distribution of resources, but their ability to extract benefit from these resources could differ according to both endogenous and exogenous factors which require consideration if justice is to be attained. The third view assumes that components of well-being are objective 'truths' and can be identified and generalised to communities across the world, rather than being a conception of social processes, individual perceptions and realities. While to some degree components of well-being are universal for humanity, the interpretation of these components requires a contextually-specific socialisation process, rather than generalising 'truths' or worse still, weighting them according to externally-imposed definitions of 'importance' (Sen, 2010). The capabilities approach explicitly recognises multiple interpretations of value stemming from individuals of different background, circumstance, opportunities and constraints. The approach stresses that the entitlements to 'do and be' take priority in terms of fair and equitable distribution rather than emphasis on resource distribution and institutional arrangements (Robeyns, 2006).

2.5.2. Linking capabilities with ecosystem services

As Norgaard (2010) argues, greater understanding is required which expounds on the social relationships of nature in achieving desired multi-dimensional objectives for human benefit. Thus, there exists a conceptually attractive relationship between ecosystem services and the capabilities approach. Capabilities themselves refer to the freedom to live the life that people have reason to value and which result in actual achievements or enhanced functionings (Nussbaum, 2003). Examples of capabilities include the freedom or opportunity to achieve specific functionings, such as to be educated, to have self-respect, to satisfy cravings, to be socially accepted, or to be well-nourished (Clark, 2009). While Sen (2010) conceptualised capabilities as being primarily individualistic, it has also been conceived as the freedom to achieve non-self goals or communal responsibilities (Robeyns, 2006).

People have different *capacities* to convert ecosystem service benefits into components of well-being. These capacities relate to the hard constraints of institutions (e.g. geo-political borders, jurisdictions) and softer constraints of pre-existing distributional equity among individuals, norms of behaviour, and relations of trust between people. If we consider the degree of distributional equity in a given context, some individuals derive surplus from ecosystems (monetary rewards), while others depend on ecosystems directly for their daily survival. As elaborated by Ballet et al. (2013), the use of natural resources by individuals for specific purposes is related to the set of opportunities and constraints that are available to them. This 'entitlement space' or agency is more representative of behaviour towards natural resources than any specific attitude towards sustainable practices. Constraints and opportunities are both resource or physical-based as well as built along ethical or moral assertions of what are correct or incorrect human behaviours in a given social context.

As Scerri (2012) argues, it is the human capacity to subjectively reflect and collectively or individually conceptualise, formulate and embody these reflections through a set of societal structures, institutions and norms. By taking a reverse reflection approach which is focused on ideas of equity and justice for current and future generations, human agency can debate, plan and re-structure society as affording justice to current and future generations by living within the

capacity of the biosphere to sustain the human species (Scerri, 2012). Indeed, the progress of human development over the course of the centuries was built less on objective 'truths' than on both normative constructions of common understandings and the agency or freedom to act or function in ways that are conducive for development in a given context (Sen, 1999). If society can once again embrace uncertainty and the realisation of normative interpretations of the resource base to which we are endowed with as well as the equitable capacity to achieve our needs and wants, we can re-formulate a new common understanding for sustainable resource management. This process will require a much greater emphasis on cultural and societal foundations for understanding existing and more ideal future institutions which govern the effective allocative and distributive requirements of material resources from the ecosphere.

The capabilities approach is therefore pivotal in linking the fundamental drivers or pressures for human well-being as the context for transformations of the biosphere through political and economic institutions. The influence of ecosystem services on capabilities can be direct in the form of provisioning goods and services, in which access to water, fuel, fibre, food and medicine can directly achieve certain functionings related to being nourished, healthy, or warm (Duraiappah, 2004). However, more indirect benefits from ecosystem services can result in improved capabilities through a combination of personal will or skills, social context, and environmental or physical conditions. In this manner, it is important to differentiate ecosystem services from ecosystem benefits; the latter of which results after the siphoning of ecosystem conversion to reflect personal, social and physical barriers to result in improved capability sets (Polishchuk and Rauschmayer, 2012). It is the human management capacity to ensure that physical or environmental conditions either improve or are maintained over time in order to effectuate a closer linkage between the service and the benefits for human well-being in terms of enhanced opportunities to 'do or be'. Determining the configuration of capabilities that could be improved from ecosystem services requires that consideration be given to both the multi-dimensional characterisation of capabilities and the multi-dimensional perspectives of ecosystem services (especially cultural services) and will thus be a context-specific process.

2.5.3. PES: Incentives for Capabilities

The capabilities approach can be used to help address ecosystem service management for well-being by incorporating a pluralistic view of human wants in contrast to a purely utilitarian perspective of value. In this respect, the capabilities approach addresses equity considerations more directly. From the perspective of payments for ecosystem services (PES), the linkage of capabilities that derive from the potential benefits of ecosystem services can result in enhancements in well-being that are connected to socially-negotiated institutional arrangements and management practices. Thus, it becomes possible to conceptualise 'payments' as the contextually appropriate design of social and individual conversion factors which can overcome deprivations and increase sets of capabilities for service providers over time as a result of sustainable land-use stewardship. These land-use practices can then result in stewardship of ecological endowments so that capabilities can be enhanced for social actors at broader spatial and temporal scales (service beneficiaries).

Finally, the multi-dimensional consideration of values inherent in the capabilities approach corresponds well to the highly deliberative and inclusionary participation of social actors that is required to successfully design and implement a PES scheme in an integrative and adaptive manner. Consideration of capabilities suggests the incorporation for the diversity of perspectives, histories, social positions and resulting opportunities and constraints. Hence, designing payments for achieving increased capability sets requires an evaluative tool that can best satisfy policy strategies according to the set of contextually specific conversion factors that link management of ecosystem services to benefits in terms of increased capabilities. The combination of social inclusion and explicit evaluation of policy strategies against criteria that can increase capabilities lends well to the use of transparent decision-making methodologies such as social multi-criteria evaluation (SMCE).

2.6. A methodological approach for designing PES for IWRM

2.6.1. Introducing Social Multi-Criteria Evaluation (SMCE)

Any decision-analysis tool for complex environmental problems must attend to institutions and social actors with conflicting objectives and unequal power distributions; these considerations should be assumed to exist especially at multi-scale levels of governance for socio-environmental problems. Accordingly, certain tools that prioritise one value system above any others ought to be sidelined for serious inclusion in understanding complex environmental problems. These tools include cost-benefit analysis (or benefit-cost analysis) which seeks to compare the positive and negative aspects of a decision context (be it a project, policy or social decision) through the lens of utilitarian value which effectively view socio-ecological problems under an economic lens, oftentimes reducing other considerations to forms more amenable to commodification (Munda, 2004). A plurality of value systems, participatory mechanisms, and local definitions of problems by local social actors bounded by regional (or national) and global priorities are all pre-requisites for any decision-analysis or valuation technique geared to addressing natural resource management problems.

As such, techniques which focus on conflict analysis, broader systems thinking, and systematic and transparent evaluation of decision alternatives are both important and required to enhance the likelihood of more legitimate and effective decisions for typically uncertain and complex environmental problems. As mentioned by Ostrom (1990), trust, cooperation and shared understandings are necessary to strengthen institutions for multi-stakeholder and multi-scalar decision contexts. This is particularly relevant for management decisions associated with common pool resources where the public goods nature of the resource requires management that is cooperative, reciprocal and adaptive (Fisher et al. 2010).

Social multi-criteria evaluation (SMCE) is a valuation technique that considers diverse and often conflicting values present in society (i.e. social incommensurability) while “orchestrating” multiple scenarios of uncertainty from inter-disciplinary science (i.e. technical incommensurability). (Munda, 2004). Unlike traditional multi-criteria evaluation, SMCE

prioritises the deliberative and public process of the valuation and the integration of both technical (science) and citizen or traditional knowledge systems for approaching environmental decision contexts. In this way, SMCE attempts to incorporate socio-economic, ecological, cultural, political, and technical dimensions together, a feature missing in current environmental impact assessments, cost-benefit analysis, risk analysis or life-cycle analysis tools used in decision-contexts ranging from projects, policies and products (Munda, 2004).

2.6.2. SMCE Theory

The theoretical background for SCME as a valuation technique lies firmly within the field of ecological economics. This field adheres to the concept of strong sustainability in the incommensurability of capital transformations, recognising the irreplaceable character of critical natural capital. The elevated position of natural capital reflects incomplete knowledge of ecological systems as the foundation of socio-economic organisation (Neumeyer, 2007). It also follows "...an absence of a common unit of measurement across plural values" (Martinez-Alier et al., 1998).

Complex systems theory and adaptive decision-making form the foundation of SMCE. Socio-ecological decisions are continuously reflexive processes that rely on learning since the social actors are themselves influenced by socio-economic and ecological contexts (and are continuously and consciously influencing these contexts at the same time). Consequently, the generation of information and its interpretation into possible solutions is a cyclical process as new information continuously sheds light on the decision context. As Funtowicz and Ravetz (1994) argue in their description of 'post-normal science' characterising the socio-ecological decision context: "scientists cannot provide any useful input without interacting with the rest of the society, and the rest of society cannot perform any sound decisions without interacting with scientists" (Munda, 2004). Thus, a mutual cooperation-building and evaluation space is needed to bring together technical experts (policy makers and scientists) with local understanding and traditional knowledge systems.

Additionally, the field of political ecology questions power asymmetries in identifying what ideology (or normalisation of ideas) has the power to simplify the complexity of values into one stream, and for what purpose (Martinez-Alier et al., 1998). Political ecologists argue that many complex environmental problems facing humanity are largely institutional, since it is a specific ideology of norms and rules that shape human relationships with nature. To rectify this power imbalance in institutional arrangements, political ecologists call for more deliberative democracy forums which encourage critical thinking and analysis of all institutions, no matter how ingrained, in order generate social learning and understanding and potentially re-build institutions from the recognition of current multi-scale problems, rather than assume current institutions as a given.

2.6.3. Approaching SMCE

SMCE involves a number of stages, beginning with an institutional analysis of secondary materials, collection of primary data from in-depth interviews and focus groups, and proceeds with an evaluation phase in which decision outputs or alternatives are compared with a combination of technical and social criteria reflecting the decision context. Oftentimes, the steps overlap each other and will need to be iterative to continuously improve the evaluation of alternatives in light of social learning. The first stage analysing secondary data seeks to understand the interests and objectives of social actors which are useful to target relevant questions later in the evaluation that can define the zone of bargaining (Vignola et al. 2012). In-depth interviews and focus groups reveal different ways of 'knowing' as well as the needs, expectations and values of social actors. Focus groups in particular are useful opportunities to identify and deliberate upon intermediate viewpoints between actors in order to reach a consensus while encouraging the free expression of opinions in a transparent manner (Munda, 2004).

In the evaluation stage, the set of researchers (not the social actors themselves) take into consideration primary and secondary information obtained in order to develop alternatives and criteria for evaluation. In this process, alternatives are ranked according to criteria and the results cautiously communicated to interested social actors for feedback and possible re-analysis

(Gamboa, 2006). A key component of devising alternatives and criteria for evaluation is attention to social and environmental justice. Justice can be classified in three ways: a) *recognition* of the knowledge systems, norms, plural values, and cultural identities of social actor groups involved in a conflict; b) *procedural*- or the inclusiveness of the process of decision-making; and c) the *distribution* of costs and benefits (McDermott et al., 2013). In this manner, diverse perspectives are brought to bear on complex socio-ecological problems and eventual outcomes garner greater legitimacy, acceptability, and compliance. Figure 2.2 illustrates how social considerations of recognition, procedural, and distributional justice interact with PES design and implementation to influence ecological outcomes.

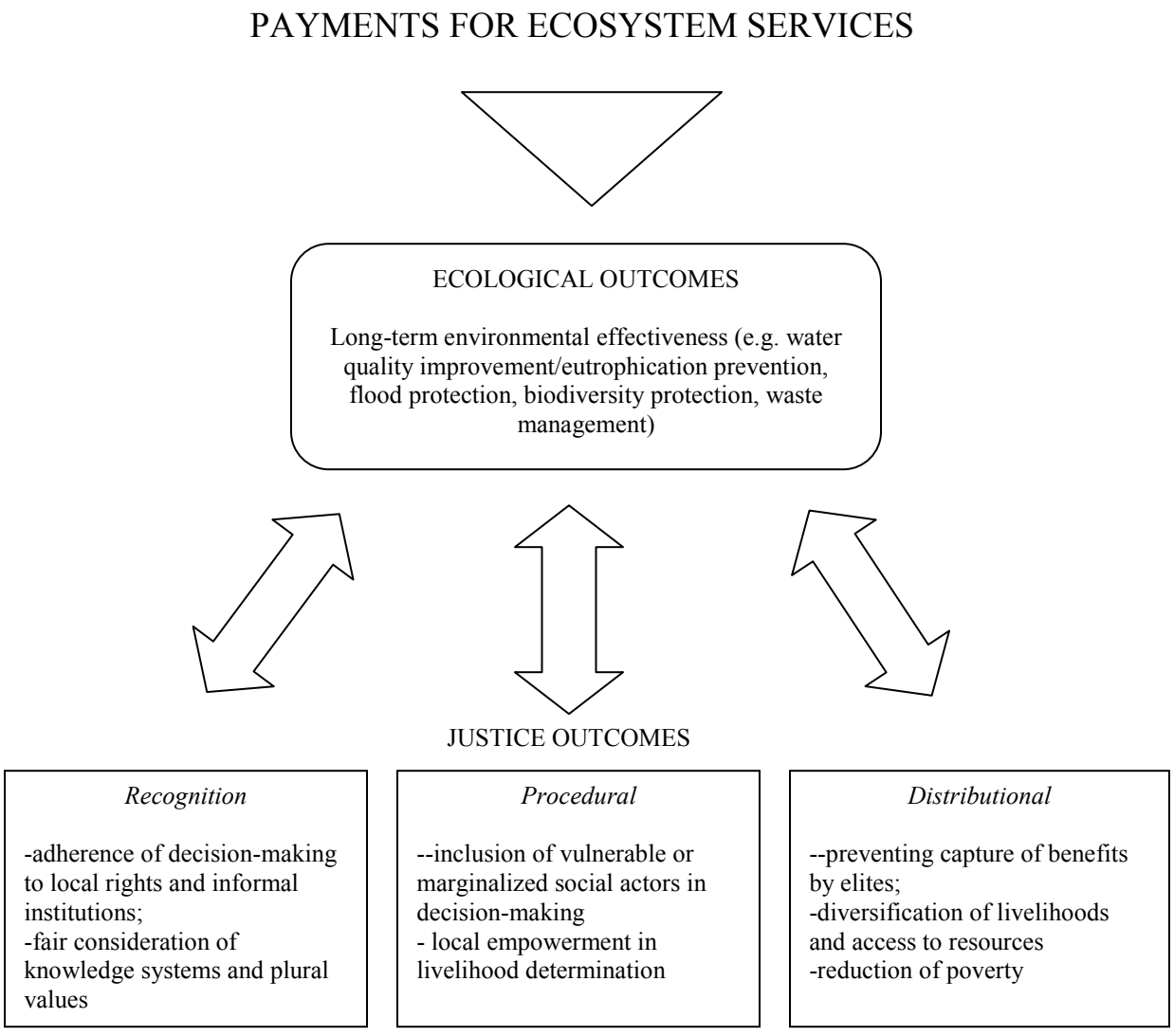

Figure 2.2: PES schemes are fundamentally dependent on meeting recognition, procedural and distributional outcomes of social justice in order for the application of incentives to influence long-term ecological objectives in a watershed basin (adapted from Pascual et al., 2014).

2.6.4. SMCE for designing capabilities-enhancing PES schemes

Evaluation techniques such as SMCE seek to expose information asymmetries, value diversity, and potential conflicts arising from mistrust which are all essential components to overcome in establishing PES schemes (Corbera et al. 2007; Vatn, 2010). To the authors' knowledge, the use of SMCE to evaluate PES has never been done. While a focus on the technical design of enhancing ecological outcomes from PES is an important one, the use of SMCE in designing payments recognises that acceptance of the terms and conditions of PES is a social process that must be attuned to existing capability sets of social actors. Where existing deprivations plague service providers, any land-management strategy which closes off capability sets to overcome these deprivations must be identified and minimised in the management strategy. For example, where upstream service providers are suffering from food insecurity, payment conditions that effectively limit the production of land that provides food for household subsistence will be capability reducing and will likely enhance existing deprivations irrespective of a payment based on overcoming absolute opportunity costs.

Removing deprivations to basic capabilities for well-being, especially for primary resource dependency, is a more stable and effective target for a set of payments or incentives than simply overcoming highly variable and dynamic opportunity costs according to market efficiency. At the same time, capability-based payments must remain flexible to ever-changing needs within a larger socio-economic milieu which continues to influence the inequitable distribution of deprivations across the social landscape. Broader scale population dynamics, shifting market opportunities, and the development or introduction of new technologies and infrastructure are frequently major drivers of land-use change and augment, impede or make obsolete management priorities and collective understandings for common pool resource management (Fisher et al. 2010; Chomitz and Gray, 1996; Geist and Lambin, 2002). Moreover, incentive provision may result in highly variable responses in relation to desired management goals depending on cultural

and social norms in a given community. PES that is blindly implemented without regard to these broader dynamics will very quickly be overwhelmed and subsumed by them (Fisher et al. 2010).

2.7. The behavioural implications of incentive-provision

2.7.1. Underlying assumptions

PES schemes are underpinned by an unwavering belief that incentives are powerful enough motivators to influence land-use behaviour change in favour of ecosystem service maintenance. In order to validate this assumption, it will ultimately be necessary to understand how local culture and existing social norms regarding environmental stewardship aligns with the provision of a range of incentives. Often it is assumed that land-users are intrinsically selfish and will behave in a rational manner with the offering of incentives to act in certain ways. While such a premise has a strong conceptual appeal and has been proven a powerful motivating force for behavioural change, outcomes may be less predictable when diverse value systems exist- particularly for public goods which do not otherwise have a market price associated with them.

Since livelihoods are at stake in experimentation with PES schemes as well as $30 billion of global investment per year, there is much to gain in empirically assessing the extent to which assumptions of behavioural change associated with incentives holds true (Yanez-Pagans et al., 2013). Such information may provide valuable insight into the existing social capital and norms existing in a given community and thus how to tailor PES schemes to existing local understandings. Furthermore, empirically understanding how incentives influence behaviour for the management of natural resources has been poorly studied. The introduction of incentives for natural resource management has been shown to have unintended consequences, some of which increase perceptions of inequity or even lead to environmental blackmail or continued dependence on external funding. A better understanding of the conditions for which incentives can synergise cooperation and reciprocity or conversely be antagonistic towards this behaviour is needed before large scale roll-out of PES-like mechanisms should be conducted.

2.7.2. The interaction of incentives and norms of reciprocity and trust

Reciprocity refers to behaviour of an individual responding to either the friendly or hostile actions of another, while trust emerges from confidence in the ability or expectations of another after repeated interaction (Fehr and Falk, 2002). It has been argued that private incentives disrupt the trust or reciprocity between actors that is necessary for collective action in maintaining public goods (Ostrom, 2005; Gneezy et al., 2011). Likewise, cooperative behaviour is unlikely to emerge in contexts where trust between actors is low (Vollan, 2008). In the presence of selfish individuals, the risk of incentive provision is the encouragement of free-riders to obtain benefit thereby disrupting established relationships of trust (Narloch et al., 2012). Where communal incentives are provided, trust in the form of reciprocity has an even more salient role since an individual will only acquire the agency to strive for group payoffs if she is confident that individual contributions will attain a defined threshold of success or if sufficient trust exists in the community leader(s) who invest or distribute benefits to the community (Fehr and Falk, 2002; Kerr et al., 2012; Narloch et al., 2012). In this respect, payoffs from communal incentives are riskier from an individual's standpoint since the same level of collective action effort by an individual depends on the actions of group peers above a set threshold of success (Travers et al., 2011). Laboratory experiments have identified that voluntary contributions to a collective endowment are reduced when others' perceive an individual's contribution to be insufficient or when monetary fines or explicit control of appropriate contribution is enforced by others. (Fehr and Gächter, 2002; Falk and Kosfeld, 2006, Vollan, 2008). These studies have employed the use of fines or sanctions rather than incentives yet highlight the salience of existing relationships of trust in crafting monetary or nonmonetary incentives and disincentives for collection action (Gneezy et al., 2011).

2.7.3. The crowding out of intrinsic motivation

The endogenous motivations for contributing to the management of collective resources reflects a desire for distributive equality of resources as deeply embedded within the social values, cultural preferences or expectations of particular societies (Henrich et al., 2005). In a similar vein to the erosion of trust and social pressures of guilt and shame through the introduction of

incentives, the genuine care that an individual possesses for other group members is also at risk of being crowded out (Kerr et al., 2012). Furthermore, the size of the incentive matters, the form in which they are given (e.g. in-kind, cash payments or relaxation of regulations and taxes), as well as whether such incentives are distributed to individuals or communities. For instance, too high of an incentive gives a signal of risky or unenjoyable conditions which may deter desirable behaviour while too low of an incentive can be perceived as a token insult to hard work that was better off accomplished without payment (Ariely et al., 2009; Gneezy et al., 2011). Indeed, the provision of any incentive can send a signal that an activity is difficult or unenjoyable and therefore requires stimulation in order to have the task accomplished (Bénabou and Tirole, 2006). Depending on the nature of incentives provided, prosocial behaviour may be harmed if moral values are aggravated by monetary incentives (Frey, 1994; Cardenas and Carpenter, 2008). Where strong social norms and informal institutions for collective action are preexistent, the introduction of communal incentives in which everyone benefits, can produce a synergistic effect in mobilising effort resulting in a crowding-in effect (Vollan, 2008; Travers et al., 2011). Where employees are paid equally according to a desired output of the group, effort-enhancing norms may be triggered through positive reinforcement in parallel with social norms (Fehr and Falk, 2002). A study by Narloch et al. (2012) examined the role of incentives in crowding-in or out agrobiodiversity improvement in two regions of the Andes where PES-like reward systems were being proposed. They found evidence of a crowding-in effect of individual monetary incentives and pre-existing norms of reciprocity over free-rider behaviour. However, they also determined that communal incentives (monetary or non-monetary) can result in crowding-out through a downward spiral of reciprocity associated with the signaling of free-rider behaviour ultimately undermining collective action.

Deci's 1971 seminal study within the field of social psychology highlights how intrinsic motivation can be impaired by extrinsic incentives particularly when incentives become withheld at some point in the future. Where cash incentives were not provided (e.g. the control group), participation in solving a puzzle was constant; this contrasted to the group in which cash payments were provided where participation was significantly reduced after payments were suspended. Here, intrinsic motivation was replaced by expectation of payment resulting in the lack of all motivation once payments were no longer provided. The study's basic design was

replicated nearly 130 times since the original study with a consistent finding that incentives may stimulate motivation only if there is reliability that the incentive will be contingently provided over time (Kamenica, 2012). Given the time lags associated with land-use behaviour and impacts on ecological functioning, this understanding suggests that the interruption of PES incentives could render potential ecosystem service providers less likely to engage in ecological land-use stewardship than they would have been before the PES scheme was implemented. This outcome is more likely where social norms of land-use stewardship existed prior to PES implementation (Gintis et al., 2008). Thus, motivation for different reasons may become constrained to that of rational self-interest with the introduction of individual incentives. From an inter-generational perspective, the implications of this homogenization of motivation will ultimately inflate the costs of maintaining common pool resources as informal institutions break down (Meier, 2007).

2.7.4. Designing PES in the light of behavioural economics

Given the influence of reciprocity, reputation and trust as well as the potential that intrinsic motivations may be crowded-out through the use of incentives, it is not possible to assume that incentives will necessarily influence behaviour in desired ways. This has particularly salient ramifications for the long-term success of PES initiatives. It will become necessary to firstly examine in close detail historical and present social institutions and political relationships between actors and how these have coalesced with water resource management. For example, leveraging upon existing community norms of 'naming-and-shaming' through incentives or demerits (disincentives) can facilitate desired behavioural outcomes that reinforce existing social norms rather than fundamentally altering them. Individuals commonly act in response to social norms and acceptance as well as individual needs and expectations (Vatn, 2009). Accordingly, incentives which resonate with a particular configuration of individual and collective values that is characteristic of a particular community may be more likely to result in expected behavioural outcomes than prioritising individual or collective needs alone.

Connecting Text

The previous chapter has emphasized the inherently political nature of land-use management for the delivery of ecosystem services. The design of PES schemes is thus a normative process. Attempts to arrive at objective PES templates for optimising particular objectives geared towards economic efficiency, well-being prosperity and watershed protection outcomes will inevitably prioritise the interests of certain social actors over others. In order to reveal the social ramifications of PES design and implementation in a watershed setting, it is firstly important to identify the role of PES within the broader toolbox of policies for the management of multiple watershed goods and services. A more nuanced understanding of ecosystem services reflecting their spatial and economic characteristics can aid in differentiating which services are more or less amenable to market trades and which should arguably be managed through collective action or strict policy at higher institutional levels.

In the following chapter, a theoretical framework is introduced which develops institutional governance arrangements that promote an integrated and adaptive water resource management vision. Within this framework, the potential for PES to serve as an implementation tool for IWRM and AM is highlighted as well as the possibility of linking human well-being improvements to payment design. The two general objectives of the thesis specified in *1.2.1* are discussed, while the specific objective 1 and 2 are explicitly answered here. This chapter will set the stage for later empirical research (in Chapters 5-7) which identifies the analytical flaw of conceiving PES schemes as market arrangements rather than as collective action agreements.

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Chapter 3

Recasting Payments for Ecosystem Services (PES) in water resource management: A novel institutional approach

3.1. Abstract

Understanding linkages between human well-being and ecological stewardship at the land-water nexus is needed in order to develop effective, equitable, and resilient institutions to govern watershed resources. In this paper, we argue that payments for ecosystem services (PES) plays a useful role for achieving integrated and adaptive water resource management, but only if attention is drawn to: a) nested governance arrangements which reflect horizontal coordination across space according to the economic characteristics of watershed goods and services as well as hierarchical legitimacy between higher and lower levels of governance; b) 'payments' that are socially negotiated rather than designed according to oversimplified efficiency claims for watershed services and c) 'payments' that are well placed to overcome the individual, social and physical constraints associated with watershed goods and services so that capabilities or the freedom to do and be can be enhanced. This paper illustrates the impossibility of effectuating sheer market-based trades for regulating, cultural and supporting ecosystem services due to their inherent non-rival characteristics. Furthermore, a heuristic approach to characterising watershed goods and services clearly demarcates the extent to which PES can serve as an implementation tool for integrated and adaptive water resources management.

Keywords: PES; nested institutions; integrated water resource management; capabilities approach

3.2. Introduction

The Food and Agricultural Organization (FAO) of the United Nations identified specific policy and institutional support recommendations for visioning a 'new generation of watershed management programmes' (FAO, 2008). These recommendations emphasize the role of

innovative financing, increased social deliberation of policy choices, and highlight socio-economic development within watershed management frameworks. Furthermore, they also recognise the role of incentive-based approaches such as 'payments for ecosystem services' (PES) as playing an integral role in overcoming shortcomings of the integrated water resource management framework (IWRM). The latter has been criticised for its lack of specificity as a set of generic strategies to suit different contexts without identifying means for coordinating watershed actions and enabling appropriate institutions and policies (Biswas, 2004). Incentive-based approaches for enhancing water management efforts have been heralded as a mechanism for promoting ecosystem functioning while also positively impacting well-being of communities in a given watershed. These recommendations resonate with practitioners who are promoting an *integrated and adaptive* approach to water resource management (IAWRM) which take better account of bundles of watershed goods and services and the multiple use and management of these goods and services (Berkes et al., 1998; Folke et al., 2005; Huitema et al., 2009; Pahl-Wostl et al., 2007; Yashiro et al., 2013). This human-centred approach to watershed governance recognises the essential goods and services that watersheds provide to people, such as consumable water for drinking and irrigation, flood protection, ecological resilience, and cultural and/or spiritual values (Postel and Thompson, 2005).

While the conceptualisation of PES was initially predicated as an efficient market-like arrangement (Engel et al., 2008; Wunder, 2005), criticism has focused on the perils of commodifying ecosystem services. These have included the substantial narrowing down of ecosystem complexity to individual and distinguishable 'services' and single value systems which do not recognise ecological, social, or spiritual values as separate from an income dimension (For a review of arguments on the fictitious efficiency of commodifying ecological services as markets see Corbera et al., 2007; Kosoy and Corbera, 2010; McAfee, 2012; Norgaard, 2010). Given that watershed resources are needed for human survival and that watershed governance is dependent on social institutions and evolving knowledge systems, we follow Muradian et al. (2010:1205) definition of PES "a transfer of resources between social actors, which aims to align individual and/or collective land use decisions with the social interest in the management of natural resources." Thus, PES is essentially a socially construed

arrangement that in contrast to a Coasian-based arrangement cannot assume perfect information or negligible transaction costs (Tacconi, 2012).

According to this socially-attuned definition of PES, environmental effectiveness of incentives for managing watersheds should be seen as a continuous effort resulting from fine-tuning land-use behaviour that reflects both emerging scientific knowledge and social perceptions of enhancements in service quantity or quality. PES can be envisioned as a form of social contract of best practice for watershed management that results in flexible socially agreed upon terms for mutual benefit, monitored over time and based on agreed upon management and well-being enhancing activities (Tacconi, 2012). Under this view, PES can have the potential to furnish service providers with whatever 'payment' is socially determined as suitable for overcoming barriers or deprivations to enhance well-being *even if such investments are not necessarily economically efficient.* Such a shift in perspective reflects an adaptive negotiation process as opposed to equating exchange values with predictable improvements in service delivery. Furthermore, such incentives would harness both social perceptions and emerging ecological knowledge on the influence of land-use practices to the delivery of particular ecosystem services. This paper aims at identifying the role of PES as a social tool for advancing the IAWRM vision by proposing a governance framework that pays closer attention to the multi-dimensional characteristics of water resources.

Significant ontological complexity in relation to the scale at which natural resource problems are observed combined with epistemological complexity in terms of dynamic and diverse forms of scientific and traditional knowledge necessitates careful deliberation in order to legitimise socio-ecological decision-making (Functowicz and Ravetz, 1994). This socialisation process within a PES scheme cannot be understated, as economic incentives are unlikely to change personal attitudes unless social norms are already in place to lubricate cooperation and negotiation for resource stewardship (Pretty, 2003). As Vatn (2010) argues, the application of PES will be effective and sustainable to the extent to which PES strengthens the will to solve environmental problems by acting cooperatively, rather than retaining the focus on individual utility as the primary motivation for cooperation.

The participatory approach taken towards watershed management has recognised that socioeconomic implications of well-being are at least as complex as managing the biophysical environment, largely as a result of interdependencies and arbitrary separation of the human and the biophysical system (Postel and Thompson, 2005). This assertion is based on the understanding that individuals and communities are more likely to manage ecosystems sustainably when the values, and not only prices, for ecosystem services are acknowledged and understood. A well-being perspective is therefore important in any natural resource management framework to understand how stewardship of land or water resources can overcome deprivations that households and communities experience on a daily basis (Alkire and Santos, 2010). Such a perspective provides a basis for understanding the multi-dimensional aspects of poverty and the impact external interventions such as policy tools can have for improving livelihoods. This perspective can be viewed in terms of specific assets or functionings that communities and individuals require in order to enhance their *capabilities* or ability to do and be in ways that affect well-being (Nussbaum, 2003). While poverty alleviation was not originally envisioned as the principle aim of a PES scheme, the impact that incentives can foster through collective deliberation for well-being improvements should not be dismissed.

Despite increasing critical analysis of PES theory, few studies have sought to examine how and where its application is best fit for purpose in maintaining or improving ecosystem service delivery. With the exception of recent studies linking incentive-based negotiation with common pool resource management, few studies have identified the institutional mismatches associated with applying PES as a market-based or state-driven intervention (Escobar et al., 2013; Fisher, 2013; Fisher et al., 2010). A nuanced understanding of economic, spatial, and well-being characteristics associated with ecosystem services is required to most appropriately respond to the political objective of integrated and adaptive resource management.

This paper aims at broadening the theoretical corpus of PES in the light of implementing IAWRM. It responds to a number of questions raised in actualising this vision. Specifically:

1. How do the economic characteristics (e.g. rivalry and excludability) of ecosystem services inform the set of institutions required for effective and long-term natural resource management?

2. In what ways is human well-being dependent on the resiliency of ecosystem services, particularly for achieving social equality in terms of human capabilities?
3. Under what institutional parameters can PES policy promote flourishing socio-ecological watershed systems?

In addressing these questions, we expand upon a nested institutional framework for socio-ecological governance and situate the parameters under which a variety of policy tools, including PES, are best employed within this framework. The scholarship on this framework is embryonic with only a few authors emphasizing the physical qualities of ecosystem services and how these align with appropriate governance from the local to global scale (Brondizio et al., 2009; Lebel and Daniel, 2009; Vatn, 2009; Galaz et al., 2012; Yashiro et al., 2013, Duraiappah et al., 2014). Accordingly, this paper contributes substantially to the theoretical articulation of this framework. Nested institutions re-conceive governance across geographical space according to the economic characteristics of rivalry and excludability of ecosystem goods and services. Moreover, emphasis is placed on the hierarchical imperative of governing natural resources by aligning property rights regimes with the characteristics of ecosystem services. By doing so, the framework advocates the enhancement of both individual and social well-being attributed from the effective management of ecosystem services.

The following section introduces this framework of nested institutions and the economic characteristics of watershed goods and services that critically limit the extent to which PES can be applied as a policy instrument for delivering specific services. Section 3.4 describes how human well-being depends on the maintenance of ecosystem services with particular emphasis on the improvement of human capabilities to 'do and be' what is of value. Section 3.5 examines how the implementation of PES within a nested institutional framework can integrate individual capabilities with the collective capacity to successfully adapt to uncertainty and complexity in a watershed context. Section 3.6 expounds on the continuously evolving nature of a nested institutional framework in actualising IAWRM as well as the opportunities and challenges of PES arrangements within this context. Section 3.7 concludes.

3.3. Nested governance arrangements for water resources

The governance of watershed goods and services involves management of the commons- which are essentially shared sets of interests and values towards natural resources by groups of individuals or communities residing in a given watershed basin where these goods and services derive from (Ostrom, 2005). A heuristic approach for classifying watershed goods and services can be conceived according to their characteristics of excludability and rivalry (Farley and Costanza, 2010). Excludability refers to the set of institutions required to divide and manage groups or individuals in relation to their capacity to provide or prevent others from utilising a good or service (Kemkes et al., 2010). For example, the amount of fish collected from a river can be claimed and later consumed by a particular individual thereby excluding the right of others from accessing any fish caught. The exclusion of actors benefitting from the natural filtration of wastewater on the other hand is difficult since the provision of this service is unlikely to align with the discrete actions of individuals according to their property rights. Instead, the service may depend on a contiguous area of wetland, spatial and temporal nonlinearities and thresholds as well as uncertainty at the frontier of the evolving socio-ecological system. The ease with which ecosystem service provision becomes excludable depends on geographic scale to which benefits accrue, the number or actors impacting and utilising natural resources as well as the degree that targeted land-use activity can be directly attributed to clearly identifiable patterns of ecological degradation (Escobar et al., 2013). For example, point sources of industrial effluent or direct release of human waste into a water body by clearly identifiable land-users can be directly attributed to reductions in water quality. These features of excludability in the provision of ecological services align with literature on collective resource management, particularly where boundaries are clearly defined and where congruence exists between local conditions and provision rules (Ostrom, 1990). However, as the number of land-users impacting water quality increases, determining ecological additionality of marginal land-use changes corresponding to property rights will become increasingly difficult and costly to predict.

A rival good suggests that the consumption of a particular good or service reduces the quantity or quality available to others either spatially or temporally. (Brown et al., 2007). A non-rival good, however does not diminish even with infinite consumption by multiple individuals, such as the

cultural or aesthetic values attached to watersheds (Farley, 2010). As such, the incidence of free-riders to the provision of non-rival goods and services is an inherent characteristic, but can vary according to how congestible the service is. Where the number of beneficiaries of an otherwise non-rival good or service is high enough, the service will exhibit rival properties. For example, aesthetic services offered by high landscape quality are non-rival but could become rival if the high number of beneficiaries reduces the quality of aesthetic value the landscape provides. A number of conditions promote co-operative collective management precisely in emphasizing the non-rival characteristic of benefits deriving from natural resource management and include: the existence of shared understandings on the importance of particular watershed services (e.g. water quality or flood protection), political and social homogeneity, the degree of dependence of these services by beneficiaries, and the existence of locally devised management rules and sanction mechanisms (Corbera et al., 2007; Agrawal, 2001; Ostrom, 1990). Accordingly, management of watershed goods and services should pay close attention to the gradations of rivalry, as this inherent feature will imply the requisite institutional structures necessary to govern these resources. The sum of watershed services each reflecting a unique combination of excludability and rivalry characteristics requires classifications of these goods and services along the lines of: a) private, b) public, c) toll or club, and d) open access. Hence, institutional arrangements should reflect the physicality of watershed services and adapt to these four classifications accordingly.

As Ostrom (1990) specified, institutions define boundaries or constraints on behaviour, are sustained by dynamic and adaptive consensus, and are manifested as societal norms or more formal rules. We argue it is the conceptualisation of nested institutions which frames the role that PES might play in implementing IAWRM in managing the watershed commons. A nested institutional approach calls for a complete reformation of human governance systems to reflect contextual forms of knowledge, deliberative participation and a shared understanding of human needs and ecological limitations associated with the biosphere. It is argued that varying property rights arrangements for bundled sets of ecosystem goods and services requires legitimisation from higher levels of government and that a nested institutional approach facilitates the management of a bundle of ecosystem services through horizontal coordination across geographical space and vertical legitimacy from higher to lower levels of governance (Yashiro et al., 2013).

This approach shares many of the characteristics of polycentric governance with the latter being characterised by multiple decision centres at different scales arising spontaneously as a product of competition between different ideas or ways of life, but with a shared understanding that is culturally or institutionally enforced providing a built-in mechanism for self-correction (Aligica and Tarko, 2012). The key distinguishing feature however is the recognition that multiple decision centres have unequal power relations between each other. Indeed, unequal power relations exist even within the various domains of polycentric scholarship, such as law, policy, and economic relationships. As argued by Aligica and Tarko (2012: 245), a polycentric order would not allow for a single domain of social order or single decision structure to monopolise the "legitimate exercise of coercive capabilities". Instead, an 'invisible hand' exists between multiple domains and decision structures with the autonomy to establish and enforce active exercise of preferences and opinions for relationships between people. However, the use of "invisible hand" in this context is not only a misrepresentation of market dynamics that structure society but also leaves too many questions answered since it assumes that power asymmetries do not need to be explicitly acknowledged within and between multiple decision centres. For example, a polycentric dynamic seeks to self-correct mal-distributions of authority in domains or decision centres that would otherwise be characteristic of monocentric systems. However, abiding by an "invisible hand" narrative has led to an increase in the vested power of neoliberal overtures of land privatisation and commodification of nature and its services (Castree, 2008). Power imbalances may also exist between other domains of polycentric scholarship; however, this is beyond the scope of this paper. In contrast, nested institutions, when applied to the realm of water resource management, recognises the inherent power imbalance between the economic and governance domains of natural resource management yet aids in bridging the economic characteristic of watershed goods and services with property rights governance regimes that was otherwise left out of the Polanyi-Ostrom research on polycentricism and common resources. Moreover, a nested institutional approach links the concept of subsidiarity with polycentricism by assigning natural resources to the appropriate level of governance to be managed most effectively according to their economic characteristics with the deliberative spaces, evolution of shared or common understandings and spontaneous emergence or exit of plural interest groups or decision centres.

The inherent feature of nested institutions for natural resources distinguishing it from polycentricism is the *hierarchical* linkage between resource characteristics and governance regimes. Such hierarchy is premised on the normative objective of ensuring a socially acceptable standard of well-being while maintaining a socially determined rate of natural resource degradation that keeps society from overshooting biophysical limits. The inherent feature which distinguishes nested institutions from subsidiarity is the ongoing reformulation or adaptive revision of the common goals that underpin the rules of the game (i.e. the structure of the social order). The cohesion of a nested institutional design rests on the existence of social capital that builds trusting relationships within and across institutional levels through adaptive and iterative cycles which re-evaluate and articulate mutual understanding over time. Indeed, several studies have argued that a critical requirement for designing social-ecological governance arrangements depends on a set of shared value systems to ensure resilience to external shocks (Berkes et al., 1998; Brondizio et al., 2009; Yashiro et al., 2013). Furthering the literature on shared value systems, a nested institutional approach also resonates with the argument provided by Norgaard (2010) that a broad set of institutions that support sustainability and the concept of ecosystem services are first needed if economic market failures are to be re-adjusted effectively and equitably. Figure 3.1 provides an illustration and explanation of the nested institutional approach for the hierarchical governance of ecosystem goods and services while linking the framework with the recognition of hierarchy between ecosystems, social systems and the market (Gowdy and O'Hara, 1997).

Attention to economic characteristics of watershed goods and services and legitimization of nested governance levels can aid in better situating PES as a policy tool for satisfying socio-ecological benefits over time. Since watershed services such as water quality provide both local collective and global benefits but are directly influenced by the land-use actions of individuals, the use of water and land resources by individuals would ideally be suited to community norms and regulations of behaviour (where benefits from resource management accrue at the collective level.

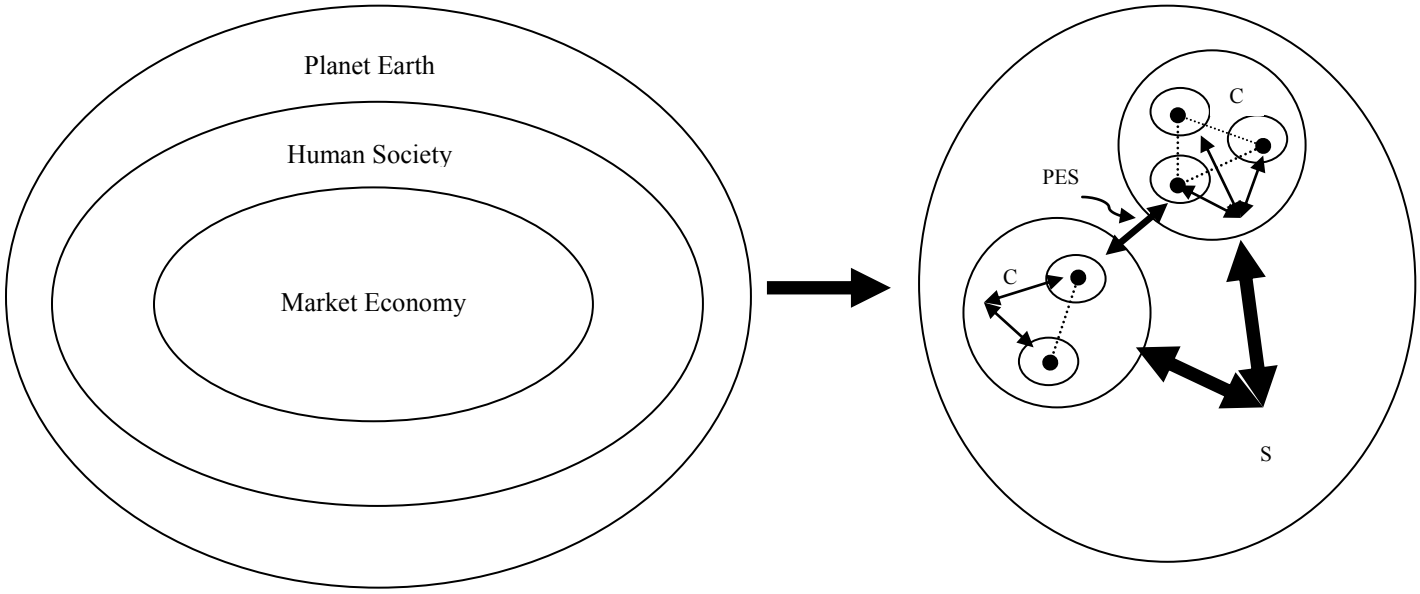

Figure 3.1: Nested Institutions for the management of watershed goods and services

An illustration of nested institutions for the management of watershed services (right), adapted from the vision of sustainability hierarchies as proposed by Gowdy and O'Hara (1997) (left). Nested institutions distinguish the State (S) which regulates public ecosystem goods and services such as global carbon sequestration and devolves governance of other watershed services such as regulating or cultural services to collective associations or communities (C) which in turn govern ecosystem provisioning goods amongst individuals while establishing norms of practice for trade amongst individuals (smallest ovals) for these goods. PES incentives can operate as a negotiation between regulating ecosystem services as a form of trade between communities (arrow linking collective associations). The double-headed arrows represent hierarchical direction of governance based on watershed service characteristics, through deliberative negotiations between geographic scales. The shapes and sizes of the illustrations used are depicted for conveying understanding and do not necessarily represent ideal governance boundaries.

These local collective norms are themselves situated within regional norms and regulations (where benefits from resource management accrue at the regional or global level). Since PES calls for specific behaviour to be adopted by individual upstream service providers, its implementation is best suited to the collective level of governance with higher level governance legitimizing the management of regulating, cultural, and provisioning good and services to the collective level. In order to effectuate these norms and rules of behaviour, governance is best seen in a hierarchical manner in terms of responsibilities of specific spheres of social organization to manage particular sets of watershed goods and services. However, unlike traditional top-down governance, communication between public and collective spheres would be based on a shared understanding that the articulation and crafting of policy solutions to meet

collective users' needs would be bounded by the impact these demands have on regional or global watershed goods and services and the well-being these provide to beneficiaries on broader scales. Similarly, the design of policy solutions to meet individual needs from watershed goods and services would be bounded by the endowment of collective users' needs as distributed by the public sphere in ensuring the delivery of services which provide regional or global benefits.

In furthering this institutional design, we propose a conceptual framework that best identifies the place for PES in managing watershed services as well as the institutional level for which PES should be applied in order to effectuate integrated and adaptive water resource management. By identifying a nested institutional approach, we are suggesting that there does not exist a 'one size fits all' mentality for managing watershed services. Therefore, embracing a definition of PES characterised as sheer markets for rival and excludable services strangles the effective delivery of ecosystem goods and services characterised by varying degrees of rivalry and excludability that do not align with sheer markets The specific configuration of rivalry and excludability of a particular good or service influences the institutional arrangement and consequently the regulatory style adopted for service procurement and well-being enhancement. The framework (Figure 3.2) is separated into four classifications discussed further in section 3.5. These classifications direct attention towards micro (individual and community considerations and property rights) and macro (multi-scale socio-economic and biophysical boundaries) considerations and uncertainties by offering a participatory evaluative space for identifying perceived capabilities stemming from water resources for both upstream and downstream actors. The following section will examine these classifications of goods and services more closely by expounding on the concept of capabilities in strengthening the human dimension of watershed management.

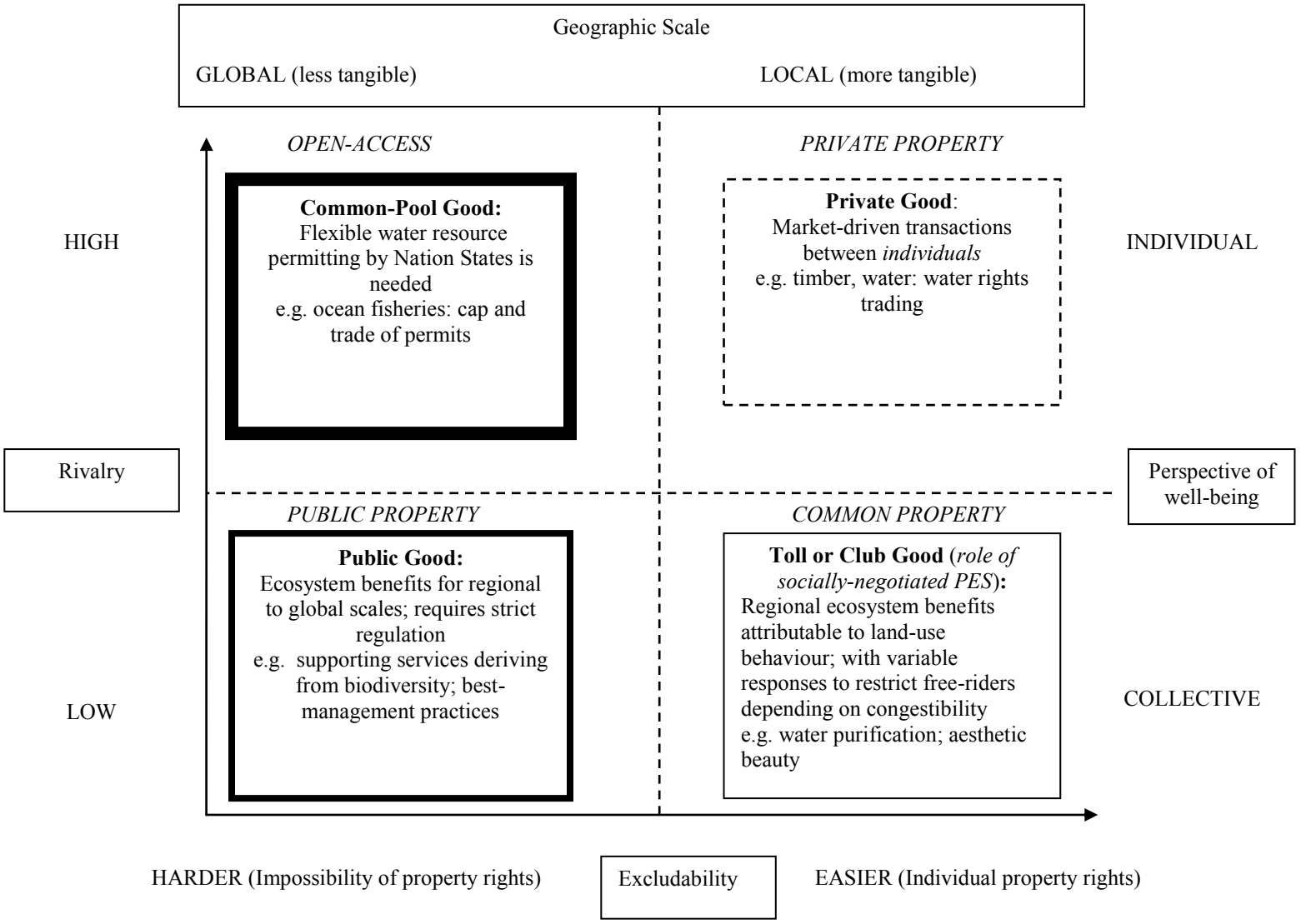

Figure 3.2: A heuristic framework for ecosystem goods and services (adapted from Yashiro et al. 2013)

The thickness of boxes surrounding each characterization of watershed service reflects increasingly larger scales of institutions needed for effective governance, with private goods being the most nested among toll or club goods, public goods, and common-pool goods respectively. Open Access/Common pool watershed services cater to individual perspectives of well-being due to their highly rival nature, but affect interests at the global scale. Public goods have collective benefits that accrue to global, regional, and individual interests. Toll or club goods have collective benefits that accrue largely to regions and individuals. Private goods have primarily individual benefits and accrue to individuals. Socially-negotiated PES transactions can be crafted for common goods watershed services such as water purification in which individual or community property rights agreements exist, but which possess inherent non-rival properties. Examples of ecosystem services and policy approaches are provided in each box.

3.4. Multi-dimensional well-being within watershed management: the Capabilities Approach

It has been recognised that any measure of poverty is both a value judgement and reflects multiple dimensions of experiences (Sen, 1980). The state of poverty extends beyond income effects to include specific functionings allowing individuals greater choice and direction over their own lives (Soussan and Lincklaen-Arriens, 2004; Sen, 2010). These functionings, envisioned as entitlements to improve the state of livelihood opportunities, can aid in more holistic development. They include positive changes in the flows of natural, social, human, physical, and financial dimensions (Reddy et al., 2004). Accordingly, poverty is an experience that can be present in multiple dimensions at any location and at multiple points of time in the lifetime of an individual, household, or community.

As Duraiappah (2011) pointed out, more research is needed to understand how and why natural capital improves these multiple dimensions of well-being. Given that those most deprived in the dimensions of well-being mentioned are the least likely to possess sufficient bargaining power to argue for specific benefits distribution arrangements, it is not difficult to imagine how a PES scheme can reinforce existing inequities amongst stakeholders (Pascual et al., 2010). Several studies have assessed the role of PES in improving well-being but have focused narrowly on household income effects associated with the economic incentive (Gauvin et al., 2010; Jack et al., 2008; Uchida et al., 2007). Hope et al. (2005) revealed that government-led payments for forest-related ecosystem services in Costa Rica hardly influenced well-being, in part because payments were not geared towards the livelihood constraints related to market access, credit, and transport infrastructure that were considered more important than partial compensation payments. A common concern that follows efficiency-oriented PES, as observed in the case of Costa Rica's national scheme, is the targeting of larger land parcels of wealthier and more socially vocal service providers at the expense of smallholders and the landless (Grieg-Gran et al., 2005; Kosoy et al., 2007). Few studies have considered social deliberation and institutional pluralism for designing PES schemes beyond efficiency objectives. A study by Vignola et al. (2012) employed a value-based procedure for negotiation to deliver ecosystem services based on decision-analysis. The application of this approach was based entirely on end objectives of

stakeholders and the appropriate institutional means to achieve these ends. We further this approach in calling for payments to reflect deprivations in capability sets for stakeholders.

The concept of capabilities re-directs improvements to well-being stemming from what people can actually do, be or have a choice over, as opposed to distributing abstractions in new ways, such as monetary wealth, that indirectly influence abilities to do and be (Robeyns, 2006). It also shifts the utility-based goal of development studies to one built on equity, while fully embracing a normative and socially-derived account of what development means (Nussbaum, 2003). The ability to do and be is not only a process of solitary agency but is affected by wider institutional arrangements and societal relations with others and may indeed necessitate the consideration of communal welfare even solely from the perspective of narrow self-interested utility. The emphasis on capabilities focuses on human life and a recognition that each individual should have the same opportunity to be or do whatever they desire. Whether they harness such opportunities for their own well-being is secondary to providing the *choice* in the first instance. In essence, enabling the conditions in which people can make choices over their life (for better or worse) can foment the achievement of further dimensions of well-being. As Nussbaum (2003) pointed out, determining well-being as one measure of resource or utility cannot expect to cover the bases of all components of well-being since individuals have differing abilities to convert such resources into functionings for well-being. In this regard, we consider Sen's (1980) example of a physically handicapped individual who may be monetarily wealthy but still have difficulty being accepted by others or possessing certain skill sets excluded to her including aspects of life that are important for physical or mental well-being.

From a broad anthropocentric perspective, both the ecosystem service concept and the capabilities approach are fundamentally associated with benefits for human well-being. Thus, there exists a conceptually attractive relationship between the two concepts. Polishchuk and Rauschmayer (2012) developed a framework for linking ecosystem services with the capabilities approach in order to better understand the link between basic human needs and the sustainable management of resources to ensure these needs are met equitably. In this framework, the authors recognise that the achievement of enhanced capability sets from ecosystem services depends on: a) the physical configuration of the biotic and abiotic landscape in a particular location; b) the

social rules and norms that influence how and the extent to which human societies can modify and manage that landscape to provide a bundle of ecosystem services or trade-off certain services for others, and c) individual capacity in terms of knowledge, motivation or ability to utilize the ecological resources people have in order to enhance functionings from them (Polishchuk and Rauschmayer, 2012). For example, the capability of having high productive yields over time from farmland is not only a function of the physical configuration of ecological structure within a watershed basin that provide the bundle of services that ensure sustainable yields (such as nutrient cycling and erosion prevention), but also on the conversion factors of individual knowledge and social pressures, norms or institutions to manage these physical barriers to most effectively deliver the benefits associated with watershed services. Thus, there is both an environmental (inherent) and human (dynamic) dimension to converting ecosystem services into the benefits that improve capabilities. These inherent and dynamic dimensions closely resemble the inherent rival and dynamic excludability characteristics of ecosystem goods and services as mentioned earlier. Thus, capability sets deriving from ecosystem services are intrinsically bounded by the rival nature of the particular service and the associated institutional structures that govern such resources. Indeed, it is the human management capacity to ensure physical or environmental conditions either improve or are maintained over time in order to effectuate a closer linkage between the service and the benefits for human well-being. This management capacity depends on the ease of excludability of the service(s) to reflect land-use behaviour according to context-specific property rights. In a given context, deliberative and unrestrained stakeholder participation is critical to capture diverse perceptions of ecosystem service "benefits", perspectives of what is considered as "well-being" and the opportunities and constraints that individuals have in achieving these considerations of value in their lives (Polishchuk and Rauschmayer, 2012; Martínez-Alier et al., 1998).

The relevance of this interpretation of capabilities for PES lies in the role of negotiated exchange through mutual agreement, which may only arise successfully when such exchange is not constructed with the aims of living up to externally-imposed 'goals' or conceptualisations of success (Vignola et al., 2012). This objective is in stark contrast to the established perspective of PES fixated on efficiency-oriented payments as the *only* 'means' by which socially optimal 'ends' can be met. Indeed, behavioural economics studies have illustrated that benefits deriving

from public goods and services may result from highly contextual configurations of ethics, cultural traditions or norms of behaviour regarding societal relations to nature (Bowles, 2008; Cardenas et al., 2000; Fehr and Folk, 2002). Rather than continuing to simplify socio-ecological relationships as a zero-sum game between natural and human systems, a capabilities-focused approach to PES would permit negotiation according to more diverse and culturally specific ways of human interaction. It will enable both technical and socially valid conceptualizations of land-use activity and ecological relations as well as providing the evaluative space for fair and legitimate outcomes. In this manner, PES could be a conduit in defining watershed development as being welfare-enhancing for both upstream and downstream entities. Thus, in designing a well-being focused scheme for achieving capabilities improvements, it would be necessary to start by understanding the social context, traditional knowledge, current livelihoods, distributional issues, and the needs and values of ecosystem service beneficiaries and providers. This information would serve as a prelude for elucidating key design aspects of an incentive scheme, such as the nature of support and terms of a given contract, distributive equity of arrangements, length of contracts, and the potential role of local organizations and other external intermediaries (Vignola et al., 2012).

3.5. Situating PES interventions within nested governance arrangements

Now that the capabilities approach has been illuminated, we can turn attention to how such an approach can be applied within PES; the applicability of incentive-based negotiations from a nested institutional perspective; and the extent that well-being improvement can be achieved in applying this framework for watershed management. . Starting from the identification of common pool watershed goods and services that are both rival (and highly congestible) and costly to exclude (upper left corner of Figure 3.2), it becomes clear that such resources are most vulnerable to exploitation unless strong regulation and management regimes are defined to ensure their protection. This is due to the limited physical stock of the service (e.g. number of commercial fish in the sea) combined with the geographical space of the resource flow, making it hard if not impossible to exclude. Other services such as the sequestration capacity of the atmosphere and oceans can also be conceived as common pool since they are difficult or impossible to exclude and are becoming rapidly congestible (increasingly rival). This results

from a mismatch between the rate of industrial activity and the sequestration or absorptive potential of the oceans and atmosphere. In addition to strong regulation or moratoria on exploitation decided upon at a global scale, tradeable permits distributed to polluters or fishing industries and tied to strict regulatory caps for exploitation can offer a flexible and conciliatory policy solution for governing common-pool goods. However PES, either conceptualised as a market-based trade or as a deliberative social negotiation, will be difficult to implement given the scale and the number of beneficiaries and thus potential free-riders.

Moving to more excludable goods and services that are also highly rival in nature (upper right corner of Figure 3.2), provisioning ecosystem services such as consumable water, timber, or fodder can be best allocated in a Coasian social agreement in which trades are made among individuals who vary in their endowment and demands for these resources. As Muradian et al. (2010) highlighted, very few real world situations match such efficient arrangements for regulating ecosystem services. This is due to asymmetries of information in terms of how much service providers actually should receive to match their foregone losses; how such services can be commodified into measurable quantities to monitor compliance of the agreement, and how much the service is worth to beneficiaries in relation to the compensation they offer to service providers. Norgaard (2010) added that while efficient allocations might arise, these may not necessarily be equitable given the distribution of ecological services and the wealth and bargaining power possessed by potential service providers. Thus, given that many watershed services are indivisible or not readily commodified under a utilitarian notion of value, these private-style arrangements are not suitable for the majority of watershed goods and services with the exception of water-use trades and other extractable goods and services (timber, fuel, fibre, medicine).

However, even these markets in provisioning ecosystem services will need to be governed by collective institutions that take into account the other ecosystem services (e.g. regulating, supporting and cultural) that derive from provisioning services which are both costly to exclude and are non-rival in nature. Such a consideration of markets will transcend purely individual benefits of ecological structure to consider their benefits at broader scales. In doing so, both individual and collective well-being stemming from these resources can be assessed (Yashiro et

al., 2013). Those goods that are non-rival but excludable (lower right corner of Figure 3.2) include toll or club goods and services such as water purification, flood protection and cultural and aesthetic services from watersheds and are clearly attributable to actions of particular individuals through identifiable land-use practices. Within this typology of toll or club goods and services, we can conceive of two streams of non rivalry: a) congestible non-rival and b) purely non-rival. Management strategies for both of these streams require addressing the actions of free-riding beneficiaries, but through separate means. It will be difficult to collectively mobilize free-riding beneficiaries to ensure that the quality of non-rival but congestible goods and services are maintained over time. The challenge for managing rival and congestible goods relates to disproportionate benefits from a high number of users jostling for 'space', making collective action a costly and potentially inequitable process, particularly for those individuals who are less well represented. Thus, efforts to 'de-congest' the service(s) to once again exhibit non-rival characteristics of mutual benefit, may necessitate the introduction of specialised incentives. For example, this may involve the establishment of entry fees to areas of high landscape quality in order to maintain the level of aesthetic quality a scenic area provides. On the other hand, purely non-rival goods and services (not affected by congestibility) provide equal benefits to users regardless of how many beneficiaries there are or how much or how little value is being drawn from them.[2] As such, their appropriate governance should most accurately reflect established local norms for maintaining or maximising the collective or mutual benefits that these services provide, especially where their delivery can most easily be traced to the actions of specific individuals (higher excludability). The key point here is the inherent non-rival features of these services which precludes the possibility of market-based trades, since restricting free-riders would become prohibitively costly or even impossible. Accordingly, it is in this unique arena that deliberatively negotiated- PES schemes that seek to enhance capability sets can take place.

[2] We also incorporate a dimension of 'tangibility' that applies both to non-rival goods and services as well as either excludable or non-excludable actions that influence the provisioning of goods and services. Tangibility of the service reflects the geographic scale to which specific services are realized and influences the intrinsic motivation for collective action potential (Ostrom, 1990). For example, carbon sequestration is a non-rival ecosystem service but accrues tangibility primarily at the global level, making collective action difficult regardless of whether the marginal activities of individuals for enhancing carbon sequestration are excludable or not.

Such arrangements may have slipped under the PES radar either because they are not efficient transactions or were designed according to local norms of land-use stewardship rather than conditional improvements in the delivery of watershed services. The Dhulikhel Drinking Water Supply Scheme in Nepal provides a useful example of an incentive-based negotiation for water quality management that emerged endogenously among stakeholders involved in a broader government agenda which aimed to improve water transport infrastructure to ameliorate water scarcity in the region (ICIMOD, 2010). In 2001, Bhumidada Village Development Committee, which lies within an upstream community-managed forest, realized their role in providing valuable water resources to the town of Dhulikhel and initiated negotiation with the Dhulikhel Development Board and Municipality over the provisioning of potable drinking water through specified land-use activities. These ranged from reforestation, restriction of fuelwood use to coarse woody debris and restricted grazing of livestock in the proximity of water bodies. In return, Bhumidada developed a seven-point demand over a period of 50 years reflecting a set of collectively determined development goals that included financial assistance to manage an area of spiritual importance, provisions of scholarships for students to study higher education, subsidies for medical treatment, and resources for forest stewardship and monitoring (ICIMOD, 2010). The confirmation of terms and conditions of the negotiation resulted from over a year deliberation, with negotiation over payments ongoing for over ten years.

Incentives for water flow regulation and water quantity in the Cauca Valley of Colombia provide another example of capability-focused payments (Escobar et al., 2013). Municipal water user associations were developed between downstream beneficiaries and upstream farmers of the Bolo River watershed with the aim of prioritising conservation interventions in the watershed. Deliberation through a General Assembly of the water user associations resulted in a PES mechanism whereby downstream beneficiaries financed in-kind compensation for socially perceived obstacles to development, ranging from community organisation, sanitation and agri-environmental programmes for long-term land-use productivity (Escobar et al., 2013).A key aspect of a social negotiation approach reflects processes of adaptive learning over time with regards to perceived water quality improvements and dynamic local needs. Depending on existing norms or histories of interaction between communities, transaction costs of negotiation will vary but are critical investments for achieving enduring eco-social objectives. The form that

incentives might take will naturally vary according to norms of behaviour that govern collective action over natural resources in a given context and the impact that incentives have on these norms (Kerr et al., 2012).

In order to ensure that individuals and communities beyond the local-level can also benefit from the public nature of watershed goods and services that are non-rival and also non-excludable, legitimisation of PES arrangements from higher-level governance is needed (bottom left corner of Figure 3.2). Purely public goods include biodiversity existing in a watershed basin and the benefits it provides for supporting services of the watershed (e.g. nutrient cycling, landscape structure) and hence the delivery of other regulating and provisioning watershed services. Governance at the public goods level will need to be based on strict regulation while providing legitimacy to lower levels of governance for the management of regulating and provisioning goods and services.

As mentioned earlier, the smaller the watershed basin, the more likely the provision of watershed services can be viewed as excludable. When the individual benefits deriving from particular watershed services do not diminish with increasing consumption of the service (i.e. non-rivalry), collective understandings of the benefits that derive from these services are more likely to be homogenous. The combination of shared norms or equal dependence on service benefits with locally devised access and management rules for watershed resources provides a context for legitimising the governance of watershed services for collective benefit (Corbera et al., 2007). This is especially pertinent where the provision of services is clearly identifiable to the activities of particular individuals. We argue that such characteristics (excludability and non-rivalry) of watershed services are most likely to support incentive-based negotiations for improving watershed services and enhancing well-being. However, payments may not *always* be an appropriate policy tool to promote the management when the likelihood for collective action is compromised due to economic, social or political heterogeneities between upstream providers and downstream beneficiaries. Where social trust or histories of positive interaction between and among communities is lacking, legitimising the distribution of payments to ensure improvements in watershed service provision will be difficult to achieve (Corbera et al., 2007). Moreover, the communication of emerging knowledge systems, adaptability and social learning to broader

geographical space, and consideration of dynamic economic drivers and political players contribute to the challenge of sustaining fair negotiations (Brondizio et al., 2009). This understanding has important implications for considering not only where PES can be considered as an appropriate intervention (i.e. according to the excludability of service provision or the strength of existing customary practices for water resource management), but also the ideal governance scale for maximising the likelihood that its application will enhance well-being.

The alignment of the economic characteristics of watershed goods and services with appropriate nested governance levels illuminates the trade-offs that exist in governing a typology of watershed goods and services between higher and lower levels of governance. These trade-offs are especially observable when considering the management of rival ecosystem goods and services that differ in terms of excludability. It is difficult to imagine for instance how decentralised governance could effectively manage open-access goods when they are influenced by actors beyond the local scale. Likewise, it would be grossly inefficient for higher level institutions to fully regulate the trade of highly excludable and rival commodities in which the value possessed resides in individual actors driving demand and inspiring entrepreneurship to ensure supply. Similarly, collective action for non-rival goods and services is influenced by trade-offs that exist between more or less excludable goods and services. The protection of diverse assemblages of species to ensure the resilience of critical supporting ecosystem functions requires global direction since the consequences of failure through species extinctions may sever life-support systems (de Klemm and Shine, 1993; MA, 2005) The benefits of these supporting ecosystem functions are mostly non-rival, subject to significant nonlinearities and thus may or may not be influenced by discrete and readily identifiable activities of individuals in a localised manner. (MA, 2005; Rockström, 2009) Given such uncertainty, a precautionary approach at the global scale is needed to ensure human society does not degrade species diversity beyond a socially optimal rate.

3.6. Realising IAWRM

As illustrated in Figure 3.2, the management of watershed services can be differentiated from a multi-dimensional perspective. Watershed goods and services can be distinguished both by the

economic properties of goods and services as common-pool, public, toll/club or private; relational configurations that define property regimes for different goods and services (e.g. open-access, public property, common property and private property) as well as geographic scale of governance reflecting global or watershed-scale governance, regional-scale governance, and localised market-based arrangements respectively (O'Neill, 2001). In this manner, toll or club goods and public goods can be differentiated by unique governance arrangements that are *nested* within each other (depicted as variations in the width of the quadrants in Figure 3.2). The legitimacy provided to various PES-based negotiations on a community level for regulating and cultural goods and services as well as private market trades in provisioning services provides an opportunity for adaptive co-management. This approach emphasizes both individual and collective capability sets, reflecting on local needs, resource limits and in building social capital between institutions and individuals across governance scales (Folke et al., 2005). In this manner, the management of ecological functioning (e.g. prioritizing good soil quality and natural diversity of riparian zones) in a watershed is linked to the distributional considerations of improving functionings for well-being (e.g. being able to engage in agriculture for subsistence or for cash cropping).

Hence, an IAWRM vision can be operationalised through nested institutions that transcend the boundaries of the watershed to be realized at the private, collective and public spheres of governance according to particular economic characteristics of the ecosystem good or service being considered. For example, the management of global watershed services such as climate stabilisation and habitat for biodiversity, which are both open access and public goods, is best left to a watershed-scale agency that incorporates the needs of beneficiaries on regional and global scales through strict regulation of behaviour while devolving the management of more excludable but still non-rival watershed services to the collective level. In turn, the collective or community scale would then manage watershed services such as flow regulation or erosion control in which collective well-being requires changes in individual behaviour. In this case, the use of incentives through PES interventions are leveraged in a manner that provides flexibility and local legitimacy for collective self-regulation of these ecosystem services. Finally, collective-level governance would involve the management of those watershed resources that provide private benefits such as fisheries, timber and other forest products. These goods and

services can then be traded through private enterprise in markets which follow and are limited by collective norms. There exists a mutually supporting role between nature and society in managing natural resources such as water, and this implies closer attention to appropriate governance and bargaining space for negotiation and dialogue among diverse interests.

Figure 3.3 illustrates how IAWRM can be contextualised and operationalised according to social organization and watershed goods and service characteristics from a nested institutional perspective. In meeting individual and collective 'dependencies' in the form of sustained ecosystem goods and services, appropriate watershed governance requires *managing* the endowment of ecological structure that comprises critical ecosystem processes and functions. Hence, the nature of the relationship between the ecological endowment and the dependencies in terms of rivalry and excludability offers suitability towards appropriate institutional arrangements and policy tools which feedback to ensuring ecological endowments are not exhausted over time (Daly and Farley, 2004). It is from this perspective that we can more accurately situate PES arrangements within a nested institutional framework that aligns with broader narratives or visions that respond to objectives of integration and adaptation in natural resource management.

It is important to recognise limitations of this re-conceptualisation of PES. Since the poor have a more limited range of land-use options available to them, imposing restrictions on the set of land-use options for them may actually be capability-reducing (Tacconi, 2012). This presents a barrier to utilising PES for enhancing capabilities which can only be overcome where wise land-use management positively reinforces (physically or perceptively) an increase in opportunities and states of functioning. This can be achieved when a specific land-use intervention, such as crop rotation or agroforestry systems, not only ensures soil protection, but also increase land productivity over time. Moreover, the closer the alignment between the identification of beneficiaries of watershed services, and the legitimacy of who and how payments (in whatever form) should be provided to service providers, the more likely potential that payments can be defined through negotiation to reflect capability sets of service providers (Fisher et al., 2010; Escobar et al., 2013).

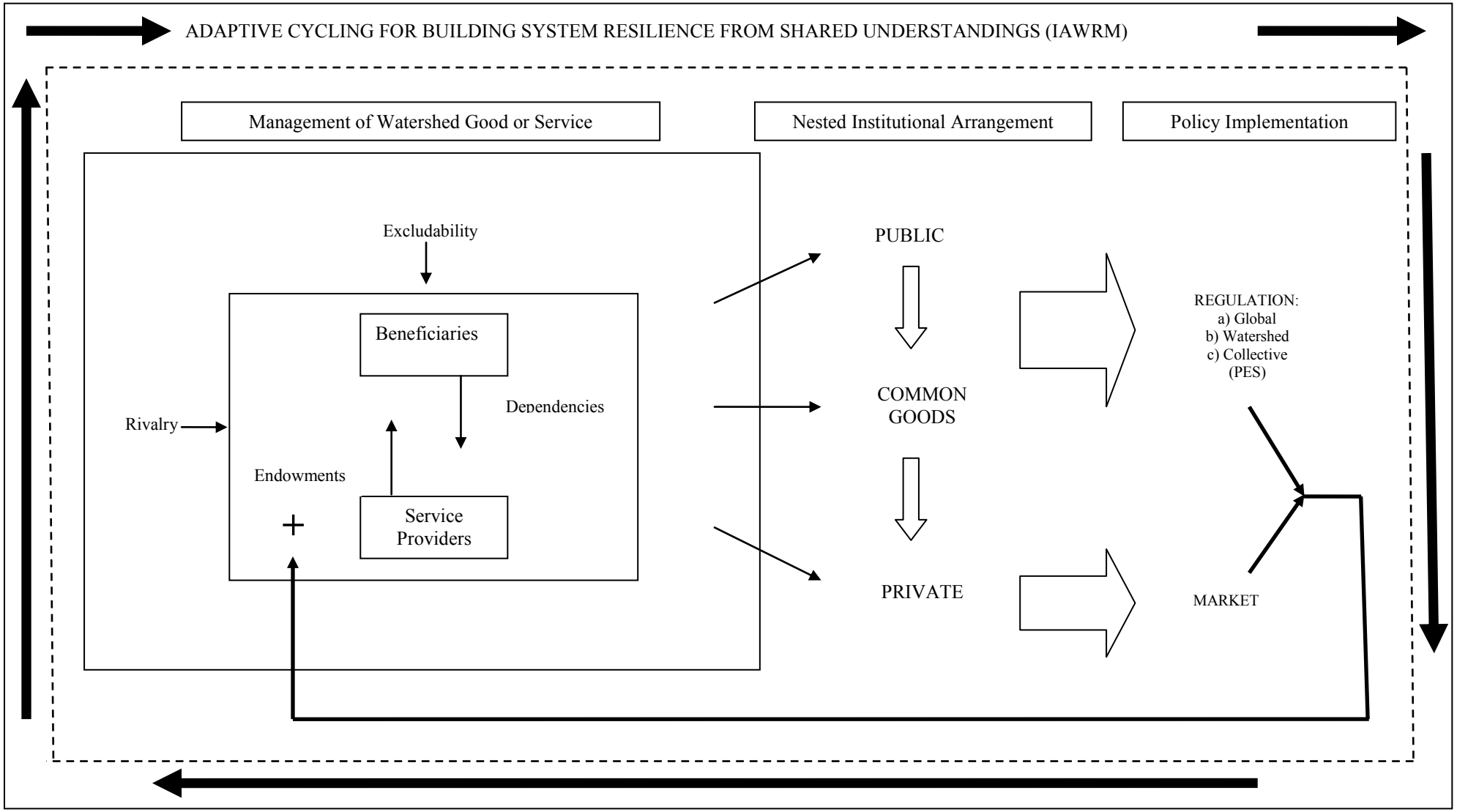

Figure 3.3: A framework for contextualizing and operationalizing IAWRM

The specific economic characteristics of watershed goods and services influence the institutional arrangement for governing a particular good or service and hence the policy tool used for implementation. Maintaining the vision of IAWRM over time is an evolving learning process that requires adaptive cycling through feedbacks of knowledge and re-articulation of shared values in order to manage functioning endowments of watershed goods and services.

However, the process of identifying and legitimising who pays and who receives payment itself requires a fair and deliberative space in order to avoid politically charged interests or the manifestation of unequal power relations between actors (Fisher, 2013). Since this process is not without costs, PES incentives should be viewed as the mechanism for self sustaining and refining collective action, rather than on the economic benefits of the incentive itself as a new source of financing for watershed services. In other words, the enhancement in capability sets associated with PES is more a process of collective negotiation and resulting social capital than the payment itself.

Socially-constructed payments might better reflect underlying needs and values of service providers and engender relevant payment designs to achieve these values through an inclusive, adaptive and deliberative decision analysis space (Lebel and Daniel, 2009; Muradian et al., 2010; Pascual et al., 2010). However, in order to frame PES as a social contract based on perceived service improvements and built according to existing informal institutions, it is necessary to clarify a particular misconception regarding transaction costs. These refer to the information investments that lead to improved capacities to manage natural resources over time and for long-term benefit. If these costs of negotiation are instead viewed as obstacles to efficient transactions, PES may fall short of being successful over the long-term. A second misconception refers to the short and rigid project-oriented timelines or ecologically unrealistic conditionality requirements often tied to payments (Fisher, 2013). This objective, often linked to the loans provided by international aid agencies, will also fall short of acknowledging context-specific biophysical and socio-economic complexity and may lead to failure in water resource management. Finally, conditionality requirements for PES arrangements must be constructed with respect to open dialogue on specific ecological uncertainties and stakeholder perceptions of well-being improvement over time. For example, service providers and service beneficiaries can identify perceived uncertainties involved in realising traceable improvements to service flows to downstream users by proposing locally relevant indicators. Likewise, similar indicators for the

alleviation of multiple components of well-being deprivation as identified by upstream service providers can be identified to evaluate the nature of compensation offered over time.

3.7. Conclusion

This paper suggests that a re-conception of payments for ecosystem service schemes within an appropriate governance framework would make greater advances in actualising an IAWRM agenda. PES can serve as a useful policy tool only if it is applied according to the particular configuration of excludable and non-rival watershed goods and services. We have argued that the application of incentive-based negotiations to manage *any other* combination of rivalry and excludability for watershed goods and services will fail to ensure: a) that all service beneficiaries are paying their fair share for the stewardship and maintenance of service flows or b) those receiving the incentives are actually influencing the service(s) for which they are being 'paid'. Moreover, we emphasize that the neoliberal conception of PES as a market-based arrangement makes the incorrect assumption that all ecosystem goods and services can be maximised as though they are highly rival and excludable. This assumption has been illustrated to be erroneous both in regards to determining clear causal relationships between land-use and service delivery or clearly articulated exchange values for the services of nature, but also as being inconsistent with multiple means by which diverse human value systems can cultivate relations and shared social norms through the use of incentives. Paying closer attention to the nuances of these economic characteristics will clarify how a bundle of watershed services can be delivered according to a hierarchy of property rights regimes.

This paper has also emphasized that the nature of PES collective agreements provides the opportunity to improve well-being through enhanced capability sets for individuals. Management of watershed goods and services requires governance at nested scales to further link ecological integrity of the resource base with the welfare-enhancing values that characterize an improved capability set for stakeholders. From this understanding of nested governance, we can visualize IAWRM as providing a bundle of watershed goods and services to beneficiaries at both local and global scales. The analysis provided in this theoretical paper could be part of useful and important groundwork towards empirically assessing the viability of PES arrangements within

common property management regimes as recent studies have advocated. As we have illustrated throughout this paper, PES should be envisioned as socially deliberated negotiations which harness incentives to actualise meaningful improvements in capabilities within cooperative governance arrangements. Future empirical investigation of this framework would also aid in more clearly distinguishing successful PES arrangements from those that have failed. Heterodox valuation and complex decision making can aid in integrating diverse social and ecological knowledge combined with economic, population, and other stochastic pressures to better reflect potential impacts of a management intervention on components of the larger system across time and geographic space.

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3.9. References

Agrawal, A., 2001. Common property institutions and sustainable governance of resources. World Development 29, 1649-1672.

Aligica, P.D., Tarko, V., 2012. Polycentricity: From Polanyi to Ostrom, and Beyond. Governance 25(2), 237-262.

Alkire, S., Santos, M.E., 2010. Acute Multidimensional Poverty: A New Index for Developing Countries. Oxford Poverty and Human Development Initiative Working Paper No. 38. Oxford.

Berkes, F.L., Folke C., Colding, J., 1998. *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*. Cambridge University Press, Cambridge.

Biswas, A., 2004. Integrated water resources management: a reassessment: a water forum contribution. Water International 29(2), 248-256.

Bowles, S., 2008. Policies designed for self-interested citizens may undermine "the moral sentiments": Evidence from economic experiments. Science 320, 1605-1609.

Brondizio, E.S., Ostrom, E., Young, O.R., 2009. Connectivity and the Governance of Multilevel Social-Ecological Systems: The Role of Social Capital. Annual Review of Environmental Resources 34, 253-78.

Brown, T.C., Bergstrom, J.C., Loomis, J.B., 2007. Defining, valuing, and providing ecosystem goods and services. Natural Resources Journal 47(2), 329-376.

Cardenas, J.C., Stranlund, J.K., Willis, C.E., 2000. Local environmental control and institutional crowding-out. World Development 28, 1719-1733.

Castree, N., 2008. Neoliberalising nature: the logics of deregulation and regulation. Environment and Planning A 40(1), 131-152.

Corbera, E., Brown, K., Adger, W.N., 2007. The Equity and Legitimacy of Markets for Ecosystem Services. Development and Change 38(4), 587-613.

Daly, H.E., Farley, J., 2004. *Ecological Economics: Principles and Applications*. Island Press, Washington, D.C.

de Klemm, C., Shine, C., 1993. Biological Diversity Conservation and the Law. International Union for the Conservation of Nature, Gland.

Duraiappah, A., 2011. Ecosystem Services and Human Well-being: Do Global Findings Make Any Sense? BioScience 61(1), 7-8.

Duraiappah, A.K., Asah, S.T., Brondizio, E.S., Kosoy, N., O'Farrell, P.J., Prieur-Richard, A.H., Subramanian, S.M., Takeuchi, K. 2014. Managing the mismatches to provide ecosystem services for human well-being: a conceptual framework for understanding the New Commons. Current Opinion in Environmental Sustainability 7, 94-100.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65(4), 663-764.

Escobar, M.M., Hollaender, R., Weffer, C.P., 2013. Institutional durability of payments for watershed services: Lessons from two case studies from Colombia and Germany. Ecosystem Services 6, 46-53.

Farley, J., 2010. Conservation through the Economics Lens. Environmental Management 45, 26-38.

Farley, J., Costanza, R., 2010. Payments for ecosystem services: from local to global. Ecological Economics 69(11), 2060-2068.

Fehr, E., Falk, A., 2002. Psychological foundations of incentives. European Economic Review 46, 687-724.

FAO 2008. 'The new generation of watershed management programmes and projects'. Forestry Paper 150. Food and Agriculture Organization of the United Nations, Rome.

Fisher, J., 2013. 'Chapter 2: Justice implications of conditionality in Payments for Ecosystem Services: A case study from Uganda. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K., Burgess, N.D., 2010. Common pool resource management and PES: lessons and constraints for water PES in Tanzania. Ecological Economics 69(6), 1253-1261.

Folke, C., Hahn, T., Olsson, P., Norberg, J., 2005. Adaptive governance of socio-ecological systems. Annual Review of Environmental Resources 30, 441-473.

Functowicz, S.O., Ravetz, J., 1994. Uncertainty, complexity and post-normal science. Environmental Toxicology and Chemistry 13, 1881-1885.

Galaz, V., Biermann, F., Folke, C., Nilsson, M., Olsson, P., 2012. Global environmental governance and planetary boundaries: An introduction. Ecological Economics 81, 1-3.

Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. Environmental Management 45, 488-501.

Gowdy, J., O'Hara, S., 1997. Weak sustainability and viable technologies. Ecological Economics 22, 239-247.

Grieg-Gran, M., Porras, I.T., Wunder, S., 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33, 1511-1527.

Hope, R.A., Porras, I.T., Miranda, M., 2005. 'Can Payments for Environmental Services Contribute to Poverty Reduction? A Livelihoods Analysis From Arenal, Costa Rica'. Newcastle: Centre for Land Use and Water Resources Research (CLUWRR). http://www.dfid.gov.uk/R4D/Output/11261/Default.aspx (accessed 10 July 2012).

Huitema, D., Mostert, E., Egas, W., Moellenkamp, S., Pahl-Wostl, C., Yalcin, R., 2009. Adaptive Water Governance: Assessing the Institutional Prescriptions of Adaptive Co-Management from a Governance Perspective and Defining a Research Agenda. Ecology and Society 14(1), 26.

ICIMOD 2010. 'Case Study: Dhulikhel Drinking Water Supply Scheme (DDWSS)' Unpublished report. Kathmandu: International Centre for Integrated Mountain Development.

Jack, B.K., Leimona, B., Ferraro, P.J., 2008. A Revealed Preference Approach to Estimating Supply Curves for Ecosystem Services: Use of Auctions to Set Payments for Soil Erosion Control in Indonesia. Conservation Biology 23(2), 359-367.

Kemkes, R.J., Farley, J., Koliba, C.J., 2010. Determining when payments are an effective policy approach to ecosystem service provision. Ecological Economics 69, 2069-2074.

Kerr, J., Vardhan, M., Jindal, R., 2012. Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. Ecological Economics 73, 220-227.

Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69(6), 1228-1236.

Kosoy, N., Martínez-Tuna, M., Muradian, R., Martínez-Alier, R., 2007. Payments for environmental services in watersheds: insights from a comparative study of three cases in Central America. Ecological Economics 61, 446-455.

Lebel, L., Daniel, R., 2009. The governance of ecosystem services from tropical upland watersheds. Current Opinion in Environmental Sustainability 1, 61-68.

Martínez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecological Economics 26, 277-286.

McAfee, K., 2012. The Contradictory Logic of Global Ecosystem Services Markets. Development and Change 43(1), 105-131.

Medema, W., McIntosh, B.S., Jeffrey, P.J., 2008. From Premise to Practice: a Critical Assessment of Integrated Water Resources Management and Adaptive Management Approaches in the Water Sector. Ecology and Society 13(2), 29-46.

Millennium Ecosystem Assessment (MA), 2005. *Ecosystems and human well-being: scenarios*. Island Press, Washington, D.C.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202-1208.

Norgaard, R., 2010. Ecosystem services: from eye-opening metaphor to complexity blinder. Ecological Economics 69(6), 1219-1227.

Nussbaum, M., 2003. Capabilities as Fundamental Entitlements: Sen and Social Justice. Feminist Economics 9(2-3), 33-59.

O'Neill, J., 2001. Property, Care, and Environment, Environment and Planning C 19, 695-711.

Ostrom, E., 1990. *Governing the commons: The Evolution of institutions for collective action*. Cambridge University Press, Cambridge.

Ostrom, E., 2005. *Understanding institutional diversity.* Princeton University Press, Princeton.

Pahl-Wostl, C., Sendzimir, J., Jeffrey, P., Aerts, J., Berkamp, G., Cross, K., 2007. Managing Change toward Adaptive Water Management through Social Learning. Ecology and Society 12(2), 30.

Pascual, U., Muradian, R., Rodriguez, L.C., Duraiappah, A., 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69, 1237-1244.

Polishchuk, Y., Rauschmayer, F., 2012. Beyond "benefits"? Looking at ecosystem services through the capability approach. Ecological Economics 81, 103-111.

Postel, S.L., Thompson, B.H. Jr., 2005. Watershed protection: Capturing the benefits of nature's water supply services. Natural Resources Forum 29, 98-108.

Pretty, J., 2003. Social Capital and the Collective Management of Resources. Science 302, 1912-1914.

Reddy, V.R., Reddy, M.G., Galab, S., Soussan, J., Springate-Baginski, O., 2004. Participatory Watershed Development in India: Can it Sustain Rural Livelihoods? Development and Change 35(2), 297-326.

Robeyns, I., 2006. The Capability Approach in Practice. Journal of Political Philosophy 14(3), 351-376.

Rockström, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E. et al. 2009. Planetary Boundaries: Exploring the Safe Operating Space for Humanity, Ecology and Society 14(2), 32.

Sen, A., 1980. 'Equality of What?', in S.M. McMurrin (ed) *Tanner Lectures on Human Values*, pp. 353-60. University of Utah Press, Salt Lake City.

Sen, A., 2010. *The Idea of Justice*. Harvard University Press, Cambridge.

Soussan, J., Lincklaen-Arriens, W., 2004. 'Poverty and Water Security': Understanding How Water Affects the Poor'. Manila: Asian Development Bank. http://beta.adb.org/sites/default/files/pub/2004/Poverty_02.pdf. (accessed: 15 March 2013).

Tacconi, L., 2012. Redefining payments for environmental services. Ecological Economics 73, 29-36.

Uchida, E., Xu, J., Xu, Z., Rozelle, S., 2007. Are the Poor Benefitting from China's Land Conservation Program? Environment and Development Economics 12(4), 593–620.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69(6), 1245-1252.

Vignola, R., McDaniels, T.L., Scholz, R.W., 2012. Negotiation analysis for mechanisms to deliver ecosystem services: The case of soil conservation in Costa Rica. Ecological Economics 75, 22-31.

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. Conservation Biology 21, 48-58.

Yashiro, M., Duraiappah, A., Kosoy, N., 2013. 'Chapter 10: A nested institutional approach for managing bundled ecosystem services: Experience from managing Satoyama landscapes in Japan', in: Rival, L., Muradian, R. (Eds.), *Governing the provision of Ecosystem Services*. Springer, London, pp. 191-206.

Connecting Text

Thus far, the need to place PES policy within a contextualised understanding of ecosystem service management and procedural and distributive justice has been discussed. However, PES in theory and practice continues to be conceived and implemented from the perspective that all ecosystem services are purely rival and excludable and hence characteristic of stock goods to be bought and sold (Ferraro and Kiss, 2002; Swallow, 2013; Bonn et al., 2014). Compounding this faulty logic of ecosystem services as stock resources that are completely detached from social processes and human agency is the fixation on economically efficient payments. These targeted payments aim to achieve particular and social and ecological objectives at minimal cost. Economically optimised payments assume that individual land-users reflect upon, choose, and engage in particular land-use behaviours solely to maximize their own utility. Moreover, the design of PES schemes under an economic perspective presumes the superiority of *external* values of project proponents rather than legitimate informal institutions which characterise local values and perspectives. There are significant equity implications of payment targeting since locally legitimate patterns of resource distribution and benefits sharing tend to be ignored (Corbera et al., 2007). For example, economic efficiency would imply that the largest land parcels or households with lowest opportunity costs would be selected for enrolment as identified service providers at the expense of those without secure land-tenure. It is hard to justify in every circumstance, how or why an individual would reflect and make a particular decision in complete isolation from social influences that operate on them.

In the following chapter, the often sought objectives of economic efficiency, ecological effectiveness and poverty-reduction are critically examined through the development of a targeting matrix for PES. The purpose of the matrix is to geographically display the distributional implications of payment allocation across the landscape according to targeted objectives in order to make trade-offs between payment designs more explicit and to reveal the underlying political implications associated with optimising payments. The general objectives of the thesis as well as specific objectives 3 and 4 are addressed here. This chapter was published as a manuscript in the journal *Geoforum* Volume 58 (pages 1-13). (Kolinjivadi, V., Grant, A.,

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Chapter 4

Juggling multiple dimensions in a complex socio-ecosystem: The issue of targeting in Payments for Ecosystem Services

4.1. Abstract

Proponents of payments for ecosystem services (PES) schemes advocate targeting payments to geographical areas that can: a) maintain or enhance ecosystem services, b) permit economically efficient arrangements and c) address poverty objectives. The location of these efficient, effective and equitable (or triple-win) solutions is viewed as the 'holy grail' of PES and is often sought in isolation to broader socio-economic pressures, political relationships, or local cultural conditions. While the plethora of PES design perspectives often follow the concepts of efficiency and effectiveness, they seldom relate to pluralistic value systems and may disparage local self-determination for influencing the form and terms of negotiation. This paper critically analyses the assumptions underpinning the design of PES schemes which seek to optimise or target efficient, effective and poverty objectives. Using a case study for a proposed PES initiative in the Kathmandu Valley of Nepal, we employ spatial analysis to geographically visualise the discrepancy between the location for a typology of targeted objectives and actual preferences which support local perceptions of natural resource use and conservation. The case study highlights the inequity inherent in targeting payments under a neoliberal framing. Instead, spatially differentiating incentives according to socially determined priorities and collective management is suggested.

Keywords: Payments for ecosystem services; efficiency; targeting; equity

4.2. Introduction

Market-oriented instruments for environmental protection seek to efficiently modify the behaviour of land-users to correspond with the needs and values of paying beneficiaries who can compensate the former for the foregone benefits of land-use change (Wunder, 2005; Engel et al.,

2008). One such instrument which has sparked widespread enthusiasm from researchers, government agencies, development banks, and donors in the last decade is 'payments for ecosystem services' (PES). It has been argued that PES and other market-like environmental policies subjugate nature and diverse social relationships as being inferior to rational self-interest and simplistic yet mythical cause-and-effect relationships (Kosoy and Corbera, 2010). Such policies rooted in neoclassical economics essentially reinforce the notion that human society is not only distinct from nature, but also that human value systems are fixed and are not shaped by social influences (Sagoff, 1998). Despite these criticisms, the quest for the optimum PES scheme according to economic, ecological and poverty alleviation objectives remains a seemingly obvious and unwavering priority for PES practitioners (Kroeger, 2013).

The optimisation of PES is increasingly being aligned with several key features of neoliberal economic policy, as identified by Castree (2008), namely: *privatization* (e.g. once private property rights are established, social actors can voluntarily negotiate and incentivize each other to influence behaviour given complete information and minimal costs of negotiation); *marketisation* (e.g. trading ecosystem services in markets where monetary compensation to surpass opportunity costs of foregone activites is considered the dominant exchange value), and market-supporting policies and organisations designed to develop the necessary *institutional apparatus* that facilitate the application of PES through a market lens. Despite reference to these considerations, very few operating PES schemes exhibit characteristics of sheer market arrangements, such as conditionally-dependent payments and voluntary participation (Muradian et al. 2010; McAfee, 2012). Regardless of these ecological and socio-economic assumptions, the conceptually attractive objectives of *cost-efficiency*, *environmental effectiveness* (measured as 'additionality' defined as the perceived ecological service benefits in relation to what would have been provided without the payments) and the combination, *cost-effectiveness* (ecological service value provided per money spent), have been identified in optimizing ecosystem service provision through PES payments under the pretence of a 'market-based' arrangement (Wünscher et al., 2006; Chen et al., 2010). Further studies have included poverty alleviation or social *equity* as another targeting goal and have sought to identify locations tagged as having the 'gold standard', by targeting service providers and their lands according to low opportunity cost, high potential

additionality and managed by poorer households (Gauvin et al., 2010; Jack et al., 2009; Alix-Garcia et al., 2008).

A wealth of studies have emerged attempting to map ecosystem services according to areas of potential supply and demand across varying spatial and temporal scales (Naidoo et al., 2008; Nelson et al., 2009; Daily et al., 2009; Zhang and Pagiola, 2011; Bagstad et al., 2013). The popularity of visualising ecosystem services has resulted in a number of decision-support toolkits of differing degrees of sophistication to generate spatial information on the extent or magnitude of ecosystem service delivery potential or defined metrics quantifying particular services (Potschin and Haines-Young, 2011). Identifying "hotspots" of ecosystem services valued by certain social groups has been suggested to facilitate the prioritisation or targeting of policy efforts to improve the delivery of services and avoid untargeted expenditure of scarce resources. Indeed, both local and global-scale efforts have been made to map the spatial concordance of areas that produce ecosystem services to the values held by beneficiaries of these services (Newburn et al., 2005; Naidoo and Iwamura, 2007; Naidoo et al., 2008; Crossman and Bryan, 2009).

In this study, we geospatially visualise the interactions between targeted objectives which exposes the political embeddedness of various PES distribution strategies. The spatially explicit nature of the tool also serves to reveal the absurdity of seeking 'gold standards' in the face of substantial epistemological and ontological complexity. We argue that targeted payments lacking local cultural meanings, attention to the situational context of poverty, or an analysis of existing political relationships influencing natural resource management will at best fail to result in long-term positive outcomes and at worst reinforce the cycle of poverty and environmental degradation.

In the following section, we outline the objectives of the paper and critically review recent studies which have applied spatial analysis to map ecosystem services and others which leverage upon such techniques to improve PES targeting. Section 4.4 describes the case study and the methodology in addressing the above research objectives. Sections 4.5 and 4.6 offer results of the analysis and a discussion of what these results suggest for improving PES arrangements. A

conclusion is then given proposing further research that adopts a collective-action framing for PES as incentive-based negotiations for socio-ecological stewardship.

4.3. Objectives and Background

While PES targeting cannot derive from a purely technical process, we argue there is value in making ethical and political ramifications explicit in the allocation of PES payments. We propose a PES targeting matrix that incorporates the various objectives in different combinations that PES proponents seek to enhance in the negotiation of incentives for ecological stewardship. In this manner, not only are trade-offs between the allocation of payments identified, but the implications of each PES design scenario become open for critical inquiry in relation to local informal institutions, power relations between actors and inequities of resource access.

4.3.1. PES Targeting Matrix

The development of the matrix employs geospatial techniques to visually illustrate how each targeted PES design scenario identifies unambiguously which households in geographic space would benefit from a given targeting scenario. Such a representation can reveal the extent to which targeting scenarios align with local norms, cultural and informal relationships, or local decision-making institutions and hence the legitimacy of the design. An empirical case study of a proposed PES scheme in the Kathmandu Valley of Nepal is examined using the PES targeting matrix in order to identify geographical locations where payments align with objectives prioritised by PES proponents. These include economic efficiency, environmental effectiveness (e.g. PES 'additionality'), cost-effectiveness, and equity. The latter is measured across two dimensions considering: a) poverty using income distribution as a measure of welfare and, b) poverty in relative terms based on perceived well-being. The interactions of each objective are presented within the matrix in order to illustrate the range of potential payment design scenarios, allow critical judgement on each objective in relation to others according to the preferences of involved social actors, and to determine the extent to which each scenario aligns with or deviates from local understandings of payment distribution preference. Moreover, to reveal the inherently political nature of choosing a particular design scenario, the distributional ramifications of using

the presumably 'objective' parameter of 'opportunity cost' is critically analysed. While previous studies have recognized the mismatch between externally defined political goals and local legitimacy in PES (Corbera et al., 2007; Pascual et al., 2010; Vatn, 2010; Van Hecken et al., 2012; Fisher, 2013, Zander et al., 2013) this study is the first to adopt an explicitly spatial argument to illustrate this point. Specifically, this paper contains the following research objectives:

1. Introduce a targeting matrix to spatially visualise the trade-offs between targeting designs prioritising economic efficiency, ecological effectiveness (i.e. additionality) and poverty alleviation;

2. Reveal the distributional consequences of using opportunity cost as a benchmark for cost-efficiency or cost-effectiveness targeting by considering livelihood constraints between more and less wealthy households identified as potential service providers;

3. Identify the diverse social, economic and environmental preferences of both upstream and downstream social actors;

4. Compare the payment design and distributional preferences of identified service providers with the full set of spatially defined targeting strategies reflecting the objectives of PES proponents, and

5. In the light of these results, this paper discusses the extent to which informal institutions and asymmetries of power between social actor groups determine the degree of self-determination and legitimacy of a particular PES targeting scenario.

The targeting matrix serves to illustrate the point that optimising PES payments is more likely to divide rather than align ecological outcomes and individual and/or societal objectives. At the heart, lies the inherent injustice of imposing a single value metric to reflect changes in social welfare. Indeed, there is a real risk that optimising payments serves to reward those individuals who are ironically least allied with improving the well-being of the majority of social actors,

specifically those with greater bargaining power and resources to dictate patterns of land-use management to serve their own conceptualizations of nature's benefits.

4.3.2. Spatial identification of ecosystem services for PES targeting and the market-based myth

A few studies have emerged which have collated household or parcel-level data on ecosystem service generating potential, opportunity costs, and / or qualitatively expressed socio-economic 'storylines' to spatially identify areas of greatest additionality, lowest cost, or potential for land-use change in the future (Wendland et al., 2010; Swetnam et al., 2011). Satake et al. (2008) used spatial mapping for PES targeting at different scales to reveal landscape heterogeneity in the distribution of carbon stocks on private land and potential equity considerations of global beneficiaries of the service. Elsewhere, Zhang and Pagiola (2011) examined the spatial feasibility of optimizing PES in Costa Rica by identifying synergies determined as the extent of spatial overlap of differing PES objectives, including poverty alleviation, biodiversity conservation and water provisioning. These studies have used different proxies for mapping optimal locations for PES targeting and have paid differential attention to trade-offs in service provisioning according to variations in scale and economic drivers over time. Yet, the prioritisation of conservation areas was often measured according to a single metric of value, such as annual water tariff revenues in the case of prioritising water conservation areas in Costa Rica (Zhang and Pagiola, 2011).

The most convincing studies have involved spatial mapping of conservation 'opportunity' as opposed to conservation 'priority' by incorporating social values, and the capacity of local institutions and engaged populations to enhance the effectiveness of conservation investments. These studies recognize that technological, economic or biophysical characteristics may indicate levels of conservation priority in terms of threat or cost-effectiveness but that social approval, cultural values and existing public policies shape conservation opportunity (Bryan et al., 2010a,b; Knight et al., 2010; Raymond and Brown, 2011). However, mapping areas of high and low social and ecological values respectively to identify policy strategies under each combination treats the development and evolution of social values as separate from natural functions and processes portending that nature and society are mutually excludable. Moreover,

the overlap of compartmentalized social, economic, and biophysical information in a seemingly 'integrated' fashion errantly obscures how entrenched institutions and powerful discourses underpin collective norms or social obligations and are inextricably linked to perceptions of nature and hence natural resource management (Lele, 2013).

Thus, despite attempts to map scenarios of land-use change, these spatial analyses are essentially 'snapshots' which conceal substantial epistemological (e.g. plural versions of value legitimacy) and ontological (e.g. scales of analysis) uncertainties inherent in dynamic socio-ecological systems. Such reductions refer to the consideration of households as self-interested actors responding purely to compensation of opportunity cost (Wendland et al., 2010) or simplifications of social processes in order to strike a balance between relevant detail of the model and usefulness for policy-makers (Swetnam et al., 2011). Accordingly, little explicit attention has been given to distributional concerns or perceptions of fairness inherent to these mapping approaches.

As Potschin and Haines-Young (2011: p. 576) argue, the transdisciplinary challenge of bridging societal and ecological knowledge forms cannot be met by "uncritical puzzle solving". Disassociating social processes from the management of public goods by compartmentalizing them under individual self-interest is analytically imprecise regardless of the methodological sophistication of the spatial analysis. Consequently, transient and shifting micro-scale relationships among actors which create spatially variable patterns of access and overexploitation of natural resources are ignored (Lele, 2013). Furthermore, geographic targeting of payments according to economic efficiency or cost-effectiveness and the process of neoliberalisation of nature often go hand-in-hand. For example, aligning limited financial flows to surpass the opportunity costs of areas presumed to maximally deliver a valued ecosystem service directly improves the ease for which the valued service can be 'bought and sold' or commodified. The recovery of opportunity costs suggests that service providers' user rights are essentially being appropriated. Once use rights are appropriated through compensation, they are considered as collateral in the exchange, resulting in a form of privatisation.

Alix-Garcia et al. (2008) was among the first empirical investigations of PES optimisation, examining cost-effective payment targeting in comparison to egalitarian flat-rate payments for PES to maintain common property forests in Mexico. Elsewhere, Gauvin et al. (2011) found that China's 'Grain for Green' PES was being targeted for parcels of land of high environmental vulnerability and low opportunity cost for land managers, but not the poverty levels of service providers. However, the authors did not provide an 'evaluative space' to judge poverty through a multidimensional interpretation of well-being. By assuming an increase in income through cash and grain payments as a proxy for well-being improvement and as the motivational stimulus for behavioural change, other deprivations related to human dignity, empowerment and ability to control one's destiny risk being ignored. The authors of both of these studies acknowledge the trade-offs that targeting payments according to efficiency or effectiveness might have on numerous and often poor smallholders or households without land-use rights. However, they purport to address this inequity problem by combining site selection based on service protection with a uni-dimensional conceptualization of poverty isolated from existing political realities and economic drivers of land-use. The ramifications of this simplification for targeting may lead to gross inequities that reinforce the poverty-environmental degradation cycle by displacing patterns of degradation across the landscape (i.e. 'leakage') (McAfee, 2012). Moreover, the decision to base compensation according to opportunity cost is a far from unbiased and objective standard. It is itself a political decision to reduce the suite of human value ethics to a single one: *Homo economicus*. Such a perspective is confirmed in the following statement: "…because we are paying exactly the value of the alternative use for the hectares of land they wish to deforest, ejidos[3]will always accept the contract" (Alix-Garcia et al., 2008: 380).

Targeting PES schemes based on cost-efficiency often reduce ecological processes as being precise relationships that align consistently and uniformly with patterns of property boundaries. While such relationships may be possible in the case of well-defined point-source pollution in relatively small geographical areas, these situations represent the minority of PES contexts (Kosoy and Corbera, 2010). The logic of optimisation also implies that complex ecological processes spanning large spatial and temporal scales somehow result in ecosystem services traceable to individual actions on small geographical areas over a short timescale. Indeed, highly

[3] Mexican villages which hold forest or grazing lands in common property

simplified relationships between the mere presence of forests and resulting hydrological benefits have become entrenched in our collective psyches that national environmental policies are being constructed according to unverifiable 'truisms' of nature (Calder, 2005). In reality, relationships between ecosystem processes or functioning and the delivery of ecosystem services that yield direct benefits from nature for human well-being are non-uniformly distributed in space and time. Furthermore, they are subject to dynamic pressures in both the anthropocentric realm of the institutional economy, the biophysical realm such as microscale topography and species diversity, and the evolving combination of these factors at the macroscale (Norgaard, 2010).

4.4. Case Study

4.4.1. Study Site

The Sundarijal catchment covers 15.76km^2 within the eastern edge of Shivapuri Nagarjun National Park (SNNP) and is a critical watershed supplying the larger Bagmati River watershed, contributing approximately one-third of the total piped water entering the Kathmandu Valley. The case of SNNP represents a classic dilemma of conservation at the cost of local well-being, since households within the park do not have legal access or rights to manage forest resources extending beyond the boundaries of private land deeds. Since the park was gazetted in 2002, households within the park have essentially been 'fenced in', breaking traditional productive landscapes of forest use and subsistence agriculture with the necessity to obtain competitive livelihoods under a context of restricted mobility and limited market access.

The hydrological services of the catchment largely accrue to hydropower generation, irrigation for rural agriculture in and around Kathmandu Valley, and urban consumption of drinking water by city residents. In addition to downstream hydrological benefits, over 400 households from three villages located within the park benefit from resources such as timber, fuelwood, and fodder to fulfil basic needs ranging from shelter, energy for cooking and heating, and food sovereignty. The villages located within SNNP and identified as upstream service provider communities for PES include Mulkharka, Chilaunegaun and Okhereni (Figure 4.1). These three

villages form a part of Sundarijal village development council (VDC)[4]. With few alternatives to utilise surrounding forest resources combined with growing populations, the majority of households have begun cultivating millet for the production of an illegal liquour as a means of livelihood. While the collection of fuelwood is necessary for cooking and heating in many parts of rural Nepal, the production of the liquour for commercial sale requires a steady energy source, forcing villagers to increase fuelwood collection. In 2008, a report by the World Conservation Union (IUCN) determined that co-managed forest and agricultural landscapes were the most cost-effective land management strategy over expansion of agricultural areas or even strict forest protection and resettlement of local villages in the Park (Karn, 2008). As such, they

[4] The municipal level jurisdiction of central government in Nepal; each VDC is composed of wards, the number of which depends on population size.

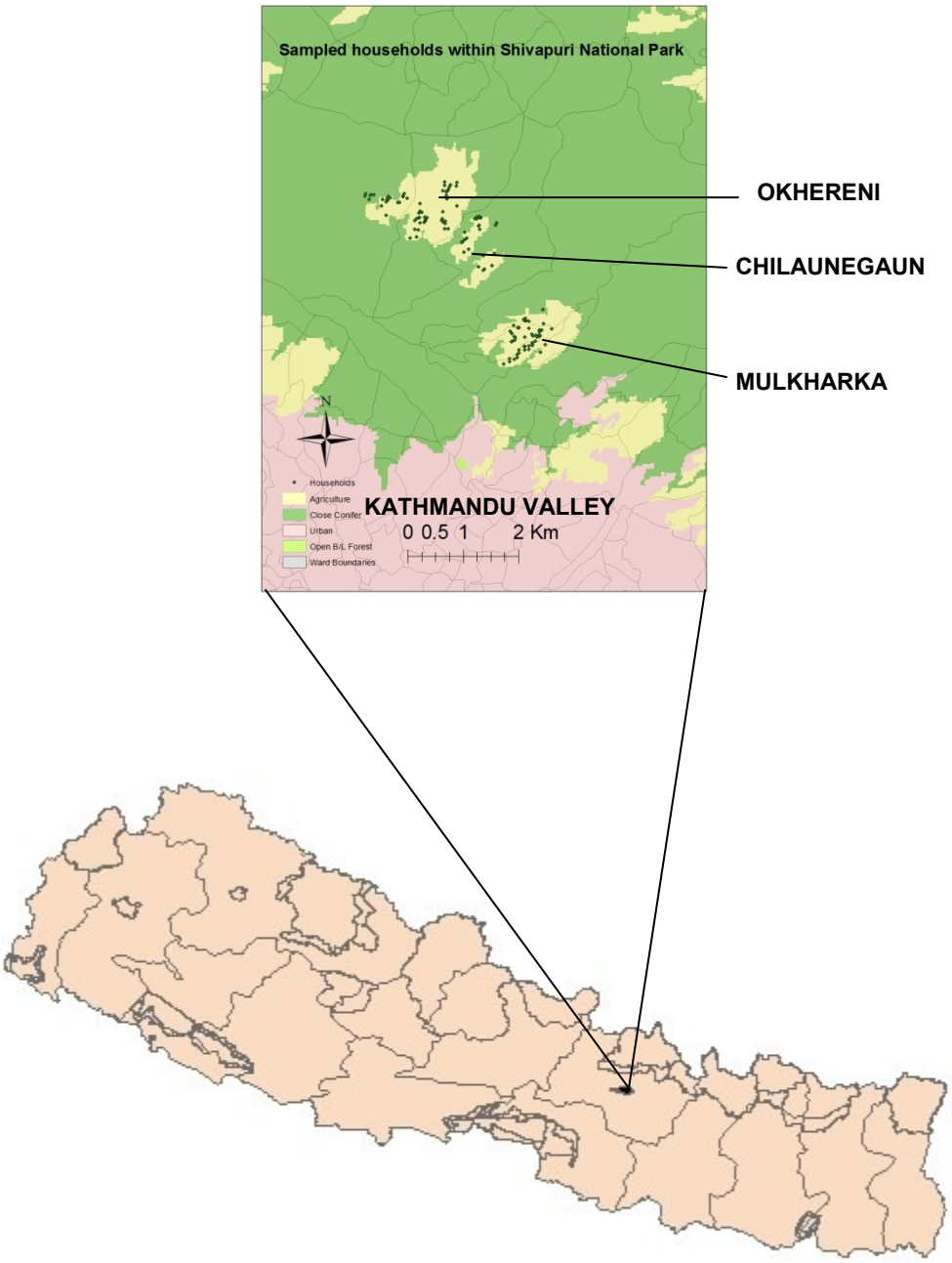


Figure 4.1: Location of study site. The insert depicts the location of sampled households and the three villages identified as potential PES service providers within the Sundarijal catchment of SNNP.

proposed a PES mechanism which would offer park settlements a role in land-use stewardship for sustained hydrological benefits (primarily water quality improvements) to downstream beneficiaries in return for payments that would enhance local development potential.[5] Specifically, households living in the park would be compensated for changing certain land-use practices, namely livestock grazing, use of chemical fertilisers, and human waste disposal, in exchange for new skills, renewable energy sources, eco-sanitary toilets and recovery of opportunity costs. While hydrological studies linking upstream land-use activities to water quality levels in the lower Sundarijal reservoir are not available, downstream beneficiaries and intermediaries claim that current land-use practices are increasing sedimentation in the reservoir and fouling water quality. By altering current land-use practices to activities more benign to surface and groundwater pollution, it is believed water quality in the reservoir will improve.

Within the park settlements, there exists a set of informal social institutions governing collective activity known as *guthi*. These are essentially social norms regarding collective management of private land that takes on a combination of religious as well as altruistic aspects of labour and financial allocation. Regular meetings are held calling for neighbouring farmers to offer support to meet labour deficits of households in tilling, seeding, and harvesting primarily millet and maize. Decisions are also made on communal work days in which men and women decide to forego household activities for the day in order to repair damages to common resources such as the path leading to the villages from Sundarijal town, or avoiding soil loss and other damage associated with landslides. Financial contributions are collected into a fund which is used to invest in tools or resources needed to undertake repair work. Through this fund, the *guthi* finances the resource and labour requirements for marriages, funerals and village festivals. The system represents a well-functioning mechanism for supplementing household-level labour and family necessities with the common interests in maintaining resources of collective benefit such as soil, road, pest management, and spiritual services. Membership to specific *guthi* associations reflects histories of inter-marriage and kinship rather than political boundaries or the present spatial proximity of particular households. The endogenously formed social preferences and

[5] As of 2014, the IUCN and other intergovernmental and non-governmental organisations that have pushed for PES implementation in SNNP have retracted interest due to claims by downstream stakeholders that they are powerless to mobilise resources or engage in further discussion without clear legal definition of the PES mechanism at ministerial level. Thus, despite the failure of the PES proposal in the Valley, we aim to critically analyse external 'targeting' of PES as was envisioned for the Sundarijal catchment and which continues to be promoted elsewhere.

associated institutions in the form of *guthi* relations suggest that land-use behaviour in these villages has been built upon cooperation and reciprocity rather than according to purely isolated household decisions.

4.4.2. Descriptive Statistics

Table 4.1 provides a description of household income, environmental effectiveness and well-being characteristics of the three upstream villages in order to illustrate how targeting objectives differ broadly across villages. It can be seen that the heterogeneity of opportunity cost and net income is substantial, given the high standard deviations of the averages. While the existence of such heterogeneity may appear conducive to the spatial targeting of payments, there is little understanding of how existing social processes or individual household constraints explain this heterogeneity. For example, household capabilities to engage in income-generating activities as well as customary or cultural land-use patterns are not reflected in these opportunity costs. The variability of wealth, livelihood opportunities and well-being perspectives indicates that collective resource management potential may also vary between the villages. In terms of social capital, the strength of guthi relations was more evident in Chilaunegaun (and to a lesser extent Okhereni) where collective meetings were held fortnightly in comparison to Mulkharka where such meetings were held haphazardly or coinciding only with major festivals.

Additionally, ethnic diversity was greater in Mulkharka with just over 20% of the sampled population belonging to ethnicities other than the dominant Tamang (compared to ethnic heterogeneity of less than 10% in Chilaunegaun and Okhereni). While there does not exist evidence to suggest that ethnicity might influence preference for payment distribution, villagers interviewed agreed that ethnicity influenced established guthi memberships and other aspects of community life. Finally, limited economic opportunity was recognised as the primary obstacle to improving collective well-being in Mulkharka while mobility was more of a burden for the farther villages. As a result, potential distributional risks of prioritising certain households over others are ignored, raising concerns over the legitimacy of externally defining how payments should be targeted. Given the heterogeneity and dynamic nature of socio-economic, well-being and cultural characteristics within these villages, any interaction of targeted objectives for

Income and Land Area	**Description**	**MULKHARKA**		**CHILAUNEGAUN**		**OKHERENI**	
		Mean	**(SD)**	**Mean**	**(SD)**	**Mean**	**(SD)**
	Opportunity cost of liquour production	USD 4878.9 / ha (7116.9)		USD 5315.8 / ha (8307.7)		USD 8571.1 / ha (9298.5)	
	Land area under production[a]	0.13 ha	(0.11)	0.18ha	(0.11)	0.13ha	(0.09)
	Average net income per household / year	USD 2607.5 (2666.3)		USD 2171.4 (1332.7)		USD 1623.8 (1042.5)	
	Percentage below 'income' poverty line	33.3%		25.0%		25.0%	
Environmental Effectiveness	Open Defecation	52.9%		25%		25%	
	Bhari of fuelwood collected per household / year[b]	171	(137.8)	228	(148.8)	236.4	(94.7)
	Bhari of livestock fodder collected per household / year[b]	251.3	(306.2)	447.0	(333.1)	461.4	(504.7)
	Stated soil loss problems ('always')	23.5%		50.0%		60.0%	
	Households using 100kg or more of urea fertilizer / year	33.3%		37.5%		35%	
Well-being	**Description**	**MULKHARKA**		**CHILAUNEGAUN**		**OKHERENI**	
		Mean	**(SD)**	**Mean**	**(SD)**	**Mean**	**(SD)**
	1=Not having enough to eat during the year	0.13	(0.05)	0.50	(0.05)	0.35	(0.05)
	1=Lack of mobility	0.76	(0.04)	0.88	(0.04)	0.95	(0.02)
	1=Disability	0.71	(0.05)	0.75	(0.05)	0.40	(0.05)
	Adult Literacy: ≤ 50%	47.1%		50.0%		50.0%	
	Children not attending school	5.9%		25.0%		45.0%	
	1=Water enters roof	0.25	(0.04)	0.38	(0.05)	0.20	(0.04)
	Sickness in preceding month	0.35	(0.05)	0.75	(0.05)	0.45	(0.05)
	1=Perception of lacking equal opportunities	0.44	(0.05)	0.25	(0.05)	0.50	(0.05)
	Consumer durable assets: 2 or less	35.3%		75.0%		55.0%	
	Poverty Score (max: 9)[c]	4.25	(1.2)	4.0	(1.1)	5.2	(1.2)
Social perceptions of obstacles to well-being	**Description**	**MULKHARKA**		**CHILAUNEGAUN**		**OKHERENI**	
	Household-level obstacles to well-being	Physical house improvement (24%)		Physical house improvement (37.5%)		Physical house improvement (36.2%)	
	Community-level obstacles to well-being[d]	Economic Opportunity (63.3%)		Mobility (83.3%)		Mobility (65.0%)	

Table 4.1: Descriptive statistics characterizing livelihoods, land-use management, perceptions of well-being, and social capital among the villages within SNNP.

[1] Land-users in the mid-hills of Nepal measure land area as 'mato muri'. 1 ha = 76 mato muri.
[b] High standard deviations for fuel and fodder use reflect differential rates of alcohol production, family member size and possession of livestock
[c]Average poverty score across villages is 4.48 (standard deviation: 0.63)
[d] Statistically significant differences in perceived barriers between Mulkharka (closer community to Kathmandu) and Chilaunegaun and Okhereni (farther communities from Kathmandu) (Fisher's Exact Test: $p < 0.01$)

payment design requires spatially explicit recognition and social deliberation of its distributional consequences.

4.4.3. Methodology

In order to illustrate trade-offs in spatially prioritising PES objectives for upstream service provision with local institutional and political realities, data collection requirements were substantial. Specifically, data were derived from potential service providers regarding opportunity costs of land-use management, perceptions of soil and forest management, formal and informal governance institutions, perceptions of household and village well-being as well as preferences for payment type and distribution. A total of 135 households, or one-third of the total number of households (i.e. 405), were selected through a stratified random sampling procedure according to the division of wards following Uchida et al. (2009). The identification of households was carried out by pooling all the households within a single ward and randomly selecting one-third for the interviews. Semi-structured interviews with an adult member of each selected household were conducted. Due to the substantial data requirements, interviews were split into three surveys: 1) opportunity cost; 2) land-use management, and 3) household well-being with questions pertaining to PES design and preference common to each survey. The selected households were randomly apportioned to one of the three sets of surveys. While we recognise the limitations of this approach in reducing the overall data sets for spatial interpolation, we were also aware that households within SNNP were being selected for other government-related research taking place at the same time regarding perceptions of buffer zone management. Given the considerable data needed for opportunity cost collection and understanding household perceptions of well-being, we had reason to believe that questionnaire fatigue and possible strategic response bias could affect the data collected. Accordingly, each

survey contained data from 45 households selected across the three villages. The geographic coordinates of each sampled household were taken in order to carry out spatial targeting analysis of opportunity cost distribution, environmental effectiveness and poverty classifications using ArcGIS.

Opportunity costs were calculated for the most profitable land-use activity[6] which was identified as liquour production. Following Richards et al. (2003), opportunity cost was identified through the following equation:

$$VAx_i = ((Ix_i / Ax_i) - (Cx_i / Ax_i) \qquad (4.1)$$

where the added value or rent of a particular product (*VA*) was calculated by the total income (*I*) minus the costs (*C*) divided by the total area in hectares (*A*) under production by product x_i. Costs include expenditures on fertilizer, trips to Kathmandu to sell the product, hired labour as well as the value of household labour measured through the minimum wage for agricultural activity in Nepal[7].

In addition to upstream communities, downstream beneficiaries of improved water quality from the catchment as well as intermediaries involved in promoting a PES scheme for the catchment were interviewed. The aim of the interviews was to specifically identify which management efforts would lead to the most socially equitable, environmentally effective and economically successful outcomes in improving water quality in the reservoir. Perceptions of the feasibility of a proposed PES mechanism were also elicited as well as the extent to which PES incentives could achieve management objectives for water quality improvement. According to the feasibility studies as well as interviews with these key informants, three main threats to water quality were identified and traced to the land-use activities of the villages within the Park (Niraula, Unpublished results, 2008; Karn, 2008). These included: a) the lack of human waste containment and open defecation to waterways; b) deforestation for energy needs as well as the production of liquour, and c) the widespread use of chemical fertilizer contributing to perceived

[6] While there was reason to indicate that the sale of cannabis provided significant returns, the sensitive nature of questioning made it impossible to determine opportunity cost data for cannabis production.
[7] As of 2008, this value was NPR 18.75 (USD 0.22) per hour (Nepal Gazette, 2008)

nutrient enrichment of waterways. A fourth threat impacting water quality was haphazard illegal road or footpath construction and the associated risk of land subsidence that follows heavy rainfall events. This threat is not attributable to specific households, but is considered an overall threat which enhances sedimentation for the Sundarijal reservoir and hampers hydroelectricity production. In order to target households contributing most to these threats[8], we identified which households are:

- practicing open defecation (determined by stated use of a private or shared toilet);
- cultivating land within 1 metre of a water body;
- collecting more than the average bhari[9] of fuelwood per year (206 bhari);
- collecting more than the average bhari of livestock fodder per year (379 bhari);
- claiming frequent soil loss problems;
- using more than the household average of 100kg of urea fertilizer per year

Poverty was classified both as: a) net income and b) well-being measured through a series of variables reflecting health, literacy, equality, physical assets, and food security. Data for poverty by net income was derived from households who were randomly selected to assess livelihood information and was classified according to the central government determination of the national poverty line.[10] However, a multi-dimensional conceptualization of poverty, on the basis that poverty is an experience that can be present in multiple dimensions at multiple points of time for a given household, underpins the rationale for expanding consideration beyond income effects (Alkire and Santos, 2010). This data was determined from households surveyed for household

[8] Land holding size was not considered, since on average households utilise only 10.5% of their total land holding for cultivation and habitation. Household interviews indicated that the remaining land holding is left fallow or abandoned due to limited labour or material resources.
[9] 'bhari' is a local unit of measuring firewood and fodder. 1 bhari = 30kg
[10] National poverty line was taken from the Third Nepal Living Standard Survey (2011) by the Central Bureau of Statistics (CBS). The survey used a food equivalent of 2,200 calorie consumption per person/day as well as access to essential non-food items as an index to measure poverty. According to market prices from 2011, it was determined that a person requires a net income of NRs 14,430 per year to meet essential food and non-food requirements (CBS, 2011).

well-being. Thus, in measuring poverty as well-being, we attached a score of one point for each household that satisfied each of the following queries:

1. not having enough to eat at some point in the year;
2. having no access to vehicles that offered mobility (e.g. bicycle, motorcycle);
3. having a mental or physical disability in the household;
4. the percentage of adults capable of reading and writing is less than 50%;
5. one or more school-age children are not attending school;
6. water enters through the roof during rain events;
7. one more member of the household was sick in the last month;
8. having the perception of lacking equal opportunities to acquire vocations or other skills desired, and
9. the number of household assets, measured as 5 consumer durables variably possessed by households (TV, mobile phone, computer, radio, satellite dish), was limited to two or less

The maximum possible 'poverty score' is 9, with each query weighted equally[11]. The choice of categories reflects a tentative set of measurable indicators consistent with a holistic perspective of poverty comprising multiple dimensions of human well-being (Alkire and Santos, 2010).

An ordinary Kriging interpolation model was used within ArcGIS to identify poor areas according to each classification (poverty targeting). This procedure was also used to interpolate opportunity costs of producing liquour (to spatially assess areas for cost-efficiency targeting) and the distribution of ecologically vulnerable areas ('additionality' targeting). The completed interpolations were then analysed to spatially determine areas where opportunity costs of producing liquour were lowest as well as where ecologically vulnerable areas were highest (cost-effectiveness targeting). The identification of these areas was carried out by grouping interpolated values into three classes using Jenk's Natural Breaks, which has been identified as a reliable method to optimize breaks between classes through the minimization of the sum of

[11] While obtaining a 'poverty point' was made through binary classifications for most of the categories (fulfilling the query or not), the choice of 'poverty' scoring for the number of assets possessed as well as adult literacy was made in order to clearly distinguish differences between households for these categories.

squares error (Swetnam et al., 2011). Areas of higher cost-effectiveness were identified where areas of low opportunity cost and high ecological threat overlap. Where areas of higher poverty (measured respectively as income effects and state of well-being) overlap with cost-effective areas, presumably 'triple win' targets can be spatially identified.

In assessing whether discrete PES objectives of cost-efficiency, effectiveness or even poverty alleviation correspond to local institutions, political realities or local perceptions, two specific analyses were carried out. The first was to assess the distributional implications of identifying 'efficient' service providers by means of opportunity cost for land-use change. This was conducted through a factorial ANOVA in analyzing whether the proportion of net income allocated to different land-use activities between households varied with annual household income. The second analysis involved a series of focus groups with upstream households in each village and interviews with key informants representing service beneficiaries and intermediaries for the proposed PES. The resulting information was also used both to triangulate the data emerging from a household perspective and to assess whether group dynamics or social perceptions towards well-being, deprivations, and PES perspectives offered additional insight. Household preference for payment type and distribution was determined and analysed using a Fisher's Exact Test to assess whether there were significant differences in payment type and management preferences between villages. The focus groups were conducted in each village according to gender (i.e. 6 focus groups for the 3 villages). An additional focus group was conducted in Okhereni to consider the views of two separate ethnic groups, Tamang and Chhetri, residing in two informally named villages within the same ward. Those selected to attend the focus groups were the same households interviewed. This was done to build rapport from the previous encounter and to encourage neighbours to also attend. Each focus group had between 10 and 20 individuals and was moderated by the lead author and four additional assistants and native Nepali speakers from the environmental NGOs advocating for PES. The resulting data were analysed for common themes exposing perceived obstacles to well-being, informal institutions, and preferences for alternative management strategies for SNNP including the role of PES.

4.5. Results

4.5.1. Targeting matrix

Figure 4.2 incorporates collected data on opportunity cost, land-use practices, net income and perceptions of well-being to create a spatially explicit targeting typology for identifying the most feasible areas for PES implementation in the Sundarijal catchment. The boundary layer depicted represents the area of each of the three human settlements within SNNP as replicated from Figure 4.1. The Ordinary Kriging interpolation model was based on the smoothest presentation of variance between household waypoints.

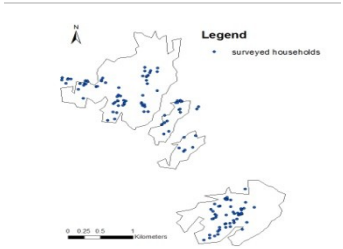
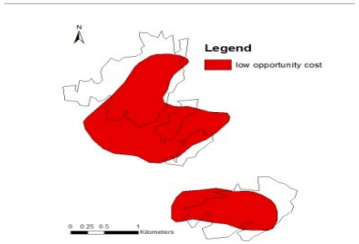
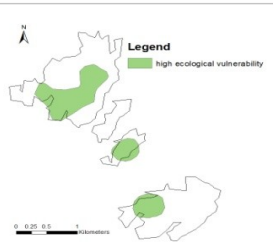
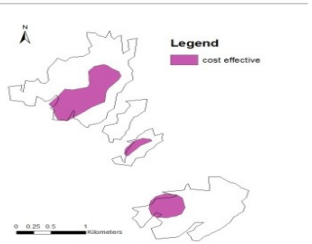
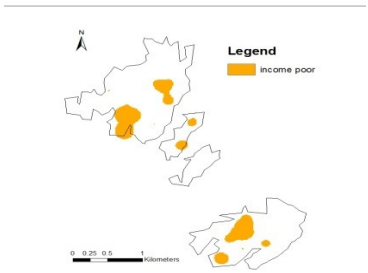
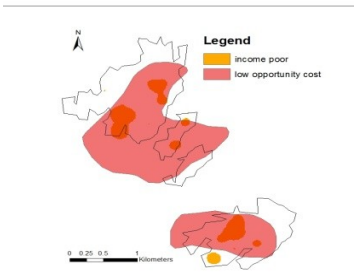
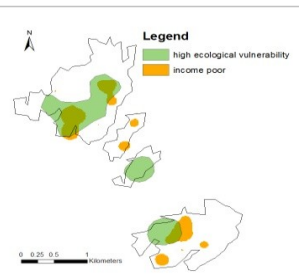
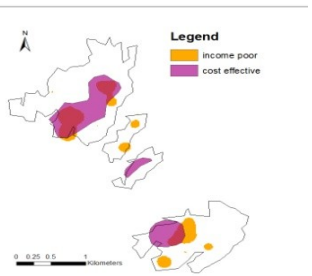
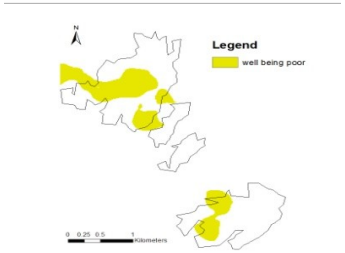
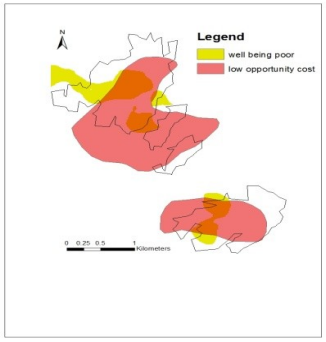
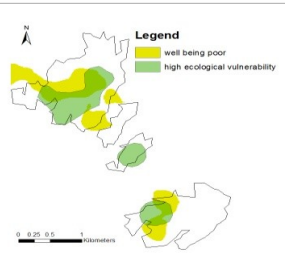
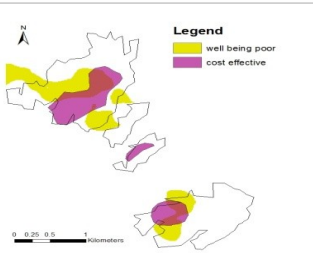
Targeting typology	Poverty Consideration Only	Cost Efficiency	Ecological Vulnerability (areas of ecological additionality)	Cost-effectiveness (greatest ecological additionality in areas of lowest cost of compensation)
No consideration of poverty				
Poverty (below national poverty rate)				
Poverty (categories of well-being)				

Figure 4.2: Geo-spatial visualisation of a typology of targeting scenarios for PES in the Sundarijal catchment, according to the objectives of economic efficiency, ecological vulnerability and two considerations of poverty alleviation. The first row of the matrix assumes the position of status quo in relation to poverty alleviation while the second and third rows adopt the two perspectives of poverty targeting. The first image in the first row does not apply a targeting scenario but simply illustrates the location of sampled households within the three villages.

The accuracy of the interpolation was assessed by examining the semivariogram of the model which indicated low variance of responses between neighbouring households and increasing variance the farther away households were located from each other. This provides a reasonable estimate that geographically delineated 'targets' of low-cost, ecologically effective, higher poverty and the combination of these classifications could be identified.

Beginning with the first row (e.g. 'no consideration of poverty'), it can be seen that the most inclusive coverage of the settlements is considered where cost-efficiency is optimized. It should be noted that households on northern and western fringes of Okhereni village are not included within the targeted zone because compensation would be considered overly high due to above average opportunity costs in producing liquour. Yet, it is these households who are the most isolated within the National Park and have argued for greater access to new markets to improve their well-being. The minimal change in area between cost-effective and ecological vulnerable areas illustrates a possible correlation between soil productivity for growing millet and the net rent of profit that emerges from the land indicating that wiser soil practices in these areas may improve productivity of the land. However, the overall targeted area within each community diminishes under these targeting strategies.

Moving on to the second row of Figure 4.2, with the consideration of poverty measured as lower than the national poverty rate in terms of income, the targeted area under each additional consideration of efficiency and effectiveness becomes smaller. While two small impoverished zones are present in Chilaunegaun and have low opportunity costs for millet production, these areas are not considered ecologically vulnerable and hence do not fall within the 'triple win' zone depicted as cost-effective and 'income-poor'. The more inclusive consideration of poverty as well-being in the third row expands the area under consideration as impoverished. Households

located within these areas scored a 'poverty' score (as described earlier) of 5 or greater. These zones shrink in size when other targeting objectives are incorporated. The 'gold standard' as coined by Gauvin et al. (2011) and shown as the bottom right-most image would appear to encompass the smallest number of households of any of the strategies.

4.5.2. The misleading targeting of 'efficiency': Linking livelihood choice with patterns of poverty

Figure 4.3 illustrates the proportion of total net income between different economic activities across all sampled households (from survey 1) as well as the poor and least poor of those sampled. The factorial ANOVA analysis indicates that the mean proportion of net income is significantly different both between different economic activities as well as the interaction between economic activity and degree of poverty. For the poorest households, the proportion of livelihood that comes from the sale of cannabis and the sale of subsistence food crops (e.g. rice, maize, wheat, potatoes, radish) increases in importance; whereas for the least income-poor households, the sale of liquour, livestock husbandry and off-farm income such as commercial, tourism activities and overseas remittance represents a greater proportion of livelihoods. While the spatial targeting of cost-efficient payments compensating liquour production corresponds to poorer households, such payments will clearly not be adequate in compensating opportunity costs as liquour production represents only 5.5% of total household income. Moreover, cost-efficient targeting will not be effective in reducing pressure on forest resources when more well-off households (from an income perspective) are responsible for the greatest pressure on forest resource degradation and potential impacts to water quality downstream.

4.5.3. Perspectives of social actors

Table 4.2 offers a number of perspectives from key social actors who represent the beneficiaries of the proposed PES in the Sundarijal catchment. The viewpoints of service beneficiaries (e.g. KUKL, ADB and NEA) highlight the emphasis on targeted payments in identifying where and in what form land-use change should be modified so as to enhance the likelihood for improvements in water quality. While there is agreement between both beneficiaries and promoters of the proposed PES scheme that livelihoods should shift to align with emerging markets, there is less

unanimous support for the consideration of distributional equity of payments. This can be identified in the diverging views of the High Powered Commission, the Chief Warden of SNNP and KUKL). While the views of the Chief Warden, VDC government, and the ADB seem to suggest that in-park villagers should ultimately determine how their needs are to be met, the majority of service beneficiaries (e.g. NEA, KUKL) and the High Powered Commission place greater emphasis on ecological and economic outcomes over the needs and perspectives of upstream communities.

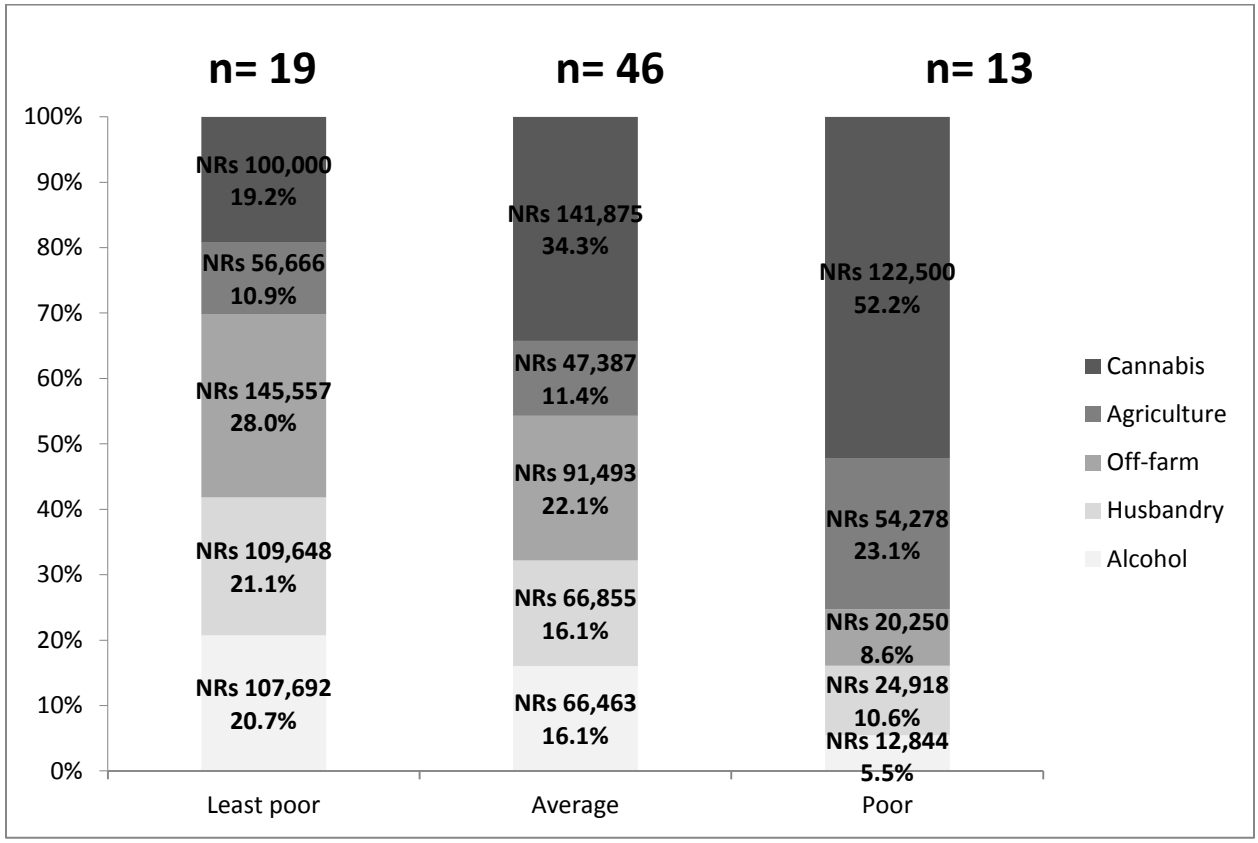


Figure 4.3: The distribution of economic activities and net income per year of randomly sampled households from Survey 1. Least poor households are classified as those with the top third net annual income of the total households sampled. Poor households are classified according to the national poverty line of Nepal. A factorial ANOVA model indicates significance of mean proportions between economic activities ($p < 0.01$) as well as from the interaction of economic activity and poverty class ($p < 0.05$). There was no significance in mean proportions between poverty classes.

Table 4.3 illustrates a social counterpoint to spatial targeting of payments according to externally determined objectives. According to a gamut of efficiency, effectiveness, and equity-oriented PES designs adapted from Pascual et al. (2010), preferences between villages for payment type and distribution are shown. In communicating payment designs to households, we emphasized that payments would be given in cash in relation to opportunity costs for 'compensation' and 'environmental additionality' designs, and based on a fixed overall budget for the 'egalitarian' design (e.g. equal distribution of payments across all households).

Stakeholder	*Equity targeting*	*Environmental targeting*	*Economic efficiency targeting*
Sundarijal VDC (I)	"Land-use decisions must come from villagers themselves, according to their understanding."	"Waste management and avoided deforestation are key"	"Payment should be greater than benefits from alcohol production to spur economic activity"
High Powered Commission for Integrated Development of Bagmati River Civilisation (I)	"Stakeholders downstream must be consulted before deciding how payments will be distributed."	"Improving sanitation and reducing forest degradation are most important"	"Generating alternative livelihoods that involve forest protection for these people. Loans can stimulate local entrepreneurship."
Asian Development Bank (I; B)	"Surely payments should reflect what communities require in terms of basic needs"	"Payments should be specifically targeted to particular forest conservation and sanitation activities that directly promote watershed protection"	Providing alternative energy options reducing pressure on the forest as well as livelihood training in new skills can shift the local economy
Kathmandu Upatyaka Khanepani Limited (KUKL) (B)	"Best solution is to compensate according to their terms for relocation. But if not feasible, a communal payment might work"	"Targeting deforestation requires shifting them [villagers] away from alcohol production"	"Demand for flower farming is high and this can be an alternative economic activity for them."
Chief Warden-SNNP (I)	This is a very delicate issue and exceedingly important. If locals are not satisfied with the outcome, at the household and user group level, it must be scrapped	Minimal damage on forest resources should be key in any programme design.	"The area is very viable for domestic and international tourism.
Nepal Electricity Authority (NEA) (B)	No comment	"We need to identify the specific areas where siltation and debris input to waterways is most extreme and target these areas first."	A portion of royalty can and has been going for local infrastructure development.

Table 4.2: Perspectives of key social actors downstream implicated as beneficiaries (B) or intermediaries (I) towards targeting payments to achieve social, ecological and/or economic objectives. The perspectives provided here were obtained from individuals whose viewpoints do not necessarily reflect the institution to which they belong.

Community: Any Payment Type[a]	**No Payments**[b]	**Individual Payments**[b]	**Communal Payments**[b]	**Of communal: VDC / LUG**
Mulkharka (n=51)	6 (12%)	18 (35%)	27 (53%)	8 / 53
Chilaunegaun (n=24)	0 (0%)	6 (25%)	18 (75%)	0 / 67
Okhereni (n=60)	2 (3%)	3 (5%)	55 (92%)	17 / 68
Community: Of Individual Payments Only[c]	**Cost-Effectiveness or Compensation**	**Environmental Additionality**	**Household Deprivations**	**Egalitarian Payments**
Mulkharka	6 (33%)	4 (22%)	3 (17%)	5 (28%)
Chilaunegaun	0 (0%)	0 (0%)	1 (17%)	5 (83%)
Okhereni	0 (0%)	3 (100%)	0 (0%)	0 (0%)

Table 4.3: Preferences towards individual and communal payments between the three villages.

[a]Statistically significant differences in payment preferences between the three communities (Fisher's Exact Test: $p < 0.05$). [b]Statistically significant differences in payment preferences between communal and individual payments (Fisher's Exact Test: $p < 0.01$) as well as communal and preference for no payments (Fisher's Exact Test: $p < 0.01$). [c]Statistically significant differences between villages in terms of preference of individual payment (Fisher's Exact Test: $p < 0.05$) with Chilaunegaun and Okhereni preferring Egalitarian and Additionality targeted payments respectively in relation to the more even distribution of individual payment preference in Mulkharka.

Furthermore, we emphasized that 'household deprivation' based payments would reflect the elucidation of collective and household needs respectively and would thus be delivered as either cash or in-kind rewards. Overall, payment preference was significantly in favour of communal rather than individual payments; with particular emphasis on local user group management of the fund rather than control by local government (VDC) [12] or SNNP authorities.

4.5.4. Determining the legitimacy of PES targeting scenarios

From household interviews and focus group discussions in the SNNP villages, it was suggested that payments should form a combination of individual incentives to improve skills sets and deprivations afflicting individuals as well as village-level benefits for collective activities that

[12]Preference for communal payments determined and managed by local user groups (LUG) was significantly different from preference for any other payment type, regardless of whether payments were individual or communally provided by the VDC (Fisher's Exact Test: $p < 0.01$).

improve soil quality, forest status and waste management. Indeed, the actual land-use activities of households are a function of a variety of capabilities in terms of intra-household dynamics and biophysical opportunities to manage land in particular ways. Moreover, we find a divergence between the degree of trust residing in formal institutions at the local level and more informal institutions built along existing social norms for collective decision-making. As the following quotations from focus groups recognize, payment distribution should be based on local knowledge of environmental threats with the support of external knowledge as well as collective decision-making on payment investment rather than what may be seen as the arbitrary decisions of outsiders whose objectives have not acquired local legitimacy or trust.

> *"We should choose representatives in our village who will look after our interests and would decide through village meetings who would receive payments for what purpose and where the whole village can benefit."* (Women focus group: Khatri Tole, Okhereni)

> *"They [Sundarijal VDC] have not delivered benefits to our villages in relation to the resources they receive from central government. There is too much corruption, so whatever payment is allocated through the VDC, local user groups should have greater involvement in discussions with experts over payment choice and distribution."* (Male focus group: Chilaunegaun)

The latter quotation reinforces the findings of Kerr et al. (2012) in which the reception to individual or collective PES payments to villages in Mexico was contingent on levels of trust between households and local leaders. A similar plea for self-determination through community deliberation and elected representatives for decision-making and payment allocation was evident in Chiapas, Mexico with regards to carbon payments (Corbera et al., 2007) Moreover, emphasis on the collective determination of payment choice and administration was a greater determinant of potential participation in the SNNP villages than whether payments should be individual or collective, in cash or in-kind, or whether they addressed targeted efficiency, effectiveness and development goals. From the interviews, it was determined that dissatisfaction with SNNP authorities was widespread due to the perception that villagers were unable to enhance their

mobility or obtain the necessary forest resources they need for survival. Security measures were also identified as lacking within SNNP and perceived to be the responsibility of the VDC and SNNP authorities.

4.6. Discussion

At this point, it is necessary to emphasize the analytical rather than methodological support that the targeting matrix provides in elucidating our arguments by clearly acknowledging four response errors. These errors reflect: a) information given by survey respondents rather than attempting to model water quality impacts associated with faecal coliform, eutrophication and soil erosion at the land-parcel level; b) the subjective choice of delimiters for poverty and environmental effectiveness classification as averages, binary responses, or simply to distinguish differences between households; c) the assumption of linear ecological impacts associated with calculating environmental effectiveness, and d) interpolating household level data to spatially visualize discrete polygons of the geographical area where PES could be specifically optimized. However, we argue that collecting more sophisticated measurements of dynamic ecological flows entails substantial costs which are unlikely to be met (Norgaard, 2010). Furthermore, the measurement of poverty is itself a value judgement, which we have attempted to expand upon rather than narrowly confine to income effects. Delimiters were chosen expressly to illustrate variations between households and can very easily be modified to reflect socially deliberated indicators rather than to impose an arbitrary ordering of the state of the human condition. These response errors mirror those of previous studies that have sought to optimize PES design (Jack et al., 2009; Gauvin et al., 2010; Wünscher et al., 2006, Narloch et al., 2011). Indeed, the crude nature of our own targeting strategy serves to reinforce the argument of this paper that it is impossible to ignore uncertainty in optimizing payments by conveniently defining boundaries devoid of political context and assumptions of static ecological relationships.

4.6.1. The power behind the 'targeting' discourse of PES

In examining the targeting typology presented in Figure 4.2, it is clear that trade-offs between the objectives are inherent in the choice of targeting strategy. These trade-offs between

effectiveness, efficiency, cost-effectiveness and poverty alleviation are however only the tip of the iceberg as geographic patterns of well-being deprivation and / or land-use practices are often intertwined with household capabilities which themselves are linked to social processes between neighbouring households. For example, the distribution of opportunity costs for liquour production between households is influenced by family size (e.g. the larger the size, the less hired labour required) as well as caste divisions within the Tamang ethnicity (e.g. the Lamas or priest caste are not willing to sell alcohol on religious grounds). Optimizing payments according to opportunity cost or environmental additionality raises the question of how such behaviour is determined not only by household population dynamics, but also land endowment and quality as well as individual capabilities to transport liquour, graze livestock, or collect fuelwood (Corbera et al., 2007). Keeping in mind micro-scale differences between households, the more inclusive the targeting strategy purports to be, the more constricted is the understanding of the underlying dynamics causing specific configurations of deprivation, profitability and ecological vulnerability at the household level.

Stepping back from the perspective of household capabilities, the implications of the targeting matrix from a distributional perspective warrant a closer examination. While the proportion of 'cost-efficient' areas is fairly uniform as mentioned above, different households stand to gain or lose from targeting specific areas. For example, if environmental additionality alone is prioritised in distributing conditional payments, households who are less well-off in the villages of Chilaunegaun and Okhereni will be excluded in the distribution of payments. This can be seen by examining the two targeting scenarios where the two interpretations of poverty alone are illustrated. Conversely, when considering the two scenarios in which areas of poverty and environmental additionality overlap, the targeted areas are reduced substantially hence influencing the efficacy of prioritising environmentally vulnerable areas. In this manner, a clear trade-off between the objectives of effectiveness and poverty alleviation is visualised. Attempts to minimise trade-offs by targeting overlapping objectives will increase the funding available to induce land-use change and improve incomes or perceptions of well-being in small isolated areas, but fall short of confronting why such social and ecological patterns occur across the landscape. Moreover, by atomising poverty (both well-being and income poverty) as isolated

from social norms and institutions, external targeting attempts are likely to exacerbate perceived inequity leading to new constellations of social and ecological vulnerabilities.

Of course, this logic is extolled by some hardliners of market-based PES theory who argue that PES should be about 'paying for what you want' rather than integrated approaches which consider social complexity and human well-being (Ferraro and Kiss, 2002; Engel et al., 2008; Wunder, 2008; Zhang and Pagiola, 2011). For these authors, PES is a technical process of matching nature with exchange value and associated financial flows clearly linked to the conservation objective at the lowest overall cost. Replacing trade-offs with 'win-win' solutions is a puzzle-solving process of merely rearranging financial incentives (Martin et al., 2013). Engagement with social actors is done on an individual basis under a strictly rational interpretation of costs and benefits as imposed by external actors. The perspectives of local people are either not prioritised or are simply dismissed as universal and subject to rational self-interest; hence manifesting a disparity of power between downstream proponents of PES and service providers upstream (Fisher, 2013).

In considering the power asymmetry between actors calling for targeted payments, it is necessary to reflect upon the 'framing' in which the proposed PES is being articulated. Far from an objective science of how environmental problems can be solved, there exists multiple ways of knowing, value sets and modes of governance for understanding, communicating, and approaching environmental problems (Sikor, 2013). It is pertinent that we understand how PES targeting privileges some values or modes of governance over others so as to identify how power differentials exist and how they may lead to injustice and unsustainable outcomes. The narrow economic focus of maximising ecological gains at the least monetary cost forms a part of a fundamental facet of the existing political economy. This refers to the expansion of markets into the environmental realm for the sake of capital accumulation (Kosoy and Corbera, 2010; Martin et al., 2013). The repercussions of this particular framing of eco-social relations serves to enhance control over resources by powerful actors downstream, glorifies self-interest over social norms for cooperation, and realigns priorities based fundamentally on economic priorities rather than ecological ones (Harvey, 2005).

Accordingly, sketching the contours of any externally determined payment distributions despite inclusive consideration of diverse equity and effectiveness objectives falls short of aligning with local self-determination. For example, providing payments to households located in the 'gold standard' targeting scenario will not compensate for histories of institutionalised racism against the Tamang ethnicity that cause variable patterns of well-being deprivation across households (Campbell, 1998). Such social reflections may explain why collective compensation was significantly preferred over individual payments in each of the three villages. It is interesting to note that preference over individual payments was significantly greater in the village of Mulkharka than the farther villages. It can be surmised that Mulkharka, with a greater divide between resource wealthy and resource poor households, associated perceptions of unequal opportunities, weaker social institutions, as well as preference for new economic opportunities might have influenced a different preference for individual payments than the more interior villages of SNNP.

4.6.2. Towards greater socialisation in the targeting process

Amidst the backdrop of mistrust and lack of consensus over common needs and priorities for natural resource management in the Sundarijal catchment, additional attempts to impose external objectives may exacerbate perceptions of inequity. This possibility may have particularly negative ramifications given a history of wider socio-political tensions in Nepal. Indeed, evidence has indicated that greater inequity amongst rural villages in the mid-hills led to an escalation of violence during the Maoist insurgency (Nepal et al., 2011). The implications of historical political economy in the region underscore the perils of poorly conceived or overly simplified conceptions of equity and the salience of legitimate and contextually crafted arrangements for land-use and natural resource management. Consequently, external perspectives of distribution efficiency and effectiveness contrast with local understandings of fairness, with such misalignment reinforced by differential power gradients between the sponsors of PES and identified service providers to the scheme (Van Hecken et al., 2012).

However, the use of incentives remains a powerful motivator of behavioural change (Fehr and Falk, 2002). We therefore make the argument that PES should return to its bare bones...as an

incentive-based negotiation. If such a negotiation is to play a role in addressing socio-ecological conflicts, it is necessary to question who proposes particular targeted objectives and what power or legitimacy such actors possess in pushing a particular agenda forward. The legitimate needs and values of negotiating parties should be articulated through continuous deliberation to bear upon what is an intrinsically political decision. We propose that PES should not be viewed merely as an instrument to uncritically influence land-use activity under any externally imposed value system. Rather, it should be oriented as a collective action exercise that emerges endogenously (or not) from reciprocal interactions over time in which the use of incentives (however negotiated) forms an integral component of sharing benefits in the management and access of natural resources (Kerr et al., 2014).

PES objectives should be malleable to existing and evolving social understandings and norms at various spatial and temporal scales. The form and condition that the payment takes must also be embedded within existing social norms appropriate for influencing individual behaviour. While our argument has been centred against the prioritisation of payments as a purely neoliberal exercise, there is nothing wrong with differentially distributing and allocating incentives across the landscape in order to satisfy both human well-being and long-term ecological health. Indeed, common pool resource regimes have successfully co-created their own natural resource management rules in this way resulting in resilient socio-ecological systems (Singh, 2013). The keyword here is ‘co-create’, in which the deliberation of evolving needs and diverse languages of valuation is a critical procedural component of distributional outcomes and resulting land-use practices. In this manner, a ‘social vetting’ process would result in vastly different outcomes than those ‘targeted’ under a neoliberal stance and would do so by placing justice and collective benefit as core components of resource distribution rules.

Given this understanding, there is room to experiment with amalgamating the more procedural justice-oriented features of collective resource management with the use of incentives to target particular objectives. An interesting avenue of research lies in using social multi-criteria evaluation to design a PES strategy according to a set of criteria reflecting the conflicting values and perspectives that exist between different social actors. Alternatives to be considered can reflect a diversity of PES targeting strategies which would be evaluated against the criteria set to

allow critical judgement to bear upon each proposed eventual incentive arrangement. The ultimate objective would be to design a PES scheme which 'targets' as many identified criteria as possible while maintaining the legitimacy of the process through continuous deliberation. We can then move beyond isolated and objective solutions in favour of a targeting approach to PES that is based upon the social resolution of diverse and legitimate ways of knowing (Garmendia and Pascual, 2013).

4.7. Conclusion

While conceptually convenient, we argue that externally imposed strategies of cost-effectiveness, efficiency and poverty alleviation for optimising PES reinforce the atomized nature of black-boxing ecological and societal relationships under a dominant neoliberal narrative. Ecosystem services are fundamentally *socially interpreted benefits* provided by nature and result from systems of local knowledge and land stewardship that together manifest in the delivery of particular services that are not only of value for human well-being but also permit society to exist (Muradian and Rival, 2012). While social norms have the potential to exhibit a strong ethic of self-maximisation, such as in market-integrated societies, why must natural resource managers ignore the possibility that different social norms or less convincingly, that *any* social norms influence the rational self-interested individual in making land-use decisions? Thus, if environmental concerns have social emergent properties, it is not possible to capture collective dynamic relationships with the environment through a single narrative no matter how holistic it is in considering multiple scales or precisely addressing multiple political objectives. Spatial visualisations that fail to be explicit about the legitimacy of the social narrative considered (or not considered) will not challenge or compare existing human-environment value systems. Rather than picking and choosing services for beneficiaries with the greatest purchasing power, we must acknowledge that socio-ecological systems are complex mosaics of highly managed and natural systems which provide a multitude of ecosystem services through a process of dynamic and evolving social processes. Hence even the most inclusive attempts at "win-win" (or even win-win-win *ad infinitum*) outcomes will still lose unless researchers, scientists and project managers move beyond treating socio-ecological systems as mathematical puzzles to be solved

and instead recognize the inherent tensions and trade-offs of local and expert knowledge and irreducible value systems.

We suggest that PES should be understood as a social construction that is mediated by local realities and how these realities may influence the form and terms of negotiation and design of solutions (Corbera et al., 2007). The emphasis on the social construction of nature's benefits to humanity and the fragile and dynamic balance between ecological relationships, human management systems and the confrontation of value ethics that underpin them preclude the possibility that conservation policy tools such as PES can be implemented *ex-situ*. Instead, deliberative processes for the distribution of landscape management providing a suite of social and ecological benefits can highlight the relevance of group identity and collective consciousness that may result in the negotiation of outcomes that are considered more fair or legitimate (Parks and Gowdy, 2013). Targeting efforts for PES should not only identify trade-offs between multi-dimensional objectives but assess the extent to which such trade-offs are socially legitimate. Spatial visualisation for targeting ecosystem services to compare multiple legitimate narratives and therefore sets of criteria held by different sets of social actors can act as a form of sensitivity analysis not of variation across scales but across different languages of valuation.

More research is needed which examines the nuanced responses of endogenous preferences or participation in collective action when a varied set of both individual and communally-oriented incentives are introduced (Narloch et al., 2012). Such analysis would be particularly relevant for the proposed PES scheme in the Sundarijal catchment, where levels of social capital and gradients of household well-being differ between villages. Spatial analysis of socio-ecological systems could involve the development of maps reflecting formal and informal institutions as well as plural values and perspectives regarding ecosystem services and local needs or constraints. These maps could be developed in concert with a participatory method such as systems dynamics modelling to examine how multiple interpretations of socio-economic and ecological systems are intertwined and how these might change over time. Only when multiple conceptualisations of eco-social relations are expressed and open to critical deliberation will PES optimisation have the legitimacy for achieving what the various 'publics' have to value.

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4.9. References

Alix-Garcia, J., de Janvry, A., Sadoulet, E., 2008. The role of deforestation risk and calibrated compensation in designing payments for environmental services. Environment and Development Economics 13, 375-394.

Alkire, S., Santos, M.E., 2010. Acute Multidimensional Poverty: A New Index for Developing Countries. Oxford Poverty and Human Development Initiative Working Paper No. 38. Oxford.

Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosystem Services 5, 27-39.

Bryan, B.A., Raymond, C.M., Crossman, N., King, D., 2010a. Comparing spatially explicit ecological and social values for natural areas to identify effective conservation strategies. Conservation Biology 25 (1), 172-181.

Bryan, B.A., Raymond, C.M., Crossman, N.D., Macdonald, D.H., 2010b. Targeting the management of ecosystem services based on social values: where, what, and how? Landscape and Urban Planning 97, 111-122.

Calder, I.R., 2005. Blue Revolution: Integrated Land and Water Resource Management (2nd edition). Earthscan, London.

Campbell, B., 1998. Conversing with Nature: Ecological Symbolism in Central Nepal. Worldviews: Global Religions, Culture, and Ecology 2 (1), 123-137.
Castree, N., 2008. Neoliberalising nature: the logics of deregulation and regulation. Environment and Planning A 40 (1), 131-152.

Central Bureau of Statistics, Government of Nepal, 2011. Nepal Living Standards Survey: Statistical Report, Volume Three. Kathmandu, Nepal.

Chen, X., Lupi, F., Viña, A., He, G., Liu, J., 2010. Using cost-effective targeting to enhance the efficiency of conservation investments for ecosystem services. Conservation Biology 24, 1469-1478.

Corbera, E., Brown, K., Adger, W.N., 2007. The Equity and Legitimacy of Markets for Ecosystem Services. Development and Change 38 (4), 587-613.

Crossman, N., Bryan, B.A., 2009. Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. Ecological Economics 68, 654-668.

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision-making: time to deliver. Frontiers in Ecology and the Environment 7, 21-28.

Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65 (4), 663-674.

Fehr, E., Falk, A., 2002. Psychological foundations of incentives. European Economic Review 46 (4), 687-724.

Ferraro, P.J., Kiss, A., 2002. Direct payments to conserve biodiversity. Science 298, 1718-1719.

Fisher, J., 2013. Chapter 2: Justice implications of conditionality in Payments for Ecosystem Services: A case study from Uganda. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Garmendia, E., Pascual, U., 2013. Chapter 8: A justice critique of environmental valuation for ecosystem governance. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. Environmental Management 45, 488-501.

Harvey, D., 2005. A Brief History of Neoliberalism. Oxford University Press Inc., New York.

Jack, B.K., Leimona, B., Ferraro, P.J., 2009. A revealed preference approach to estimating supply curves for ecosystem services: use of auctions to set payments for soil erosion control in Indonesia. Conservation Biology 23, 359-367.

Karn, P.K., 2008. Making Payment for Environmental Services (PES) work: A case study of Shivapuri National Park, Nepal. International Union for the Conservation of Nature. Kathmandu, Nepal.

Kerr, J., Vardhan, M., Jindal, R., 2012. Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. Ecological Economics 73, 220-227.

Kerr, J., Vardhan, M., Jindal, R., 2014. Incentives, conditionality and collective action in payment for environmental services. International Journal of the Commons 8 (2), 595-616.

Knight, A.T., Cowling, R.M., Difford, M., Campbell, B.M., 2010. Mapping human and social dimensions of conservation opportunity for the scheduling of conservation action on private land. Conservation Biology 24, 1348-1358.

Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69 (6), 1228-1236.

Kroeger, T., 2013. The quest for the "optimal" payment for environmental services program: Ambition meets reality, with useful lessons. Forest Policy and Economics 37, 65-74.

Lele, S., 2013. Chapter 6: Environmentalisms, justices, and the limits of Ecosystem Services Frameworks *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Martin, A., Akol, A., Philips, J., 2013. Chapter 4: Just conservation?: On the fairness of sharing benefits. *In*: Sikor, T (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

McAfee, K., 2012. The Contradictory Logic of Global Ecosystem Services Markets. Development and Change 43 (1), 105-131.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202-1208.

Muradian, R., Rival, L., 2012. Between markets and hierarchies: The challenge of governing ecosystem services. Ecosystem Services 1, 93-100.

Naidoo, R., Iwamura, T., 2007. Global-scale mapping of economic benefits from agricultural lands: Implications for conservation priorities. Biological Conservation 140 (1-2), 40-49.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Global mapping of ecosystem services and conservation priorities. PNAS 105 (28), 9495-9500.

Narloch, U., Pascual, U., Drucker, A.G., 2011. Cost-effectiveness targeting under multiple conservation goals and equity considerations in the Andes. Environmental Conservation 38 (4), 417-425.

Narloch, U., Pascual, U., Drucker, A.G., 2012. Collective Action Dynamics under External Rewards: Experimental Insights from Andean Farming Communities. World Development 40, 2096-2107.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, R., 2009. Modelling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7, 4-11.

Nepal Gazette, 2008. Notice of Government of Nepal (GoN). Ministry of Labour and Transport. Part 58, Supplementary 25 (e). Department of Print. Kathmandu, Nepal, 1-3.

Nepal, M., Bohara, A.K., Gawande, K., 2011. More Inequality, More Killings: The Maoist Insurgency in Nepal. American Journal of Political Science 55 (4), 886-906.

Newburn, D., Reed, S., Berck, P., Merenlender, A., 2005. Economics and land-use change in prioritising private land conservation. Conservation Biology 19 (5), 1411-1420.

Norgaard, R.B., 2010. Ecosystem services: from eye-opening metaphor to complexity blinder. Ecological Economics 69 (6), 1219-1227.

Parks, S., Gowdy, J., 2013. What have economists learned about valuing nature? A review essay. Ecosystem Services 3, 1-10.

Pascual, U., Muradian, R., Rodriguez, L.C., Duraiappah, A., 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69, 1237-1244.

Potschin, M.B., Haines-Young, R.H., 2011. Ecosystem services: Exploring a geographical perspective. Progress in Physical Geography 35 (5), 575-594.

Raymond, C.M., Brown, G., 2011. Assessing conservation opportunity on private land: Socio-economic, behavioural, and spatial dimensions. Journal of Environmental Management 92, 2513-2523.

Richards, M., Davies, J., Yaron, G., 2003. Stakeholder Incentives in Participatory Forest Management.: A Manual for Economic Analysis. ITDG Publishing, London.

Sagoff, M., 1998. Aggregation and deliberation in valuing environmental public goods: A look beyond contingent pricing. Ecological Economics 24, 213-230.

Satake, A., Rudel, T.K., Onuma, A., 2008. Scale mismatches and their ecological and economic effects on landscapes: A spatially explicit model. Global Environmental Change 18, 768-775.
Sikor, T., 2013. Chapter 1: Introduction: Linking ecosystem services with environmental justice. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Singh, S., 2013. Chapter 7. Local Governance and Environment Investments in Hiware Bazar, India. *In*: Healy, H., Martínez-Alier, J., Temper, L., Walter, M., Gerber, J-F. (Eds.). *Ecological Economics from the Ground Up*. Routledge, London.

Swetnam, R.D., Fisher, B., Mbilinyi, B.P., Munishi, P.K.T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N.D., Marshall, A.R., Lewis, S.L. 2011. Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling. Journal of Environmental Management 92, 563-574.

Uchida, E., Rozelle, S., Xu, J. 2009. Chapter 9: Conservation Payments, Liquidity Constraints and Off-Farm Labor: Impact of the Grain for Green Program on Rural Households in China. *In* An Integrated Assessment of China's Ecological Restoration Programs. Yin, R. (Ed.) Springer, New York.

Van Hecken, G., Bastiaensen, J., Vásquez, W.F., 2012. The viability of local payments for watershed services: Empirical evidence from Matiguás, Nicaragua. Ecological Economics 74, 169-176.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69 (6), 1245-1252.

Wendland, K.J., Honzak, M., Portela, R., Vitale, B., Rubinoff, S., Randrianarisoa, J., 2010. Targeting and implementing payments for ecosystem services: Opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecological Economics 69 (11), 2093-2107.

Wunder, S., 2008. Payments for environmental services and the poor: concepts and preliminary evidence. Environment and Development Economics 13, 279-297.

Wünscher, T., Engel, S., Wunder, S., 2006. Payments for environmental services in Costa Rica: Increasing efficiency through spatial differentiation. Quarterly Journal of International Agriculture 45 (4), 317-335.

Zander, K.K., Dunnett, D.R., Brown, C., Campion, O., Garnett, S.T. 2013. Rewards for providing environmental services – Where indigenous Australians' and western perspectives collide. Ecological Economics 87, 145-154.

Zhang, W., Pagiola, S., 2011. Assessing the potential for synergies in the implementation of payments for environmental services programmes: an empirical analysis of Costa Rica. Environmental Conservation 38 (4), 406-416.

Connecting text

Upon revealing the trade-offs between multiple objectives and across differing yet legitimate values and perspectives held by social actors implicated in a water management conflict, it becomes clear that novel methodological tools are needed which combine both technical and social knowledge in a coherent and transparent manner. The previous chapter highlighted the discrepancies of ontology or scale in locating patterns of ecological vulnerability and well-being deprivation across the landscape. Such information can be particularly useful when combined with a focus on epistemological diversity in characterising resource management conflicts from diverse perspectives. In the following chapter, we apply social multi-criteria evaluation (SMCE), a deliberative process and decision analysis aid which can break down a complex water resource management conflict into its components to critically assess each course of action and reassemble these pieces to present a coherent evaluation of policy alternatives for decision-making.

The case study of the Sundarijal catchment in Nepal presented in the previous chapter is analysed once again in evaluating various management alternatives (including PES payment options) across a range of criteria which reflect capability enhancements in improving well-being for both service providers and users in the Kathmandu Valley. The methodological and deliberative process in considering plural values and capabilities presented here address the general objectives of the thesis as well as specific objectives 5 and 6. SMCE is conducted through the NAIADE software in defining criteria as importance coefficients rather than weights in order to vary the degree of compensation that a given criterion can impose on other criteria. The results of this study contribute to a more deliberatively-based approach for co-constructing PES arrangements through collective understanding and social learning rather than through the imposition of a singular value ethic (or compensation between legitimate languages of valuation). This chapter has been submitted as a manuscript to *Ecological Economics* (Authors: Kolinjivadi, V., Gamboa, G., Kosoy, N., and Adamowski, J.). It will be the first analysis to consider multi-criteria methods for the design of PES.

Chapter 5

Capabilities as Justice: Analysing the acceptability of Payments for Ecosystem Services through 'Social multi-criteria evaluation'

5.1. Abstract

'Payments for ecosystem services' (PES) is rapidly becoming a popular governance intervention within water resources management to align land-use stewardship to conserve critical watershed services while simultaneously improving human well-being through the provision of incentives. Considerable evidence indicates that complex socio-ecological systems are characterized by substantial uncertainties as well as diverse and conflicting values both spatially and temporally. The market logic of commodifying ecological relationships according to the purely utilitarian values of individuals greatly simplifies this complexity and may result in reinforcing pre-existing inequities. As such, conceiving PES as a social contract built upon the incorporation of technical knowledge and participatory deliberation rather than market principles alone better reflects dynamic socio-ecological systems. This paper introduces two novel components for refining the legitimacy of PES in water resource management. Firstly, we broaden consideration of human well-being in PES beyond income effects by considering justice as the freedom or capability to 'do and be' whatever is desired. In this manner, 'payment' for ecological land-use stewardship is characterized as the set of incentives necessary which aid in overcoming obstacles determined through individual and collective perspectives. Secondly, this paper applies social multi-criteria evaluation as a decision-support framework to determine the acceptability and payment vehicle of PES within a set of alternative policy considerations for a complex ecosystem management decision. Through both technical and social evaluations of different management options against a set of criteria, we highlight the legitimacy that different PES designs may have for improving water quality and capabilities for well-being.

Keywords: PES; capabilities approach; social multi-criteria evaluation; water resources management

5.2. Introduction

Payments for ecosystem services (PES) aims to remunerate land managers for the stewardship activities they engage in to protect identified ecosystem services. A critical objective for a PES project is to ensure 'additionality' or the incremental improvement in the delivery or protection of ecosystem service(s) 'paid' for. Payments which are argued to be economically competitive to alternative forgone land-use activities must also coincide with specific locations where the configuration of ecosystem structure associated with proposed land-use behaviour changes best aligns with the delivery of identified ecosystem services (Brouwer et al., 2011). A key assumption here is that specified configurations of ecological structure can result in readily procurable and compartmentalized ecosystem services by way of uncertain and highly variable complex natural processes (Norgaard, 2010). Such an assumption places a blinder on wider human impacts on the biosphere across spatial and temporal scales which make predicting the ecological impacts of land-use interventions increasingly difficult and costly. Moreover, understanding how plural values associated with ecological goods and services can reinforce patterns of human development may be critical to avoid oversimplifying socio-ecological complexity. With the help of a case study, this paper will consider one such market-based approach known as 'Payments for ecosystem services' (PES) in terms of its ability to achieve positive environmental outcomes by highlighting the extent to which this tool improves perceptions of social justice. Here we employ social multi-criteria evaluation (SMCE) to compare proposed payment and non-payment options against a varied set of criteria that reflect the values of diverse social actors involved in a natural resource management conflict.

The theoretical basis of PES applications emphasizes cumulative well-being outcomes in which service beneficiaries can pay or compensate upstream providers for the service delivered if the overall benefit received by both parties outweighs the losses of anyone who might have been made worse off in the process (Sikor, 2013). Hence within PES theory, the concept of justice is reduced to a utilitarian framework associated with the Pareto principle of neoclassical economics; this is evident in how equity or justice is attempted to be included in PES schemes (Sikor, 2013). Plural conceptions of justice are not considered. For example, practitioners and academics have typically considered justice of ecosystem service provision from an efficiency

perspective in terms of income or assets, in which allocations to lower income individuals provides more 'equity bang' for the buck. (Gauvin et al., 2011; Alix-Garcia et al., 2008). Entrenched inequalities which may exist are not addressed or considered so long as there is a total gain in ecosystem services from the PES transaction (Sikor, 2013).

The possibility of linking human well-being with ecological protection of critical ecosystem service goods and flows is not only tangible, but can be achievable if we conceive of sustainability science as a fully normative and socially conceived process. Thus, it is crucial to differentiate between means (e.g. participatory process; appropriate institutional arrangements, natural resource management strategies) and ends objectives (e.g. meeting human needs) (Daly, 2007). Sen (2010) recognised these ends objectives as fundamental entitlements or freedoms as 'capabilities'. Fair deliberation on such entitlements is needed to provide the freedom or opportunities for individuals to act, do, or be in ways that they desire (Sen, 2010; Robeyns, 2006). The concept was developed as a critique to the neoclassical economics model that takes a utilitarian perspective in equating well-being with monetary accumulation (Sen, 1999). Instead, Sen (1999; 2010) claims that we must not confuse the *means* for improving well-being with the actual *ends,* and thus need to concentrate on the actual abilities and constraints that identify the extent to which individuals can achieve what they value. It also has the objective of pursuing a more equitable position for designing policy which is built on social justice and differential capacities of people to achieve what they want with the resources they have.

Consideration of capabilities requires incorporating a diversity of perspectives, histories, social positions and resulting opportunities and constraints. Hence, the likelihood of payments for improving capabilities requires an evaluative tool that explicitly evaluates policy strategies against criteria that can expand capability sets. Here, we adopt a uniquely deliberative decision-making framework known as social-multi-criteria evaluation which prioritises *procedural* and *distributional* justice of management as well as the *recognition* of diverse values surrounding the decision. While a suite of decision support methods are emerging which extend beyond 'objective' solutions and seek to bridge social legitimacy with scientific credibility (see Stagl, 2007 for a review of these approaches), we choose SMCE as its use in environmental policy

decision contexts is expanding, including the fields of water management and regional planning (De Marchi et al., 2000; Garmendia and Pascual, 2013).

SMCE is a valuation technique rooted in social choice theory. It embraces epistemological uncertainty by considering diverse and often conflicting values present in society (i.e. social incommensurability) while juggling multiple scenarios of uncertainty from inter-disciplinary science (i.e. technical incommensurability). (Munda, 2004). SMCE involves a number of stages, beginning with an institutional analysis of secondary materials, collection of primary data from in-depth interviews and focus groups, and proceeds with an evaluation phase in which decision outputs or alternatives are compared with a combination of technical and social criteria reflecting the decision context.

Unlike traditional multi-criteria evaluation, SMCE prioritises the deliberative and public process of the valuation and integrates both technical (science) and citizen or traditional knowledge systems for approaching environmental decision contexts. In this way, SMCE considers ontological complexity by incorporating socio-economic, ecological, cultural, political, and technical dimensions or scales together; a feature missing in current environmental impact assessments, cost-benefit analysis, risk analysis or life-cycle analysis tools used in decision-contexts (Munda, 2004). Evaluation techniques such as SMCE seek to expose information asymmetries, value diversity, and potential conflicts arising from mistrust which are all essential components to overcome in establishing PES schemes (Corbera et al. 2007; Vatn, 2010). Hence, the rationale for utilising SMCE in considering social incommensurability (equitable management and payment) and technical incommensurability (incentive targeting for achieving ecological and structural poverty objectives) is particularly salient in the context of PES.
In this paper, we apply a novel rationale for SMCE to score different payment and non-payment (PES) alternatives against raised economic prosperity, ecological stability, and social equity criteria that are raised by stakeholders in determining a strategy for water resource management in the Sundarijal catchment of Nepal. The following section expands on the capabilities approach from the lens of the ecosystem service framework while raising the possibility of designing PES as improvements in capability sets. Section 5.4 will explain the case study, methodology of data collection and how the SMCE was employed. Section 5.5 will offer results of the evaluation,

while section 5.6 will offer a brief commentary of procedural and distributive justice for service providers to enhance capabilities through PES arrangements. A brief conclusion is provided in section 5.7.

5.3. Capabilities and Ecosystem Services

Capabilities frame sustainability as a dynamic ethical or normative understanding that requires constant deliberation to provide a trajectory for human relations and systems within the biosphere (Scerri, 2012). As Bookchin (1985) has argued, it is the domination of humans by other humans that have influenced the state of local and ultimately global ecosystem functions and processes. Hence confronting power dynamics between social actors is necessary for deliberation of value systems to be expressed and taken into consideration. Further, as Ballet et al. (2013) pointed out, protecting and maintaining the services of the environment demands critical attention on patterns of social organisation. The authors go on to argue that individuals, communities and social institutions as a whole have unique representations of nature that are complex, multi-dimensional and confrontational. A socialisation process that places freedoms to secure human needs and wants in a fair and just manner across generations can then reformulate conceptions of development *effectiveness* as meeting human needs while mediating between different perceptions of nature (Scerri, 2012). This approach emphasizes cultural meaning while exposing relations of power in institutional structures designed to achieve capabilities. The emphasis on effectiveness (as opposed to efficiency) pertains to intrinsic uncertainty and stochastic events that make fitting human development within the 'window of sustainability' a frequently moving target. Under these circumstances, we must aim to be effective in achieving socio-ecological goals rather than efficient in terms of assuming fully objective and accurate knowledge systems.

By understanding how dependence on ecosystem services directly influences components of well-being, it is possible to identify structural constraints that limit capabilities for people (particularly the poor) to equitably access these services for enhancing their well-being (MA, 2005; Duraippah, 2004; Polishchuk & Rauschmayer, 2012). It has been widely identified that trade-offs occur both spatially and temporally in the delivery of specific ecosystem services

(Farber et al., 2002; MA, 2005; Bennett et al., 2009). Yet, it is also recognised that the delivery of ecosystem services is a social perception in terms of how benefits of nature to humanity are perceived to address constituents of well-being (MA, 2005; Fisher et al., 2010). Hence, bridging the gap between ecosystem services and well-being requires explicit attention towards dimensions of justice in terms of: whose voices influence how ecosystem services are defined, how they should be delivered and an understanding of the political ecology of past and present natural resource management in a particular context.

In making the conceptual link between ecosystem services, human well-being, and capabilities for people to act and be, it is also necessary to identify the personal and societal factors that constrain individuals from enhancing functionings to achieve their specific conceptualisations of well-being. For example, the capability of individuals or downstream communities to benefit from the regulating services of water purification in order to improve their health will be a function of externally-imposed land-use pressures, ecological trade-offs between services across space and time, existing institutions for resource distribution, social norms, and inter-personal relationships (Polishchuk and Rauschmayer, 2012; MA, 2005). Establishing a link between land-use stewardship for the protection of a suite of ecosystem services and resulting impacts on capabilities to do and be requires a deliberative evaluative space to judge welfare outcomes through a multidimensional interpretation of well-being by the stakeholders themselves (Duraiappah, 2011).

PES schemes which equate well-being enhancement with a uni-dimensional compensation approach according to opportunity cost recovery and / or conditional improvements in service delivery risk ignoring other deprivations related to human dignity, empowerment and ability to control one's destiny. For example, if adequate nutrition is identified as lacking by a given community, compensating cash payments to reforest land rather than grow food will not directly improve this indicator of well-being (Pascual et al, 2010). Likewise, service providers may value the opportunity to develop new skills or crafts which may be closely tied with stewardship of the natural resource base to continue providing endowments for livelihood choices to develop. Cash payments reflecting opportunity costs represent the means to improving what an individual or community has reason to value, but fail to directly address the *leverage* factors: personal, societal, or

physical barriers which provide the collective agency to actualize well-being improvements. Thus, we can conceptualise 'payments' as the contextually appropriate investment in social and individual leverage factors which can overcome deprivations and increase capabilities for service providers over time through sustainable land-use stewardship. Resources to invest can be characterised as financial or in-kind incentives, human labour, and/or the creation of new social institutions and norms that confront or empower identified leverage factors. From a capabilities perspective, PES negotiations that define payments in terms of these resources would actively encourage the deliberation of these multi-dimensional perspectives so as to effectuate a common understanding among negotiating parties linking land-use management with human welfare through the long-term maintenance of the critical ecosystem functions (Rosa et al., 2004).

Designing payments for improving capabilities requires an evaluative tool that can assess policy strategies in terms of how land-use stewardship for ecosystem services is linked to a set of leverage factors which either minimize perceived barriers and / or maximise the self-determination and agency necessary to act or achieve what is of value. The combination of deliberative participation and explicit evaluation of policy strategies against criteria acting as leverage factors increasing capabilities lends well to the use of SMCE. This study will be the first to link PES policy for improving water quality to the achievement of enhanced capabilities in terms of improved well-being, in this case for upstream service providers.

5.4. Site description and Methodology

5.4.1. Study Site

The Sundarijal catchment is located within Shivapuri-Nagarjun National Park (ShNP), approximately 12 kilometres north of Kathmandu, Nepal. The catchment empties into the larger Bagmati River basin and contributes over a third of the total piped water supply entering Kathmandu Valley (Karn, 2008). Specific hydrological services provided by the catchment include: regularity of water flow, quality of water, micro-climate regulation, and the regulation of soil movement (Maskey, 2008). These benefits accrue to downstream hydropower generation, irrigation for farmers on the fringes of the Valley, and for drinking water consumption by urban

residents (Karn, 2008). In order to ensure the protection of these services, a number of non-governmental organisations based in Kathmandu as well as the Ministry of Forestry have recognised the role of PES in promoting land-use stewardship which minimises soil loss and degradation of water quality within upstream communities through the provision of incentives. Figure 4.1, shown earlier, indicates the location of ShNP within Nepal as well as the location of upstream communities within the Sundarijal catchment that have been identified as service providers for the proposed PES scheme[13].

While households within the upstream communities do have recognised land claims prior to the ShNP being gazetted in 2002, they are not entitled to forest management rights due to the surrounding area being recognised as a national park. Hence, these communities have been essentially 'fenced in' to the park with limited market access and restricted mobility. The most common livelihood in these upstream communities is the production of a buckwheat liquour which requires the collection of fuelwood for heating water in the fermentation process. In addition to the low but increasing deforestation rate of 0.19% per annum, the incidence of human and livestock faecal matter contamination of waterways has raised concerns over impacts to water quality downstream (Shrestha, unpublished, 2012). Accordingly, the objective of the proposed PES would be to shift livelihoods and land-use practices that will provide greater long-term improvements to well-being while minimising impacts to the provision of hydrological services.

5.4.2. Description of social actors in the SMCE

The first step in identifying key informants and mapping the spectrum of relevant social actors involved an institutional analysis examining administrative documents produced by the International Centre for Integrated Mountain Development (ICIMOD) and the International Union for the Conservation of Nature (IUCN), as proponents of PES in the Sundarijal catchment.

[13] As of 2014, the organisations that have promoted PES implementation in ShNP have retracted interest due to claims by downstream stakeholders that they are powerless to mobilise resources or engage in further discussion without clear legal definition of the PES mechanism at ministerial level. Accordingly, the evaluation presented here no longer represents actual management alternatives being considered for the settlements within ShNP. In light of this understanding, the aim of this research is to illustrate a methodological approach for evaluating the feasibility of PES implementation as a management solution by applying explicit judgement to legitimate criteria sets reflecting the plural values of involved social actors.

In addition to PES proponents, other social actors involved in the management of water from the Sundarijal catchment include a tourism NGO, a state-owned hydroelectric power producer, a public-private water supplier, local and regional governments, an international development bank, the warden of ShNP, and two government advisory boards tasked with developing policies for the Bagmati River basin within which the Sundarijal catchment lies.

A recent study which utilised SMCE to determine management alternatives for an estuary in Spain had identified actors in the decision context according to their levels of influence on the final outcome (Garmendia and Gamboa, 2012). We adopt a similar technique by identifying social actors through various classifications according to attributes of power, legitimacy and urgency, using a framework on stakeholder theory developed by Mitchell et al. (1997) (Table 5.1). The full typology of classifications include: dominant, definitive, dependent, dangerous, demanding and discretionary (see Mitchell et al., 1997 for full descriptions). For example, the public-private water supplier for Kathmandu Valley (KUKL) is identified as a 'dominant' actor given that it has the power and legitimacy to take action towards establishing a PES scheme in the catchment, but lacks the urgency to do so. The Nepal Electricity Authority also falls within this classification. The Asian Development Bank (ADB) falls into the category of 'dangerous' stakeholder as it possesses the power and urgency to effectuate a management decision, but lacks local legitimacy or attempts to impose legitimacy of its efforts through *a priori* determined proposals and timelines. The classification of the local government (Sundarijal village development council (VDC)) and the warden of ShNP were more challenging to distinguish. They share attributes of legitimacy and urgency in calling for an incentive-based mechanism to promote land-use management upstream but have less power than higher level government or other downstream actors. Hence these actors are not quite 'definitive' but not necessarily 'dependent', and likely lie in the grey area between these classifications. The High Powered Committee for the Integrated Development of the Bagmati Civilization (HPCIDBC) was established under the Ministry of Urban Development for implementing infrastructural projects to improve the quality of water in the Greater Bagmati Basin. The Committee not only has the power to influence water management in the upper Bagmati Basin (including within ShNP) but also has the legitimacy (i.e. government mandate) and urgency to effectuate an intervention. In this sense, this agency can be considered a definitive stakeholder and has been identified by the

ADB as the ideal implementation or executing agency for upstream water management projects. Indeed, funds provided by the ADB for infrastructural developments as a management alternative for improving water quality are being channelled directly to the HBCIDBC. The intermediaries (e.g. ICIMOD and NETIF Nepal) have legitimacy and express urgency for change, but lack the power to make decisions and are thus considered 'dependent'. In contrast, residents of the communities have legitimacy to influence the decision context but lack power or urgency in influencing the decision and are hence 'discretionary'.

5.4.3. Information collected from social actors

SMCE incorporates both qualitative and quantitative data elicitation and highlights relationships in the views of social actors in order to better establish the space where agreement and disagreement lies within the decision context (Gamboa, 2006a). The methodology for data collection described below was chosen according to theoretical elaboration of SMCE described by Munda (2004) beginning with historical and institutional analyses, in-depth interviews, focus groups and stakeholder workshops. A description of the feedback processes employed is shown in Table 5.2. Specifically, the research team:

> a) conducted 135 semi-structured personal interviews with 1/3rd of the total household population (e.g. 135 households) within the identified upstream communities of 'service providers'. Households were randomly chosen through stratified random sampling[14]. Interviews assessed perceptions of household health; food security, economic opportunity; education; social equity; barriers or challenges for improving household well-being; perceptions on the relevance of soil conservation measures and the consequences of poor soil management; perceived benefits from the surrounding Shivapuri forest, and perceptions of the proposed PES. Determination of opportunity cost of producing on the land was also calculated for a third of interviewed households (i.e. 45 households selected once again through stratified random sampling).

[14] According to the full set of households within each Sundarijal VDC ward that included ShNP communities. Wards are sub-jurisdictions within each VDC in Nepal.

Actor	Type	Summary of Position
Sundarijal VDC	Dependent	Supportive of payments that aid in transitioning livelihoods and which are ecologically targeted
High Powered Commission for Integrated Development of Bagmati River Civilisation (HPCIDBC) /	Definitive	In favour of engineered approaches to improving watershed services; does not believe central government will prioritise the interests of upstream land users in water management decisions for the Bagmati River. PES could play a role once a centralized river basin organization is developed. This agency would also be tasked with environmental monitoring related to impacts from dam construction.
Asian Development Bank (ADB)	Dangerous	The financier of the engineered approach to enhancing water supply and quality. ADB supports the HPCIDBC as taking on the role of river basin organization for the Greater Bagmati and the primary implementing agency for the Bagmati River Improvement Project.
Kathmandu Upatyaka Khanepani Limited (KUKL)	Dominant	Claim that villagers should be removed, with PES as a secondary consideration; recognize the importance of watershed management for targeting water quality improvements
SnNP Warden	Dependent	Supportive of PES as a mechanism that local user groups can argue for relative to their motivation for improving forest and soil resources; also in favour of engineered approaches as a more immediate means of improving watershed services
Nepal Electricity Authority (NEA)	Dominant	Royalties already provided could be used for cost-efficient soil, forest or waste management in Sundarijal catchment if such a priority was made at the DDC level; not supportive of PES over the status quo
Kathmandu Valley Water Supply Management Board	Dominant	Emphasis is placed on groundwater management and engineered water supply projects, but PES is as a negotiated approach is also necessary
Kathmandu DDC	Dominant	In favour of incentives for land-use management that enhances water quality for the valley, but takes a back seat in terms of action without the support of the Ministry of Forestry and Local Development Ministry for linking payments with well-being or livelihoods in the National Park
NGOs / Inter-governmental institutions (ICIMOD, IUCN,NETIF)	Dependent	Supportive of PES as a means to bridge well-being and ecological objectives together; believes payments should be motivate land-use change and should be designed for cost-effectiveness
Upstream communities within ShNP	Discretionary	Accepting of PES, but not supportive of engineering approaches in the area that have received approval without their knowledge; distrustful of VDC

Table 5.1: Key social actors implicated in the decision-context of water resource management in the Sundarijal catchment. *Additional social actors that are interested to learn more of the PES concept, but are not currently active in its promotion in Sundarijal include: Forest Action Nepal (NGO); Ministry of Forestry (has power to make decision over PES, but skeptical of how it would function and more in favour of short-term engineered approaches); Local Development Ministry; water-user committees (Kathmandu); Water Tariff Fixation Commission; Sundarijal Sarokar Samiti (NGO). The perspectives provided here were obtained from individuals whose viewpoints do not necessarily reflect the institution to which they belong.

Questions were asked in an open-ended manner with follow-up probing questions, such as *"Why do you feel this is important?"* in order to extract fundamental motivations (Gregory and Keeney, 1994; Vignola et al., 2012).

b) Conducted 7 focus groups to deliberate upon the insights that emerged from the household interviews. Between 15 to 20 people participated in each focus group, which were separated according to gender, community, and in some cases ethnic differences that existed within the same community. During these focus groups, perceived deprivations which corresponded to those reflected in individual interviews were identified and classified according to the categories of economic opportunity, social equity, and ecological stability.

c) Carried out 8 individual semi-structured in-depth interviews with key informants identified based on their involvement in promoting PES or as potential financiers or beneficiaries of the hydrological services of the catchment. Perceptions were captured in relation to the threats of sediment load and landslide impacts for hydropower generation as well as the impacts of poor waste management for Kathmandu's water supplier in providing potable water for industrial and commercial users. Information was also obtained on the viewpoints of downstream beneficiaries regarding the needs and constraints of upstream communities as well as the potential of reinvesting resources or royalties back into upstream watershed management.

d) Conducted an open meeting with over 20 participants including members of the village local development committee and wider social actors promoting or involved in the approval of the PES scheme in order to evaluate PES in relation to alternative proposed management strategies for the catchment.

ACTIVITY	PLACE AND DATE	PARTICIPANTS
1. Preliminary Meeting	International Centre for Integrated Mountain Development (ICIMOD) *August 28, 2012*	• NGOs; Sundarijal VDC; KUKL; ShNP warden
2. Interviews	Upstream villages: 135 semi-structured interviews • Perceptions of well-being • Opportunity cost of land production • Land-use management practices (*September 9-November 15, 2012*) 10 key downstream informants (*November 28, 2012- December 28, 2012*)	• Household head (135) • KUKL; ShNP Warden; Kathmandu DDC; Sundarijal VDC; ADB-Nepal; KVWSMB; HPCIDBC; NEA; Ministry of Forestry
3. Preliminary Focus Groups (8)	Upstream villages (3 in total) separated by: • gender • caste (in one village) *October 20-November 27, 2012*	• Adults from households interviewed (between 5 and 25 individuals)
4. Final focus groups (3)	Open discussion and presentation of management alternatives of park in each of the three upstream villages *November 14-December 10, 2012*	• Village members (between 10-15 individuals)
5. Open Meeting	Roundtable Discussion of PES feasibility *(Falccha): December 26, 2012*	• NGOs, KUKL, KVWSMB, , upstream community leaders; Sundarijal VDC (local government); Kathmandu DDC (regional government); Ministry of Forestry (national government)

Table 5.2: Description of interactions made with social actors between the period of September and December, 2012.

5.4.4. Alternatives

The proposed management alternatives that were identified were those raised by PES proponents, the ShNP warden, local government (VDC), the ADB and HPCIDBC, as well as local communities within ShNP. Each alternative was recognised as a legitimate strategy during the preliminary meeting. Table 5.3 summarises the proposed management options. These are classified as: (a) 'no payment' options, including status quo (A1-A3), (b) 'targeted payment' options (B1-B4), and (c) 'collective payment' options (C1-C3). Alternatives A1-A3, B3, C2 and C3 reflected proposed management interventions for the catchment. Alternatives C1, B1, B2 and B4 characterised a range of PES designs with varying emphasis on conditionality, efficiency, equity and poverty alleviation described in detail in Pascual et al. (2010). Each of these objectives reflected the interests of the PES proponents (e.g. ICIMOD and IUCN).[15] The final set of management alternatives was exposed to opinion during the open meeting to confirm the relevance of each strategy and to avoid any biases emerging from the research team.

5.4.5. Criteria

The leverage factors that bridge the stewardship of water quality services provided by the Sundarijal catchment with increased capabilities to achieve what is valued can be conceptualised as criteria to which payment or non-payment strategies should be assessed against. Each criterion reflects either personal or collective constraints to empowerment raised in the semi-structured interviews with households and focus groups respectively. They are essentially a technical translation of the needs, preferences and desires of social actors relating to fundamental objectives for resource management in the catchment (Gamboa and Munda, 2007).

[15] While recognizing that the introduction of alternatives that reflected the interests of PES proponents but were not explicitly identified by these or other social actors is discouraged in the application of SMCE, we also recognize that the research team is itself a social actor and must therefore be explicit in the overall objectives or contributions to knowledge that are sought after in the determination of management intervention, in which PES is identified as one possible consideration (Munda, 2004).

Alternative Scenarios	Levels
No payments	A1: **Status Quo-** No management intervention in maintaining or enhancing watershed services in Shivapuri National Park is taken nor any attempts to improve opportunities for new livelihoods or other economic activity in the area A2: **Ecological Preservation-** The management intervention adopted does not involve payments, but involves the removal of all villagers located within the National Park, to maintain the area protected for water quality and supply for downstream populations A3: **Engineered Option**- Asian Development Bank loans are provided to construct a reservoir dam within ShNP as well as downstream water treatment facilities to enhance water supply and quality downstream. Substantial environmental rehabilitation and livelihood enhancement activities for Park communities are mentioned in the project plan
Targeted payments	B1: **Expected Provision**- Payments are provided according to expected provision of water quality improvements associated with land-use change .In this case, payments are given to households contributing most to water quality problems. B2: **Minimal Cost**- Payments are differentiated to economize on influencing land-use change at the lowest cost. In this case, payments are given to households with the lowest opportunity cost for land-use change that enhances stewardship of water quality downstream B3: **Market Integration** – Payments are made to finance technical training, establish microfinance opportunities, and communicate market niches or details of existing market demand. The objective is to transition livelihoods away from subsistence and/or alcohol producing economies. B4: **Pro-Poor**- Payments aim to maximize benefits for the poorest households or those with the most identified deprivations without reference to expected provision of the watershed service or the opportunity cost of the service provider. In this case, deprivations are recognized in the categories of food security, health, education and assets.
Egalitarian payments	C1: **Equal-Individual**- Payments are provided uniformly to all households in the villages within the National Park according to a fixed budget and independent of expected provision of the watershed service or opportunity cost. C2: **Common Goods**- Payments are given to Sundarijal VDC government to be invested in opportunities for all villagers to have access to regardless of opportunity cost or expected provision of the watershed service. In this case, payments may reflect micro-credit loans, training skills for tourism and/or more ecologically-benign agriculture techniques C3: **Citizen Decision/Buffer Zone User Groups**- Payments are provided to local user groups associated with the National Park buffer zone and are invested in the collective interests of each user group[a]

Table 5.3: Summary of proposed alternatives

[a] In the case of C3, it is recognized by social actors that each user group has the knowledge or could accept knowledge support regarding expected provision of watershed services, costs, and localized deprivations in making decisions on how best to allocate payments and for what objective.

The objective is to maximise or minimise the direction of each criterion according to the objectives that each represents for the decision[16]. For example, the overall objective is to reduce implementation costs, while it is recognised to be beneficial to maximise wage labour or protection from landslides associated with each alternative. Thus from a capabilities perspective, criteria are maximised or minimised in relation to whether they facilitate improvements (maximise) or overcome (minimise) barriers for improving functioning sets associated with the capability to utilise financial resources for other purposes, to learn, diversify income streams or improve water quality and risk of land subsidence. Criteria were classified under three dimensions: 'economic democracy', 'ecological integrity' and 'political democracy' reflecting underlying themes in the criteria set. Table 5.4 illustrates the selected criteria, whether they reflect personal or collective needs or constraints, the dimension they fall under, needs and expectations for the criteria, and the indicator or evaluation scale from which they will be measured or scored against alternative management strategies. The method by which each indicator is scored (i.e. qualitatively, fuzzy-numeric or crisp) is also shown.

5.4.6. Valuation of criteria

All criteria were initially elicited through interview and focus group questions from social actors themselves; the characterisation of criteria into more or less acceptable ranges was determined from both household interviews and secondary data (Karn, 2008; ADB, 2013).

[16] Criteria reflect the impacts of each alternative according to the type of indicator or evaluation scale employed. Several criteria (e.g. education, recognition of village interests, new markets, health) identified in this study are termed *constructed* criteria, since they refer to qualitative descriptions that are evaluated through a combination of social deliberation and secondary data where available on a scale ranging from very bad, moderate to very good. *Proxy* indicators indirectly relate to the objective of each criterion and were applied to some criteria such as implementation cost (using opportunity cost); deforestation (using bhari fuelwood collected per household per year), and waste management (number of households with eco-toilets). Finally, *natural* criteria are those that derive directly from the measurement of each indicator. Here, wage labour (measured as average off-farm income) was the sole example (Gamboa, 2006b).

Dimension	Criterion	Needs and Expectations	Indicator / Evaluation Scale
Socioeconomic	New markets	Training and skills building in tourism; organic agriculture and livestock husbandry (P)	*Qualitative*: Number of households involved in training workshops (source: ADB, 2013). 0 = Very Bad; 1-50 = Bad; 51-100 = Moderate; 101-200 = Good; Very good = 201 +
	Mobility	Access to market (P)	Distance from house / number of trips to market per month (numeric/fuzzy)
	Implementation cost	Cost-efficient management (P)	NPV (US$); Annual rent (for Targeted-PES options) (Source: Literature; interviews)
	Wage Labour	Improvement in off-farm income (P)	US$ (numeric / fuzzy)
	Transaction Costs	Minimising information and intermediary fees (C)	Standard deviation of average opportunity costs (B1-B4; C1); Literature (Karn, 2008; ADB, 2013) (A1-A3); 50% management fees (C2-C3)
Social Well-being	Health	Percentage of households with water-borne illness in past month (P)	*Qualitative*: Incidents of expected water-borne illness over a 2 month period: More than 29% of households= Very bad ; 29% = Bad; 20-28% = moderate; 10-19% = Good; 0-9% = Very good
	Education	Improvements in educational opportunities (C)	*Qualitative*: Payment / investment allotted for collective benefit; No = Bad; If YES: Proportion of payment invested education: 0-25% = Moderate; 26%+ = Good
	Recognition of village interests	Potential to empower informal institutions in decision-making (C)	*Qualitative*: 0% = Very bad; 1%-15% = Bad; 16-30% = Moderate; 31-45% = Good; 46% + = Very good
Environmental	Waste management	Access to toilet (septic system or eco-san toilet) (C)	Households targeted for pilot eco-san toilets (numeric / fuzzy) (Source: ADB, 2013; Open Meeting, December 26, 2012)
	Deforestation reduction	Reduction in use of fuelwood extracted (C)	Bhari[a] fuelwood collected / yr; perceived reductions in yearly consumption (numeric / fuzzy; source: household interviews)
	Landslide protection	Reduced incidents of land subsidence (C)	*Qualitative* Stabilisation of outward sloping land : 0 ha = Very bad; 1-14ha = Bad; 15-17ha = More or less bad; 17-25 ha = Moderate; 26-28ha = More or less good; 29-40 ha = Good ; 41 ha+= Very good (Source: ADB, 2013)

Table 5.4: Socio-economic, social and environmental dimensions of identified criteria set by social actors for the choice of a management strategy for ShNP; the needs and expectations associated with these criteria and indicators and/or evaluation scale by which each criterion is measured. The abbreviations P and C indicate whether each criterion characterises perceived personal (household and organisations) or collective needs and constraints to improving quality of life or quality of the decision taken.
[a] 'bhari' is a Nepali unit of measuring firewood and fodder. 1 bhari = 30kg

The evaluation scales used to elicit social actor perceptions were derived from secondary data, including a PES feasibility study produced by the IUCN and the project budget of the ADB. Fuzzy/numeric evaluation was carried out for most criteria, utilising data collected directly from household interviews.

The criterion 'transaction costs' deserves special mention as it was not explicitly expressed by the actors through interviews or focus groups but emerged as an underlying concern implicit in the needs of social actors against costly management decisions. Transaction costs extend beyond only implementation costs to account for "the costs of resources needed to define, establish, maintain, use and change institutions" as well as the communication needed to define the problems that institutional change is meant to address (Marshall, 2013). Achieving management objectives (criteria set) at minimal cost of transaction has been raised as a priority by downstream beneficiaries in the Valley. Given the uncertainties involved in predicting 'transaction costs', a heuristic framework based on an understanding of path dependent institutional choices was used to qualitatively assess potential transactions costs of each alternative. Developed by Marshall (2013), the framework was used to compare naïve implementation costs with potential technological and institutional lock-in costs associated with past and future water management decisions for the Valley. Specifically, we compared the costs of the status-quo (alternative A1) with other alternatives in order to consider: a) potential institutional transition costs of change[17], and b): perceived institutional and technological lock-in costs associated with a management change. The former incorporates costs of research, stakeholder negotiation, infrastructure construction, monitoring and evaluation while the latter includes costs associated with vested interests (e.g. private-public nature of existing institutions)

[17] Informal consensus among actors that a 50% transaction fee was a standard reality in large-scale management projects in Nepal was identified during the open meeting on December 26, 2012.

that oppose changes to management contrary to these interests (Marshall, 2013). For instance, the building of reservoirs to improve water supply as well as the creation of a river basin organisation for water management across the Bagmati river basin (associated with alternative A3) reflect impacts on institutional and technological path dependencies of past and ultimately future management changes (Siwakoti, 2004; Marshall, 2013). With respects to targeted payment alternatives (B1-B4), transaction costs were perceived according to: a) the magnitude of change in terms of the proportion of households required to change land-use practices and b) heterogeneity in opportunity cost[18] for the highest revenue yielding crop (e.g. millet) which would necessitate varied compensation across households. While determination of opportunity costs and associated targeted compensation has been argued to improve the efficiency of PES (Watzöld and Drechsler, 2005), the dynamic nature of these costs combined with the challenge of overcoming asymmetric information would reduce collective action potential and hence raise transaction costs (Groom et al., 2007).

A key feature of SMCE is the emphasis on non-compensation between criteria so that good performances in some dimensions do not overcome bad results in others. As such criteria weights are considered as importance coefficients rather than trade-offs between criteria, thereby reflecting the relative importance of a criterion by the social actor rather than substitution among them. While many algorithms exist to aggregate scores of alternatives against criteria, it is critical to choose an algorithm consistent with the degree of uncertainty reflected in the decision context and which balances technical precision, simplicity and transparency in a participatory setting (Garmendia and Pascual, 2013).For this purpose, we utilise discrete multi-criteria approach known as 'Novel Approach to Imprecise Assessment and Decision Environments' (NAIADE) since the algorithm applied provides an ordinal ranking of alternatives through a combination of crisp and fuzzy set comparison of criteria, without allowing for substitutability between criterion scores (Munda, 2004; Garmendia et al., 2010). Unlike other multi-criteria methods, NAIADE as a non-compensatory approach does not apply traditional weights to criteria to enable all dimensions of value identified by implicated social actors to be included in the evaluation (Munda, 2005). The degree of compensation in criteria aggregation can be

[18] Opportunity costs were assumed to influence the transaction costs equally for all targeted payment alternatives according to a very high calculated standard deviation of opportunity costs across 45 households within the upstream communities of ShNP.

regulated as well as alterations of preference thresholds[19] so that a sensitivity analysis can examine the robustness of assumptions made in scoring and in the overall ranking of alternatives.

5.4.7. Preference Thresholds, Aggregation and Ranking

The comparison of criteria scores for each alternative is performed by a measure of semantic distance between fuzzy membership functions as described in detail in Munda (2008: p. 136-141). The pairwise comparison of each criterion between two alternatives depends on a relation of preference whereby preference (strong or weak) and indifference thresholds are established. The problem inherent in defining preference and indifference thresholds lies in the confidence of precision required in determining crossover values. Accordingly, NAIADE introduces six credibility functions of preference and indifference expressing whether each alternative is *much better* (>>), *better* (>), *approximately equal* (≅), *equal* (=), *worse* (<) or *much worse* (<<) than another alternative. The credibility index ranges from 0 (non-credible) to 1 (credible) montonically increasing in the case of preference for an alternative (Munda, 2008: p 103). An indifference threshold above or below which an agent refuses to declare a preference for one alternative or another is identified as:

$$A_j P a_k \Leftrightarrow g_m(a_j) > g_m(a_k) + q \qquad (5.1)$$

$$A_j I a_k \Leftrightarrow |g_m(a_j) - g_m(a_k)| q$$

where *P* and *I* refer to a preference and indifference relation for a given criterion *g* on a set of pairwise alternatives *a* (Gamboa and Munda, 2007).

[19] The indifference threshold is the maximum difference between the scores of two alternatives on a given criterion that renders no difference of preference between them. The Preference threshold is the minimum distance between the scores of two alternatives on a given criterion that makes one alternative more attractive than the other for that criterion (Gamboa, 2006a). These thresholds were elicited from focus groups, in relation to differences from the status quo that would make a contribution to functioning (either positively or negatively), using standard benchmarks. For example, in the case of wage labour, the Nepalese minimum wage at the time of data collection was used as a baseline to assess changes to the status quo that would 'matter'. Similarly, current households using eco-san toilets versus the perceived distribution and use that would lead to improvements or deterioration in the case of waste management were determined.

The aggregation of criteria is based on an algorithm of the credibility indices that defines a preference intensity index of one alternative to another. The intensity index of preference based upon the credibility indices (*) of >>, >, ≅, =, <<, and < in the following equation:

$$\mu * (a,b) = \frac{\sum_{m=1}^{M} \max(\mu*(a,b)_m - \alpha, 0)}{\sum_{m=1}^{M} |\, \mu*(a,b)_m - \alpha|} \qquad (5.2)$$

where α expresses the minimum threshold on the credibility indices whereby aggregation of criteria is carried out (NAIADE Manual, 1996).

The ranking of alternatives is based upon the preference intensity index as an intersection of two separate rankings. The first considers both the *better* and *much better* preference relations ($\Phi^+(a)$) while the second the worse and much worse preference relations ($\Phi^-(a)$) between two alternatives (NAIADE Manual, 1996). These are expressed as follows:

$$\varphi^+(a) = \frac{\sum_{n=1}^{N-1} (\mu_{\gg}(a,n) \hat{} C_{\gg}(a,n) + \mu_{>}(a,n) \hat{} C_{>}(a,n))}{\sum_{n=1}^{N-1} C_{\gg}(a,n) + \sum_{n=1}^{N-1} C_{>}(a,n)} \qquad (5.3)$$

$$\varphi^-(a) = \frac{\sum_{n=1}^{N-1} (\mu_{\ll}(a,n) \hat{} C_{\ll}(a,n) + \mu_{<}(a,n) \hat{} C_{<}(a,n))}{\sum_{n=1}^{N-1} C_{\ll}(a,n) + \sum_{n=1}^{N-1} C_{<}(a,n)} \qquad (5.4)$$

where N is the number of alternatives and ^ referring to the Zimmermann-Zysno operator which permits varying degrees of compensation (γ) between criteria (from 0, minimum compensation to 1, maximal compensation). (NAIADE Manual, 1996).

5.5. Results

5.5.1. Technical evaluation

Table 5.5 presents the multi-criteria impact matrix illustrating the alternative management interventions, the capability enhancing potential inherent in the criteria set, the preference and indifference relations for each criterion as well as the score of each alternative on these criteria. The majority of alternatives are scored under fuzzy-numeric preferences or linguistic expressions

since the consequences of each policy alternative for a particular criterion cannot be clearly known. Table 5.6 provides the results of the ranking procedure in applying the SMCE under various levels of compensation. It can be seen that under various compensation levels, none of the social actors considered the status-quo (alternative A1) to be acceptable, and all considered that some kind of management change was needed. In addition, alternative A1 ranks the lowest from a capability perspective since villagers perceive that the status quo is particularly debilitating in terms of having the opportunity to achieve what is of value in life.

Alternative C3 was scored the highest, followed by A3 and C2 despite changes in compensability between criteria. The lack of rank reversals across differing levels of compensability indicates that the results of the SMCE are technically robust (Munda, 2010). Alternative C3 ranks first from the perspective of capability improvement since it increases opportunities to local user groups for soil and water quality maintenance or improvement. Support to user groups is underpinned by the existence of functioning informal institutions and reinforced through buffer zone management funding and planned extension support for diversifying livelihoods according to individual strengths and willingness to learn. In particular, this alternative scored positively for improving mobility and hence access to markets, educational opportunities, representation in decision-making, and collective efforts in reducing damage from landslides. Implementation and transaction costs were perceived to remain high, but overall drawbacks were fewer than the second leading alternative (A3).

Criteria	Units	A1	A2	A3	B1	B2	B3	B4	C1	C2	C3
New markets (maximise)	Qualitative	Bad	Very bad	Very good	Moderate	Moderate	Very good	Moderate	Moderate	Moderate	Good
Mobility (maximize) $\mu_{>>}$ & $\mu_{<<}$ = 3[a] $\mu_{>}$ & $\mu_{<}$ = 2 $\mu_{\sim}$ = 1 $\mu_{==}$ = 0.5	Distance to house (km) / number of trips per month	5.19	2.6	3.89	5.19	5.19	2.6	5.19	5.19	2.6	2.6
Implementation Cost[b] **(minimize)** $\mu_{>>}$ & $\mu_{<<}$ = 550,000 $\mu_{>}$ & $\mu_{<}$ = 200,000 $\mu_{\sim}$ = 100,000 $\mu_{==}$ = 50,000	USD	~1,410,000	~2,100,000	~ 4,757,156	~1,484,200	~1,439,313	~1,484,200	~1,446,817	~1,469,805	~2,700,000	~2,691,600
Wage Labour (maximize) $\mu_{>>}$ & $\mu_{<<}$ = 300 $\mu_{>}$ & $\mu_{<}$ = 200 $\mu_{\sim}$ = 100 $\mu_{==}$ = 50	USD	1038.51	778.88	1557.77	1038.51	1038.51	1038.51	1038.51	1038.51	1038.51	1038.51
Transaction Costs (minimize)	Qualitative	Very Good	Very Bad	Bad	More or Less Bad	Moderate	More or Less Bad	More or Less Bad	Moderate	More or Less Bad	Bad
Health (maximize)	Qualitative	Bad	Bad	Good	Moderate	Moderate	Bad	Moderate	Moderate	Good	Good
Education (maximize)	Qualitative	Bad	Bad	Bad	Bad	Bad	Bad	Bad	Bad	Good	Good
Recognition of village interests (maximize)	Qualitative	Bad	Very Bad	Good	Bad	Bad	Bad	Bad	Bad	Good	Very Good
Waste-Management (maximise) $\mu_{>>}$ & $\mu_{<<}$ = 30 $\mu_{>}$ & $\mu_{<}$ = 20 $\mu_{\sim}$ = 10	% targeted for eco-toilets	44	0	70	70	50	50	50	50	70	70

$\mu_{==} = 5$											
Deforestation reduction (minimize) $\mu_{>>}$ & $\mu_{<<} = 210$ $\mu_{>}$ & $\mu_{<} = 100$ $\mu\sim = 50$ $\mu_{==} = 25$	Bhari of fuelwood (1 Bhari = 30kg)	206.4	0	309.6	103.2	154.8	103.2	154.8	154.8	180.6	180.6
Landslide protection (minimize)	Qualitative	Bad	Very Good	Moderate	Good	Moderate	Good	More or Less Bad	More or Less Good	More or Less Good	Good

Table 5.5: Multi-criteria impact matrix of alternative management scenarios against identified criteria reflecting needs and constraints of identified social actors

[a] $\mu_{>>}$ & $\mu_{<<}$ refers to the preference relation for each criterion and provides a credibility index that an alternative is *much better* or *much worse* in a pairwise comparison with another alternative; $\mu_{>}$ & $\mu_{<}$ indicates that an alternative is preferred or not preferred in a pairwise comparison with another alternative; $\mu\sim$ indicates that an alternative is approximately equal to another; and $\mu_{==}$ indicates that the alternatives are approximately equal for that criterion.

[b] Values for 'Implementation Cost' were determined based on opportunity costs of households (for B1-B4, C1), project documents (A3), and secondary literature (A1-A2, C2-C3) (IUCN, 2008; Maskey, 2008). The value differential between alternatives C2 and C3 reflects inclusion of buffer zone management investment for the wards of Sundarijal VDC as specified in Maskey, 2008.

Compensation	First	Second	Third	Fourth	Fifth	Sixth	Seventh	Eighth	Ninth	Tenth
No compensation $\gamma = 0$	C3	A3	C2	B3	A2	B1	C1	B4	B2	A1
Minimal compensation $\gamma = 0.25$	C3	A3	C2	B3	A2	B1	C1	B4	B2	A1
High compensation $\gamma = 0.75$	C3	A3	C2	B3	A2	B1	C1	B4	B2	A1

Table 5.6: Rankings under differing levels of compensation

Alternative C3 was followed by A3 which emphasised the construction of water quality and storage operations. The project document of the Bagmati River Improvement Project (alternative A3) stipulates channelling resources to develop new market opportunities in upstream ShNP communities as well as to improve off-farm wage labour by employing individuals in the construction of an 861,000m^3 dam to be built within the Park. The dam would establish a reservoir to increase the city's water supply during the dry season. The emphasis within the project proposal for upstream areas of ShNP has been on ecological land-use stewardship as well as overcoming the development needs and challenges of Park communities (ADB, 2013).[20] The form that this empowerment takes incorporates alternative C3 in formalising local user groups through a ShNP buffer zone management plan. Intuitively, alternative A3 encompasses alternative C3 given that it would promote all of the same objectives as the latter (on paper) but scored the poorest in terms of implementation cost and continued forest degradation (as a result of dam construction activities). As will be highlighted later, the legitimacy of A3 could also be called into question given that the components of the strategy did not initially result from the involvement of upstream communities.

In terms of the other management strategies, C1 had a poor performance due to ambiguities in relation to how equal payments to all households would be geared toward directly addressing any of the fundamental ends objectives that characterise the criteria set. Alternative C2 referred to communally deliberated payments and management practices. While this strategy was ranked third highest since it addresses critical obstacles in capability sets (e.g. mobility, waste management, associated health improvements, and education improvements), it performed less well in reflecting ideal representation of household voices in decision-making. This was due to perceptions of mistrust with local government (e.g. Sundarijal VDC) who would be tasked with determining and implementing a communal investment 'payment' in exchange for improved land-use stewardship and livelihood change. Improvements in educational materials or the qualification of educators and in the distribution of compostable toilets were seen as most likely communal investments. In the case of the latter, it is interesting to note that little relationship existed between perceptions of improving the distribution of toilets and reduced incidents of

[20] 12.5ha of ShNP will need to be inundated for the development of the Dhap reservoir (0.03% of habitat loss) according to ADB (2013).

water-borne illness. As raised in one focus group, there is a difference between merely distributing toilets and actually changing behaviour to reduce faecal contamination of waterways.

> "*The bigger problem is for people to understand the connection between illness and poor waste management. If they* [referring to Sundarijal VDC] *distribute eco-san toilets, maybe some people will start using them, but not everyone."* (Women of Mulkharka)

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It is noteworthy that targeted payments are not scored among the highest due to their mediocre influences on improving the greatest range of fundamental objectives, even when minimal compensation between criteria is permitted. Surprisingly, the optimisation of payments for the least well-off was not viewed as particularly egalitarian or fair. From focus group discussions, it was revealed that local user groups were better placed to determine how the needs and constraints of individual households could be met or overcome rather than accepting the external assumptions of what may be seen as poverty reducing or well-being enhancing.

> *"We should choose representatives in our village who will look after our interests and would decide through village meetings who would receive payments for what purpose and where the whole village can benefit."* (Women of Khatri Tole, Okhreni village)

> "*...local user groups should have greater involvement in discussions with experts over payment choice and distribution."* (Males of Chilaunegaun village)

This view was clarified in the open meeting where community representatives expressed concern over the fair distribution of incentives in contrast to the perspective shared by some service beneficiaries (e.g. KUKL, NEA-Sundarijal) for cost-efficient or effective soil and water quality management. In terms of involvement in the management or negotiation process, local communities have made it clear that decentralising decision-making related to livelihood choice and the rights to manage and allocate payments according to local perceptions of erosion-vulnerable areas, well-being constraints of individuals and the wider community are essential components of a management strategy for improving soil and water quality. This priority is reflected in the higher ranking alternatives of C3 and A3. Both alternatives also suggest that a

combination of local control in decision-making combined with external support in terms of financial resources and ecological expertise as being the ideal strategy for expanding capability sets for ShNP communities.

Such a strategy presupposes an institutional arrangement in which local user groups are formally recognised and, directly contribute to land-use decisions taken at higher levels of ShNP management. The proposed buffer zone management plan for the Park provides a pragmatic institutional framework for this goal. As the chief warden of ShNP mentioned, the role of NGOs who are otherwise promoting PES-like arrangements in the Park should focus their awareness-raising efforts on bridging the gap between land-use stewardship and opportunities to overcome barriers to livelihood and well-being improvement (Bhattarai, *pers. comm*.). Figure 5.1 illustrates the proposed buffer zone management framework associated with alternative C3 and at what steps the various fundamental objectives raised as the criteria set in the SMCE could be achieved through this framework.

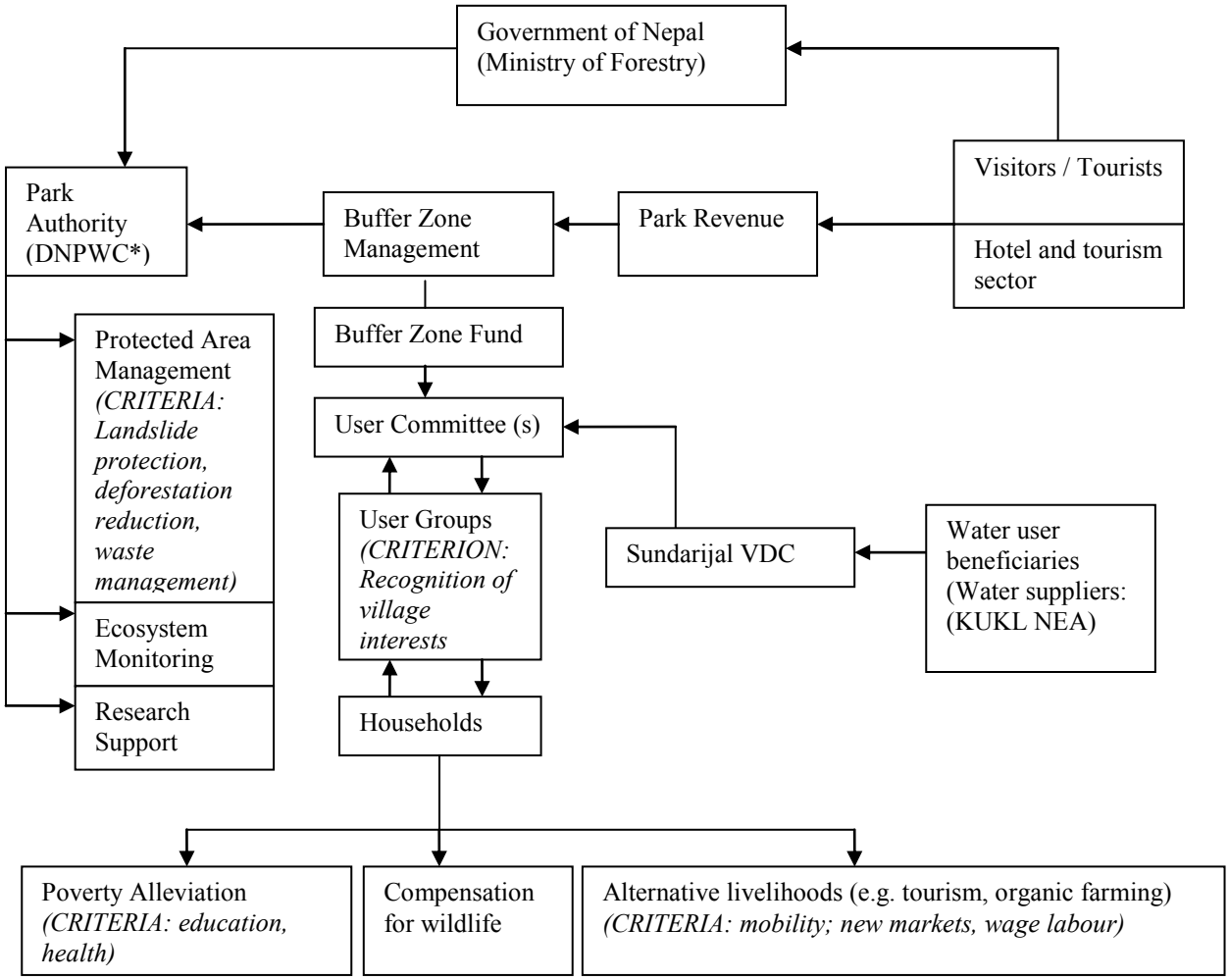

Figure 5.1: Planned buffer zone management plan for ShNP including potential for incorporating improvements in capabilities (defined in the criteria set raised by upstream villagers) characterising alternative C3. *The Department of National Parks and Wildlife Conservation.

5.5.2. Social evaluation

The configuration by which social actors are merged according to the differences or similarities of their preferences for particular alternatives on the set of criteria is represented through a social impact matrix as presented in Table 5.7. In the determination of a similarity matrix between actors for the alternatives being considered, we consider the following equation:

$$\Lambda = \{\lambda_p\},\ p = 1, 2 \ldots., P, \text{ with } \sum_{p=1}^{P} \lambda_p = 1 \qquad (5.5)$$

as the vector of weights attached to each of P social actors, where A is the total set of alternatives and λ a weight attached to each social actor reflecting their relative importance for the decision (Gamboa and Munda, 2007). The application of the Minkovsky distance (d_{ij}) based on the linguistic distance between group i and group j is determined and aggregated to obtain clusters of actors' preferences in a dendrogram as shown in Figure 5.2 (Gamboa and Munda, 2007, NAIADE Manual, 1996).

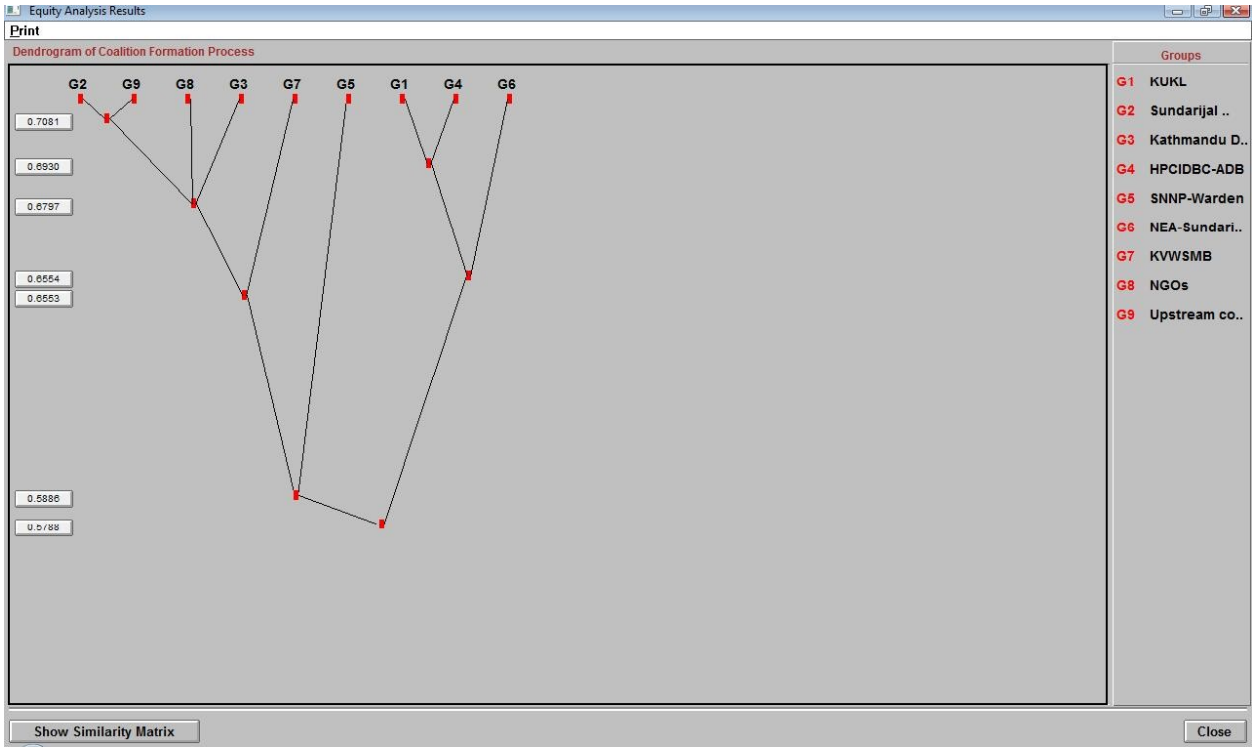

Figure 5.2: Dendrogram forming coalitions of social actors

Social Groups	Unit Number	A1	A2	A3	B1	B2	B3	B4	C1	C2	C3
KUKL	G1	Moderate	Good	Good	More or Less Good	More or Less Good	Good	Moderate	More or Less Bad	More or Less Bad	More or Less Bad
Sundarijal VDC	G2	Bad	Very Bad	Moderate	Good	More or Less Bad	Good	More or Less Good	More or Less Good	Good	Good
Kathmandu DDC	G3	More or Less Bad	Bad	More or Less Good	More or Less Good	Moderate	More or Less Good	More or Less Good	Moderate	More or Less Good	More or Less Good
HPCIDBC / ADB	G4	More or Less Bad	More or Less Good	Perfect	More or Less Good	More or Less Good	Good	Moderate	More or Less Bad	Moderate	Bad
ShNP Warden	G5	Bad	Extremely Bad	More or Less Good	Moderate	More or Less Bad	Good	Moderate	More or Less Bad	More or Less Bad	Very Good
NEA-Sundarijal	G6	Moderate	More or Less Good	Good	Good	More or Less Good	Moderate	Moderate	Moderate	Moderate	Moderate
KVSWMB	G7	More or Less Bad	Bad	Moderate	Moderate	More or Less Good	More or Less Good	Moderate	Good	Very Good	Good
NGOs	G8	Very Bad	Extremely Bad	Bad	Very Good	More or Less Good	Good	Good	More or Less Good	Good	Good
Upstream communities	G9	Bad	Very Bad	More or Less Bad	More or Less Good	Moderate	Good	More or Less Good	Good	Good	Very Good

Table 5.7: Social impact matrix of social actors' perceptions towards the management alternatives considered

At each node, the variance of opinions of actors within each cluster is further distinguished, while a commonality of preference between the actors within a cluster is recognised. In this manner, efforts to identify overlaps and points of contention in understanding elements of the decision context can facilitate the establishment of an appropriate negotiation space between social actors as well as reducing the potential for conflicts associated with miscommunication over time (Vignola et al., 2012). A caveat of describing similarities and points of contention between social actors relates to the difficulty of capturing representative organisational preferences when interviews and meetings characterise the preferences of only a single individual within the organisation. As such, the commonalities do not reflect personal relationships or histories of contention and collaboration between these actors, which is more likely to influence the formation of coalitions than what is presented here (Gamboa, 2006a).

From Figure 3, two distinct clusters of actors or 'coalitions' and a single actor can be identified which defines the initial negotiation context for involved parties to coordinate their goals. Downstream beneficiaries including the hydropower facility in Sundarijal town, Kathmandu Valley's private-public water supplier, and the coordinating agency for the Bagmati River Basin form one coalition (C_1), while the views of upstream communities, local and regional governments form another (C_2). Both coalitions are defined by medium-high credibility based on their credibility scores of 0.6554 and 0.6553 respectively. The ShNP warden joins C_2 at a lower credibility score of 0.5886 since affinity to the interests of upstream communities and governments is tempered by the warden's approval of the planned activities advocated by coalition C_1. Particularly noteworthy is the close relation of values shared between upstream communities and the Sundarijal VDC government, forming a higher level cluster within C_2 of high credibility (0.7187). This information contradicts perceptions emerging from the focus groups with villagers that VDC objectives differ from their interests. Hence presenting the results of the dendrogram in a clear and transparent manner can aid in improving trust between these groups.

An additional feature of SMCE is the emphasis it places on social justice in terms of the impacts that particular alternatives could have on marginalised social actors or those who traditionally have had less power of self-determination. Specifically, SMCE allows the use of veto power to increase the weight of particular social actor groups or coalitions of actors to influence the

ranking of alternatives so as to best satisfy perceptions of justice (Munda, 2005). The application of veto power implies an explicitly ethical posture on the expectations of equitable decision outcomes and thus requires a transparent process of deliberation prior to consideration (Gamboa and Munda, 2007). As described below, the raised concern over procedural justice in the construction of alternatives from the perspective of upstream communities implies a possible role of affording veto power to ensure a more equitable balance of decision power.

In order to ensure the robustness of the evaluation, we undertook a sensitivity analysis in altering the preference thresholds attributed for each criterion in comparing alternatives. Alterations of about 50% change were made both above and below the original thresholds provided (from Table 5.5). The analysis revealed no changes in the performance of the alternatives. Alternatives C3, A3, and C2 remained the first, second and third ranking alternatives respectively. We can claim therefore that the credibility of preferences for the alternatives is more rigorous in the face of the uncertainty that might arise in comparing alternatives against the criteria set analysed.

5.6. Discussion

5.6.1. The extent of procedural justice in recent water resource management in the Kathmandu Valley

The concept of participation goes beyond a process of consultation to include procedural justice in terms of how well engaged social actors are in the design and functioning of a project (Fisher, 2013). In the case of institutions for water management in the Sundarijal catchment and the broader Bagmati River basin, the ADB provided US$15 million in 2003 for institutional reform in the water sector and shortly afterwards the Government began the process of privatising water supply institutions in the Kathmandu Valley (Maskey, 2008). The Nepal Water Supply Corporation (NWSC) was a fully government operated entity responsible for planning, investing, operating and managing water utilities in the Valley. With institutional reforms, this agency was split into KVWSMB and KUKL, the latter of which is a for-profit utility designed to price water according to full cost recovery (Maskey, 2008). ADB was involved in appointing early directors to KUKL as a condition to the loan provided for institutional reform. While severe water shortages were commonplace, with demand outstripping supply prior to institutional reform,

these have worsened in the last decade as a result of financial mismanagement and internal controversies related to frequent turnover of key managerial positions (Bhandari, Water and Energy Users' Federation-Nepal (WAFED), *pers. comm.).* In particular, unequal distribution of water according to ability to pay is further influencing water availability in the Valley (Arun Shrestha, NETIF-Nepal, *pers. comm.).*

Meanwhile, a controversial project funded largely by the ADB known as the Melamchi Water Supply Project, proposed for nearly three decades, was initiated in the year 2000, involving the construction of a nearly 30km tunnel to divert 170 million litres of water per day from the Melamchi River for consumption in the Kathmandu Valley. The promotion of this project has paid little attention to the fact that 60% of the city's water supply derives from groundwater sources through public wells that remain poorly managed and in a state of disrepair (Bagmati Action Plan, 2008; Shrestha et al., 2012). A focus on repairing supply infrastructure and demand management by promoting behaviour to aid in the recharge of aquifers has been sidelined in favour of new infrastructure projects that meet the objectives of foreign donors and contracting companies as well as the Government eager to tap into substantial funds.

Incidentally, the Melamchi project was suspended in 2012 after the foreign contractor terminated the contract, under pressure partly from upstream communities who could not ensure that their water rights to the Melamchi River would be secured. For the indigenous Majhi ethnicity, the ability to fish, critical to their food security, was in danger of being lost. A formal complaint was submitted by WAFED to ADB citing that local communities did not have access to information relating to the results of the EIA, assessments of options, and lending conditions. Documents that were released were not in local languages and were only accessible online. Moreover, active participation from local communities was absent in undertaking the EIA. In terms of environmental impacts, the Melamchi project has disturbed local community-managed forests raising concerns over future rights of management. The diversion of the river has impacted food security and biodiversity (Siwakoti, 2004). Finally, the 'Social Uplift Program' heralded by ADB was much criticized by local communities as failing to respond to perceived priorities, constraints, or even to a democratic process in determining end goals.

In the case of the Sundarijal catchment, the ADB has failed to compare alternative considerations to their proposal (alternative C3) that may be derived through a more deliberative process of the opportunities and challenges of actors across the Bagmati River basin. For example, judicious use and regulation of groundwater, rainwater harvesting, and improved stewardship of existing surface water sources seems a logical first step in maintaining the sustainability of existing water supply services, yet has been ignored in project planning (Shrestha et al., 2012). In other words, there is no indication from the ADB's proposal as to why the construction of reservoirs in ShNP overrides all other possibilities in terms of realising net benefits, regardless of how they may be measured.

5.6.2. Procedural justice and the SMCE

Table 5.8 presents the rankings of alternatives from the perspective of each coalition of social actors. While it can be said that alternative C3 is technically robust, it may generate controversy from a social conflict perspective. This is because C3 is ranked lowest among ranked alternatives representing coalition C_1. Equally, the second highest ranked alternative (A3) is ranked lowest by coalition C_2. Accordingly, social compromises between these coalitions may lie in other alternatives that are ranked in middle positions by all social actors. Of these, alternatives B1 and B3 are ranked highest by both coalitions and hence would be the only interventions that are both technically acceptable and minimise social conflicts. It is clear from the analysis of the ranking sets of coalitions that alternative A1 (status quo) is not acceptable by any actor. From a purely technical perspective different alternatives become more acceptable when a consideration of social dynamics and inherent power asymmetries between coalitions of actors is incorporated in the analysis. This trade-off between technical performance and social conflict is characterised by diverging perceptions of justice over the same ecosystem service by different actors and hence cannot be resolved without an inclusive and transparent deliberative process to expound on these views in light of the decision context (Table 5.9).

Coalition	First	Second	Third	Fourth	Fifth	Sixth	Seventh	Eighth	Ninth	Tenth
C_1	A3	A2	B1	B2	B3	B4	C2	A1	C1	C3
C_2	C3	C2	B3	C1	B1	B4	B2	A3	A1	A2

Table 5.8: Rankings of two coalitions of social actors

Social Actor Coalition	Claim on Water quality ecosystem services	Principle of Justice
C_1	Maximise return on investment in terms of privatising water supply for Kathmandu Valley water users	Utilitarian: Market justice
C_2	Depend directly on the ecosystem components upstream that influence water quality downstream	Libertarian

Table 5.9: Diverging realisations of justice by the two major coalitions of social actors involved in the management of water quality services deriving from the Sundarijal catchment. Here 'Libertarian' is defined as 'the state of being unconstrained by other persons from doing what one is able to do" (p. 12, Sterba and Peden, 1989).

Perhaps the strongest indication of the robustness of the rankings is the lack of rank reversals observed by considering both varying compensation between criteria and alterations in preference thresholds between alternatives for a given criterion. This indicates that even with uncertainty in preference relations, consensus increases. The robustness of the ranking also illustrates that the trade-offs between prioritising highly scoring criteria and the resulting outcomes this may have on the set of considered alternatives has been socially recognised and accepted as legitimate. Furthermore, flexibility in terms of how actors perceive changes in the scores of a given alternative against a criterion still results in a confident technical ranking of alternatives.

5.6.3. The viability of PES for capabilities

The determination of a comprehensive criteria set to meet fundamental ends objectives has recognised *personal* barriers referring to skills sets, education and good health, *physical* barriers relating to access or mobility, and *societal* barriers relating to costs and having a voice to influence decisions. However, as evident in the determination of management strategies for upstream communities, linking functioning sets to 'do and be' with ecosystem service delivery remains particularly constrained by societal barriers. Hence, for management alternatives such as targeted PES schemes that prioritise particular social or environmental outcomes (alternatives B1-B4), environmental preservation (e.g. alternative A2) or loan-based conditions (alternative A3), it is critical to distinguish between adopting technical features that reflect accountability to external or more powerful actors versus accountability to all social actors (Fisher, 2013). While we do not purport that power asymmetries should be eliminated, they ought to at minimum be recognised. Such recognition requires that legitimacy and procedural justice take a core role in how management strategies are conceptualised prior to arranging loans for outreach or awareness-raising on a particular policy decision (Corbera et al., 2007).

A policy mechanism such as PES that seeks to bridge human activity with environmental stewardship requires a political economic analysis as to whether a socially-just contract can actually be implemented. Such an assessment underscores a broader discussion on the significance of unequal power relations between social actors and the extent to which marginalised upstream households can participate in decision-making regarding their own affairs (Sikor, 2013). The emphasis on institutional and political feasibility is fundamental to the capabilities approach since the enhancement of freedom to 'do and be' hinges on the real experiences or evaluative judgements of individuals rather than political objectives (Ballet et al., 2013; Sen, 2010). A challenge for the capabilities approach lies in addressing the static nature of evaluative judgements. While perceptions of capabilities space may be shared by future generations, these cannot be dictated by current generations. Hence a negotiation of payments that reflect deprivations in capabilities would be a dynamic process that would require a renewed deconstruction of value judgements through ongoing social inclusivity in order to account for

changing socio-economic pressures, environmental conditions and individual or societal constraints.

5.7. Conclusion

The decision to apply a PES scheme to address an otherwise complex socio-ecological problem requires an explicit consideration of justice in terms of the plural values of social actors that link the objectives of ecosystem service stewardship with improvements in well-being. The SMCE identified highest ranking management alternatives according to the full set of valued criteria and revealed the trade-offs between a technical evaluation and the social equity implications in comparing alternatives among separate coalitions of social actors and their interests. Despite strong concordance for alternatives which build upon buffer zone management and decentralised decision-making for devising resource management rules, several of the PES payment alternatives were illustrated to be more socially acceptable once the risk of social conflict over a divergence of interests was revealed. This research is the first to use social multi-criteria evaluation as an alternative valuation method to compare PES and non-PES management alternatives to address a socio-ecological conflict over water quality services. Unlike other multi-criteria methods, cost-benefit analysis, systems dynamics modelling or other policy evaluation techniques, SMCE accounts for the diversity of value systems under weak-comparability, that is, without reducing the contribution of diverse values through measurement standardisation. Accordingly, it is best suited in conflicts of high uncertainty, power differentials between actors, inequality in purchasing power, and rapidly evolving underlying drivers of system change. Such contexts often characterise socio-ecological conflicts where PES incentives are proposed to improve social and environmental outcomes

The elicitation of criteria in the SMCE was framed in defining payments according to a justice perspective known as the capabilities approach. Through this approach, PES payments among other alternatives were designed according to recognised deprivations in ‘doing and being’ what is of value in life. In this manner, the choice of management strategy best reflects justice according to individual needs and perspectives as opposed to an external assessment of a ‘just state of affairs’. Thus, the payment designs are scored in the SMCE in terms of the extent to

which they satisfy capability sets that are recognised as deprived by potential service providers. In this manner, advocates of PES-like approaches can better align negotiated incentives to the inseparable social and natural processes which co-produce ecosystem services of value. The consideration of ends objectives of policy interventions as capability improvements through stewardship of ecosystem services can aid in enhancing overall well-being insofar as pluralistic considerations of justice are incorporated in decision-making.

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5.9. References

Alix-Garcia, J., de Janvry, A., Sadoulet, E., 2008. The role of deforestation risk and calibrated compensation in designing payments for environmental services. Environment and Development Economics 13, 375-394.

Asian Development Bank 2013. Nepal: Bagmati River Basin Improvement Project. Project Administration Manual. Kathmandu, Nepal. http://www.adb.org/projects/documents/bagmati-river-basin-improvement-project-pam

Bagmati Action Plan 2008. Submitted by: Joint Venture of Astra Development Network Pvt. Ltd., Geospatial Systems Pvt. Ltd., Innovative Solution Pvt. Ltd. http://www.ncf.org.np/upload/files/1132_en_1310646546bagmati_action_plan.pdf

Ballet, J., Koffi, J-M., Pelenc, J., 2013. Environment, justice and the capability approach. Ecological Economics 85, 28-34.

Bennett, E.M., Peterson, G.D., Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. Ecology Letters 12 (12), 1394-1404.

Bookchin, M., 1985. Ecology and Revolutionary Thought. Antipode 17 (2-3), 89-98.

Brouwer, R., Tesfaye, A., Pauw, P., 2011. Meta-analysis of institutional-economic factors explaining the environmental performance of payments for watershed services. Environmental Conservation 38 (4), 380-392.

Corbera, E., Brown, K., Adger, W.N., 2007. The Equity and Legitimacy of Markets for Ecosystem Services. Development and Change 38 (4), 587-613.

Daly, H.E. 2007. Policy, possibility, and purpose. In: Daly, H.E. (Ed.). Ecological Economics and Sustainable Development: Selected Essays. Cheltenham, UK. Edward Elgar. Pp. 215-227.

De Marchi, B., Funtowicz, S.O., Lo Cascio, S., Munda, G., 2000. Combining participative and institutional approaches with multicriteria evaluation: an empirical study for water issues in Troina, Sicily. Ecological Economics 34, 267-282.

Duraiappah, A., 2011. Ecosystem Services and Human Well-being: Do Global Findings Make Any Sense? *B*ioScience 61(1), 7-8.

Farber, S.C., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecological Economics 41 (3), 375-392.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K., Burgess, N., 2010. Common pool resource management and PES: Lessons and constraints for water PES in Tanzania. Ecological Economics 69 (6), 1253-1261.

Fisher, J., 2013. Chapter 2: Justice implications of conditionality in Payments for Ecosystem Services: A case study from Uganda. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Functowicz, S.O., Ravetz, J., 1994. Uncertainty, complexity and post-normal science. Environmental Toxicology and Chemistry 13, 1881-85.

Gamboa, G., 2006a. Social multi-criteria evaluation of different development scenarios of the Aysén region, Chile. Ecological Economics 59 (1), 157-170.

Gamboa, G., 2006b. Social Multi-Criteria Evaluation in practice: Two real-world case studies. Doctoral Thesis. Universitat Autònama de Barcelona. Barcelona, Spain. p. 23.

Gamboa, G., Munda, G., 2007. The problem of windfarm location: A social multi-criteria evaluation framework. Energy Policy 35 (3), 1564-1583.

Garmendia, E., Gamboa, G., 2012. Weighting social preferences in participatory multi-criteria evaluations: A case study on sustainable natural resource management. Ecological Economics 84, 110-120.

Garmendia, E., Gamboa, G., Franco, J., Garmendia, J.M., Liria, P., Olazabal, M., 2010. Social multi-criteria evaluation as a decision support tool for integrated coastal zone management. Ocean and Coastal Management 53 (7), 385-403.

Garmendia, E., Pascual, U., 2013. Chapter 8: A justice critique of environmental valuation for ecosystem governance. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. Environmental Management 45, 488-501.

Groom, B., Grosjean, P., Kontoleon, A., Swanson, T., Zhang, S., 2010. Relaxing Rural Constraints: a 'Win-Win' Policy for Poverty and Environment in China? Oxford Economic Papers 62 (1), 132-156.

Gregory, R., Keeney, R.L., 1994. Creating policy alternatives using stakeholder values. Management Science 40 (8), 1035-1048.

Jacob, A.L., Vaccaro, I., Hartter, J., Chapmans, C.A., 2008. Integrating landscapes that have experienced rural depopulation and ecological homogenization into tropical conservation planning. Tropical Conservation Science 1 (4), 307-320.

Karn, P.K., 2008. Making Payment for Environmental Services (PES) work: A case study of Shivapuri National Park, Nepal. International Union for the Conservation of Nature. Kathmandu, Nepal.

Kosoy, N. and Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69 (6), 1228-1236.

Marshall, G.R., 2013. Transaction costs, collective action and adaptation in managing complex social-ecological systems. Ecological Economics 88, 185-194.

Martínez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecological Economics 26, 277-286.

Maskey, N., 2008. Investing in Ecosystem Services: Opportunities and Challenges for Shivapuri National Park, Nepal. MSc Thesis. Lund University. Lund, Sweden.

Millennium Ecosystem Assessment (MA), 2005. *In*: Hassan, R., Scholes, R., Ash, N. (Eds.). Ecosystems and Human Well-Being: Current state and Trends, vol. 1. Island Press, Washington, D.C.

Mitchell, R.K., Agle, B.R., and Wood, D.J., 1997. Toward a Theory of Stakeholder Identification and Salience: Defining the Principle of Who and What Really Counts. The Academy of Management Review, 22 (4), 853-886.

Munda, G., 2004. Social multi-criteria evaluation (SMCE): methodological foundations and operational consequences. European Journal of Operational Research 158 (3), 662-677.

Munda, G., 2008. *Social multi-criteria evaluation for a Sustainable Economy*. Springer-Verlag. Berlin-Heidelberg.

Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., et al., 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. Conservation Letters, 6 (4), 274-279.

NAIADE Manual, 1996. Joint Research Centre of the European Commission. Ispra, Italy.

Norgaard, R.B., 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. Ecosystem Services. Ecological Economics 69, 1219-1227.

Nussbaum, M.C., 2003. Capabilities as Fundamental Entitlements: Sen and Social Justice. Feminist Economics 9 (2-3), 33-59.

O'Neill, J., 2001. Property, Care, and Environment. Environmental and Planning C. 19, 695-711.

Pascual, U., Muradian, R., Rodriguez, L.C., Duraiappah, A., 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics. 69, 1237-1244.

Polishchuk, Y., Rauschmayer, F., 2012. Beyond "benefits"? Looking at ecosystem services through the capability approach. Ecological Economics 81, 103-111.

Robeyns, I., 2006. The Capability Approach in Practice. Journal of Political Philosophy 14 (3), 351-376.

Rosa, H., Kandel, S., Dilmas, L., 2004. Compensation for environmental services and rural communities: Lessons from the Americas and key issues for strengthening community strategies. International Forestry Review 6 (2), 187-194.

Scerri, A., 2012. Ends in view: The capabilities approach in ecological/sustainability economics. Ecological Economics 77, 7-10.

Sen, A., 2010. *The Idea of Justice*. Harvard University Press, Cambridge.

Sikor, T., 2013. Chapter 1: Introduction: Linking ecosystem services with environmental justice. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Siwakoti, G., 2004. "Donors Rejection of Governance and Human Rights: Two Case Studies of Hydropower and Water Supply Projects in Nepal." The Reality of Report.

Stagl, S., 2007. Emerging methods for sustainability valuation and appraisal- SDRN Rapid Research and Evidence Review. London: Sustainable Development Research Network. www.sd-research.org.uk

Sterba, J.P., Peden, C., 1989. *Freedom, Equality and Social Change: Issues in Contemporary Social Philosophy*. The Edwin Mellen Press. Lewiston, NY.

Van Hecken, G., Bastiaensen, J., 2010. Payments for Ecosystem Services: justified or not? A political view. Environmental Science and Policy 13, 785-792.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69 (6), 1245-1252.

Vignola, R., McDaniels, T.L., Scholz, R.W., 2012. Negotiation analysis for mechanisms to deliver ecosystem services: The case of soil conservation in Costa Rica. Ecological Economics 75, 22-31.

Wätzold, F., Drechsler, M., 2005. Spatially Uniform *versus* Spatially Heterogeneous Compensation Payments for Biodiversity-Enhancing Land-Use Measures. Environmental and Resource Economics 31 (1), 73-93.

Connecting text

The use of social multi-criteria evaluation has illustrated how complex and uncertain decisions involving plural value-based criteria can be assessed through an inclusive and deliberative process in order to fairly evaluate management alternatives. In this way, procedural justice is emphasized in legitimising outcomes. PES payments must also be subject to deliberation and procedural scrutiny, since the introduction and distribution of incentives (whether monetary or not) is not a politically neutral decision. Moreover, the type and intended purpose of the set of incentives delivered emerges both from a process of negotiation, the needs and aspirations of ecosystem service providers and beneficiaries, and crucially the interaction of social norms with rational self-interest in a particular context. It is this latter point which requires closer examination since PES conditionality makes the assumption that economically efficient incentives will always induce land users to behave in desired ways.

In exploring the behavioural assumptions underpinning PES schemes, the following chapter introduces a range of incentive types and a control to examine how incentive provision for managing collectively owned irrigation infrastructure influences behaviour to contribute labour. This research takes place at a PES pilot location in the Kyrgyz Republic. Piloting a PES mechanism within this setting provides a unique opportunity for understanding how incentives interact with a deep tradition of collective resource management. Appreciating the interaction of PES incentives with social norms and informal institutions along the frontiers of economic valuation of the environment may also provide useful insights in discerning the implications for PES conditionality in more or less market-integrated societies. In examining the behavioural assumptions of PES through existing processes of collective action, this research addresses the general objectives of the thesis as well as specific objectives 7 and 8. This chapter has been accepted for a special issue on the behavioural and ethical responses to economic incentives for resource management. It will be submitted to the journal *Development and Change* (Authors: Kolinjivadi, V., Charré, S., Kosoy, N., and Adamowski, J.).

Chapter 6

Economic experiments for collective action in water resource management in the Kyrgyz Republic: Implications for Payments for Ecosystem Services

6.1. Abstract

Payments for ecosystem service (PES) schemes have become increasingly popular in attempting to promote ecological stewardship and conservation behaviour through provisioning of economic incentives often as market-inspired transactions. The scholarship and empirical application of PES schemes has tended to focus on contract design and conditionality of payments rather than more fundamentally examining whether incentives do indeed influence behaviour in desired ways. In this paper, a set of incentive framed treatments are introduced to existing institutions promoting otherwise unpaid collective action for common pool resource management in order to explore the effects of incentive provision on the propensity to participate in the maintenance of collectively owned irrigation canals. The experiments take place in the Kyrgyz Republic, characterized by the transition of nomadic collective traditions, reinforced by subsequent Soviet rule, to advances towards democracy, market integration, and the first PES pilot in the country. We reveal the close interplay between the framing of incentives, the influence of village leaders mobilizing collective activity and social norms of reciprocity, trust and enforcement in mobilizing collective work. Each framed experiment exhibits a unique configuration of 'I' 'We' and other regarding rationalities, providing useful implications for this new frontier of PES implementation.

Keywords: Payments for Ecosystem Services; common pool resources; behavioural economics; incentives; reciprocity

6.2. Introduction

In his book *Predictably Irrational*, behavioural economist Dan Ariely provides an anecdote musing on how awkward it would be to offer payment to granny for her lovely Christmas dinner

at the one time of the year when the entire family gets a chance to come together. He argues that people often adopt a unique behavioural framing under market norms and that the inclusion of monetary incentives in an otherwise social setting can disrupt pre-existing social norms. Neoclassical economic theory posits that people will always respond rationally in maximizing self-interest when deciding upon a particular course of action. (Yanez-Pagans, 2013).

Such logic has been harnessed to influence the behaviour of land-use managers to improve perceptions of ecosystem services for wider society through 'Payments for ecosystem services' (PES) as an presumably more efficient manner than strict regulation (Engel et al., 2008; Wunder, 2008). This policy tool is gaining traction within the conservation and development field and is being piloted in locations across the Global South despite challenges of ingrained political relationships between social actors and pre-existing norms over managing common pool resources which make purely self-interested behaviour difficult to observe in practice (Muradian et al., 2010; Vatn, 2010; Van Hecken et al., 2012). While incentives can be a powerful influence on behaviour (Fehr and Falk, 2002), a salient research agenda requires a better understanding of the circumstances in which incentives can facilitate or discourage behaviour and the types of incentives that resonate with nuanced combinations of social and market norms characteristic of a particular context.

Numerous studies have highlighted and criticized the inaccuracy of the canonical prediction of *homo economicus* as the perfectly rational individual whose every behaviour is conducted by a careful calculus of costs and benefits (in money) to personal utility (Nyborg, 2000; Fehr and Folk, 2002; Bowles, 2008). Yet it remains unclear what constellation of social and economic institutions best shape attitudes and behaviour towards fair or equitable outcomes in a given setting (Henrich et al., 2005). Co-evolutionary theory posits that genes and human culture are linked and made evident in cumulative processes of social learning passed down over generations, ranging from products (e.g. technology) to patterns of social interaction (e.g. institutions) (Tomasello et al., 2005; Gintis et al., 2008). Through social learning, human behaviour is hence malleable to the institutions and norms unique to locally evolved cultural, social and physical environments (Henrich et al., 2005). Such insight has profound implications

when culturally learned practices of natural resource management respond to market-based overtures.

With the exception of a few recent studies, the PES literature has been preoccupied with getting 'prices right' to reflect values for ecosystem services or compensating foregone benefits from land-use change rather than examining the primacy of the assumption that incentives indeed result in behavioural change (Yanez-Pagans, 2013; Kerr et al., 2012; Narloch et al., 2012). Intrinsic motivation of care and responsibility for those land-users already engaging in ecologically sound land-use practices is dismissed in favour of strict adherence to the conditions of contracts by assuming self-interested behaviour (Salzman, 2005; Clements et al., 2010). In certain instances, this fragmentation of induced land-use promotion panders to the self-interest of those with power (e.g. resources, land, status, voice) resulting in significant inequity in the distribution of benefits from payment and ultimately the erosion of collective action norms and even the risk of social unrest (Corbera et al., 2007; Vatn, 2010; Matulis, 2013). While payments may be seen as equitable under this context in some situations (Escobar et al., 2013; Ricó García-Amado et al., 2013), this is not always the case. Where social norms exist to govern individual behaviour for collective resources, the introduction of economic incentives may have unintended consequences which could either reinforce norms for resource stewardship (crowding-in) or at worst 'crowd-out' the moral imperative which otherwise lubricates traditional histories of land-use stewardship and/or informal institutions required for collective resource management (Cardenas, 2000; Fehr and Falk, 2002; Vollan, 2008; Kerr et al., 2012).

This research builds upon the literature in better understanding how the provision of incentives influences individual proclivity to contribute to natural resource management. However, our aim is not to compare different incentive treatments to determine which are more or less contextually appropriate for PES implementation. Rather this exploratory study seeks to understand how different signals conforming to the 'I', and 'We' or other-regarding narratives interplay with existing social norms and informal institutions in order to reveal insight for promoting longer-term natural resource management. This study distinguishes itself from previous research in three ways. Firstly, we compare participation in framed field experiments with post-hoc surveys of participants to compare how well the actual behaviour of individuals under non-hypothetical

conditions conforms to stated perceptions. Secondly, our experiments analyse the role of an established mobiliser who encourages individuals to participate in order to distinguish between pre-existing reputational effects and the role of the incentive in influencing participation. Thirdly, we provide recommendations in a setting where PES is being piloted with the aim of improving the design of incentive structures and to assess the extent to which incentives can align social norms with individual gain in improving the delivery of ecosystem services. Given that financial flow predictions for REDD[21] and PES programmes are expected to approach $30 billion annually, there is much to gain in discerning how incentives could aid or hinder environmental management (McDermott et al., 2013). We carry out the analysis in the Kyrgyz Republic, representing both the first Central Asian PES pilot and a unique setting for examining the behavioural implications of incentives for natural resource management.

The following section provides an overview of literature seeking to better understand how incentives influence behaviour. In Section 6.4, the research setting, field experiment and methodology for analysis are described. The results and a discussion on the implications of these findings for PES implementation is presented in Section 6.5, followed by a brief conclusion.

6.3. Modifying behaviour through incentives: Insights for PES

The theoretical foundation of PES is based upon Rational Choice Theory and the Coase Theorem. Both theories predict that the voluntary trade of environmental goods and bads will be allocated efficiently assuming property rights are established and that trading parties are fully informed of their values for service provision and the costs of foregone opportunities (Garmendia and Pascual, 2013). Thus, if incentives reflecting the values of the beneficiaries of specific ecosystem services can compensate or surpass opportunity costs of service providers, the latter will always behave rationally in accepting the contract and engaging in pro-environmental land-use behaviour (Pagiola and Platais, 2007). However, most people adopt cognitive heuristics borne out of daily social interactions reflecting subtle cues of private or social appropriateness of behaviour rather than being supposedly innate rational agents (Henrich et al., 2005; Vatn, 2009).

[21] Reducing Emissions from Deforestation and Forest Degradation

In this section, we identify some ways in which extrinsic incentives can influence the proclivity to engage in collective action, which contradict the theoretical PES logic.

6.3.1. Institutions of 'I' and 'We'

Institutions refer to norms, rules, customs and conventions that guide and shape behaviour and how people interact in social contexts (Vatn, 2005). Classic institutionalists such as Thorstein Veblen posit that such norms and rules are socially constructed. Hence, they do not lie outside the primacy of the human individual but actively shape and provide meaning to individual behaviour. In this manner, institutions critically support human behaviour by providing meaning to complexity through established rationality and formal or informal collective consensus (Vatn, 2009). On the other hand, new institutionalists assume that the individual is autonomous to externally determined constraints in which an individual rationally maximizes her utility within such constraints (Singh, 2013). Within this modality, relationships with other people and the physical environment have a purely instrumental value in facilitating or hindering individual utility maximization (Vatn, 2009).

The rationalities of 'I' (or self) versus 'We' (or social entity) are separate cognitive framings that an individual adopts in which ultimate behaviour proceeds according to both individual interests as well as on a normative guiding principle as to what is socially decent and appropriate in a given situation (Etzioni, 1989). These rationalities are manifested when particular institutions are introduced and entrenched over time. Market-oriented rules conceptualize value in one form, exchange value, which extols calculative rationality and individual utility and presupposes an individual perspective of 'I' (Vatn, 2009). Conversely, endogenous institutions characteristic of established patterns of interaction and social learning shape an individual's behaviour according to 'We'. This may take different forms according to the plurality of social interactions and roles of individuals in a community (e.g. within a family unit or member of a team; as a female, or elder etc.). (Vatn, 2009). Accordingly, the formation of institutions influences individual rationalities for behaving as 'I' or 'We' in a given context and can have profound implications when the ultimate objective of behavioural change is for collective benefit. The question arises as to the viability of adopting an individualist posture to addressing problems inherently defined

by their complex interactions with others and which affect the common interest. In the case of PES, the introduction of incentives is a normative strategy to achieve or induce social benefits in terms of improvements in ecosystem services by appealing to the 'I' rationality of the presumably self-interested individual in order to think as 'We'. The key lies in exploring the interplay of the dual 'I' and 'We' rationalities characteristic of combining market norms in an otherwise social framing of outcomes in order to validate assumed behavioural responses.

6.3.2. The interaction of incentives and 'other regarding' motivations

'Other regarding' motivations refer to behaviour responding to non-selfish objectives and evolve from social interactions that influence individual payoffs (Akçay et al., 2009). In this section, we highlight several of these motivations including decentralized norm enforcement, framing and image motivation, as well as the potential ramifications of introducing extrinsic incentives under these motivations. The first of these, decentralized norm enforcement and informal institutions, have the potential to influence behaviour and maintain high levels of cooperation (Carpenter et al., 2004; Fuster and Meier, 2010). Such social norms function in contributions to the public good, including the protection of the environment. Through processes of self-regulation, free-riders to collection action are sanctioned by those contributing more, especially so in small communities where the actions (or lack thereof) of individuals working for the common good is most visible (Ostrom, 1990). In turn, would-be defectors are less partial to defect in order to maintain good standing and a desire to be liked or respected. (Ariely et al., 2009; Cardenas et al., 2011). Evidence of guilt and shame as the driving mechanism for prosocial behaviour has been empirically determined in a number of experimental public good games in which contribution increased after participants experienced 'disapproval' by other group members (Masclet et al., 2003; Gintis et al., 2008; Hopfensitz and Reuben, 2009; Fuster and Meier, 2010). However, when incentives are provided, this internal sanctioning has been found to disintegrate since free-riders transition from having to make a social 'excuse' for not contributing to a justifiable or excusable individual one. In other words, the shame or guilt experienced for not participating is reduced as free-riders have chosen to forego the incentive (Rodriguez-Sickert et al., 2008). Hence incentives replace the necessity for social norm enforcement since those receiving the incentive for contributing to the common good receive something that free riders do not.

Heyman and Ariely (2004) propose a distinction between two markets- one characterized by social norms of exchange (e.g. gift economies or 'tit for tats') and the other by utility maximization reflected in monetary pricing. Their work dovetails Fiske's (1992) relational theory in which behaviour under social norms is characterized as instances of communal sharing, equality goals or the role of authority as guiding motivations for behavior rather than monetary exchange. They argue that effort in collective action through the use of incentives will result in different interactions depending on the market framing in which an activity is taking place. In a monetary market framing, behaviour is argued to be guided by reciprocity in which effort is directly commensurate with compensation (Fehr and Falk, 2002; Heyman and Ariely, 2004). However, in a social framing, effort may be consistent irrespective of payment such as the tireless effort offered in times of crisis or contributions to a potluck dinner for a group of close friends. Even the mere mention of 'monetary payment' signalled a shift of individual norms from a social to a monetary market setting. They conclude that social-norm based exchanges are fragile to the introduction of extrinsic incentives and suggest that effort-based payment be distinguished from employment-based payment reflecting the social or monetary market relationship that each represents. Hence, the application of PES predicated upon individual payments may cue a signal altering the set of social or monetary market norms that an individual adheres to, thereby influencing the quality and consistency of land-use stewardship associated with the delivery of ecosystem services.

The desire to gain social approval through public exhibition consistent with existing social norms is another powerful motivator influencing behaviour, which exhibits unpredictable interactions in the presence of extrinsic incentives (Fehr and Falk, 2002). Essentially, people want to be seen by others as doing 'good', but by offering incentives, the signal of doing good for others may become one of doing good for self-interested reasons (Bénabou and Tirole, 2006; Ariely et al., 2009). Image motivation reflects the behaviour of an individual in response to the expected payoffs from others' perceptions of their behaviour. Where strong social approval exists for visible contributions to the public good or for public benefit, the introduction of private incentives can lead to social disapproval and reduced contributions to the public good (Ariely et al., 2009). For example, if motivation for behaviour stems from a moral imperative such as the

act of being honest or perhaps a sense of personal responsibility to safeguard the natural environment, then offering extrinsic payments influences the social or self-approval that is otherwise derived from moral deeds (Fehr and Falk, 2002).

6.3.3. Designing PES incentives in light of behavioural economics

Recent critical analysis of PES has stressed the deeply political nature of introducing payments for land-use change, in particular the role of informal institutions, past experiences of conflict and cooperation between involved social actors, as well as power asymmetries reflecting heterogeneity in information access or opportunities to influence outcomes (Vatn, 2010; Kosoy and Corbera, 2010; Muradian et al., 2010; Van Hecken et al, 2012). PES projects worldwide have largely been externally imposed through government-sponsored policies or small-scale projects mediated by non-governmental organizations in response to development aid (McAfee, 2012). While the introduction of incentives is bound to influence behaviour positively or negatively with respects to collection action, the social ramifications of *externally* introducing incentives may not have the same effect as when social groups organically enter and terminate incentive-based agreements on their own accord. In identifying opportunities and challenges for PES implementation in a particular setting, it is vital to identify how endogenous motivation for contributions to collective resources or public goods might harness incentives or less formal institutions to maintain behaviour.

Few studies to date have analysed the possibility that PES incentives might alter existing social norms for natural resource management. Payment design in PES can take a number of forms, ranging from monetary compensation in the form of conservation rent (Pfaff et al., 2006); payment in-kind (Sommerville et al., 2010); payment as the exchange of labour (Turpie et al., 2008), or more informal contractual agreements based on collective resource management principles (Escobar et al., 2013). These payment modes can be characterized by more formalized market framings (e.g. payment as conservation rent or in-kind payment) to more informal negotiated contracts (e.g. 'tit for tat'). Most understanding of the psychological or behavioural implications of incentive provision has relied upon laboratory data or experimental games. While these approaches have the advantage of clearly identifying targeted effects of payment and non-

payment treatments on behaviour, the empirical validity of these experiments remains questionable (Henrich et al., 2005). With respects to PES implementation, real-life understanding of the role of incentives versus existing social norms in prompting the management of ecosystem services is lacking, hence predicating PES incentives on untested assumptions (Vatn, 2010). Yanez-Pagans (2013) examined the impacts of Payment for Hydrological Services (PSAH) in Mexico on cooperative behaviour in common property communities. The author determined that while work effort for those with secure land tenure increased with the introduction of private incentives, they did not induce positive spillover effects on unpaid voluntary work which was otherwise mediated by informal institutions present in the community.

In another recent study carried out by Kerr et al. (2012), framed experiments in communities where PES was being piloted in both Tanzania and Mexico involved offering varying incentive payments to measure collective action responses. These authors determine that incentivisation under the presence of strong pre-existing social norms for cooperation yields insignificant motivation to supplement collective behaviour. They also identify increased effort associated with incentive provision, particularly among those previously unengaged in collective activities. Nevertheless, two main caveats of their approach are evident. The framed experiments of incentive provision were one-shot activities and thus do not take into account social norms of reciprocity associated with repeated interactions or to achieve a collective goal. Secondly, despite the empirical nature of the analysis, the experiments were imposed by the researchers as two hour litter collection activities rather than being embedded within existing or familiar practices of collective action. Accordingly, participants may be more inclined to participate for the novelty of being a part of 'research' rather than for the motivations sought out by the researchers (Pirard and Billé, 2010). We aim to improve upon this design by engaging in repetitive experiments over four weeks in harnessing upon existing collective activities planned in the communities assessed.

We argue that understanding the behavioural economics of PES is vital before making assumptions that conditional ecological improvement tied to payment can be achieved. Given the existence of developed social institutions for collective management in this setting, the following research questions are identified in terms of participation in the maintenance of collectively

owned irrigation infrastructure. It is hoped that insight into these questions might offer recommendations regarding PES feasibility within the villages where these framed experiments were carried out:

1. Does participation differ between two incentivised and two non-incentivised collective action treatments carried out on a weekly basis over four weeks? Does participation change within a single treatment over the four-week period?

2. To what extent do individual, communal or no incentives at all better reflect perceived proclivity to contribute in the collective endeavour as procured by participants in the framed experiments?

3. What other motivating variables can be identified from the framed experiments that may interact with incentive provision in unpredictable ways?

6.4. Research Context and Methodology

6.4.1. The 'ashar' tradition of voluntary labour for collective resource management

Early democratic reform in the Kyrgyz Republic following independence has allowed for greater decentralisation of power than surrounding countries. As a result, civil society organisations and foreign-aid agencies funding projects in rural development, water management and nature conservation have placed much interest in the Kyrgyz Republic in achieving their goals in Central Asia (Earle et al., 2004). Despite being forced to settle into *kolkhoz* or collectivised farming during the Soviet-era, traditional institutions of social organisation that characterised the pre-Soviet nomadic lifestyle of the populace continue to be practiced. These highly developed rules are known as *ashar*[22] and involve voluntary labour for collective resource management (Fuhrmann, 2006). The most important role of *ashar* is its capacity building and empowerment potential within the community in fomenting collective action for projects that would benefit the whole community (Earle et al., 2004). During Soviet rule, a similar pattern of voluntary labour

[22] Also known as 'assar' in Kazakhstan or 'hashar' in Uzbekistan

for the maintenance of public amenities was espoused in weekly *subbotniks* which served to reinforce the *ashar* traditions of Central Asia. Having persisted during the Soviet-era to build and repair village houses, bridges and irrigation canals, the form that *ashar* currently takes has evolved into the general maintenance of public infrastructure (Fuhrmann, 2006). The work is carried out principally by men usually over the course of a day and involves as many able-bodied individuals available to contribute. *Ashar* is typically directed by an experienced and respected individual who is appointed to direct important village decisions or activities. As such, *ashar* participants often have little say in the choice of activities or in managing the choice or planning of the work schedule. Often, the *ashar* is announced by the *Ayil Bashi*[23]calling on villagers to contribute their labour at a specific time and date. Thus, despite the community-empowerment vision of the *ashar*, its success for collective resource management depends on villagers obeying orders from above (Furhmann, 2006).

6.4.2. Study site

The Chon-Aksuu catchment is located along the southern slope of the Kungei Ala-Too mountain range separating the nations of Kazakhstan and the Kyrgyz Republic. The catchment traverses through both alpine and low-mountain ecosystems and drains into the 6,236km^2 Lake Issyk-Kul, the world's second largest alpine lake. Along the shores of the lake are a number of villages of agro-pastoralists engaging in irrigated agriculture but also follow a seasonal migration to higher pastures in summer.. The grazing patterns upstream combined with forest degradation are increasingly impacting water quality downstream as a result of soil compaction and subsequent loss. The Central Asian Regional Environmental Centre (CAREC) is an inter-governmental organising specialising in environmental sustainability and natural resource management projects in the region. They have piloted a PES programme in collaboration with the local federation of water users associations, the village pasture committee, the forestry enterprise's local office and village-level governments with the aim of scaling up project outcomes to the national level. The established PES scheme is based on a labour in-kind reward provided annually by downstream water users to upstream ecosystem managers. The choice of an in-kind reward was guided by

[23] Elected but unpaid village leader who reports to the *Ayil Okmotu* (e.g. the lowest form of local government, recognized as an organ of the State)

local expectations that financial payments would have been hampered by mismanagement. It is expected that conservation activities implemented through this reward scheme will reduce ecosystem degradation upstream and subsequently improve water quality downstream.

6.4.3. Framed experiment

The research took place in the four villages of Kojoyar, Grigorievka, Kashat and Temirovka (Figure 6.1) from late September to late November 2013. Each village is included in the pilot PES scheme as representatives of the water-user association (WUA) and as beneficiaries of water quality services provided upstream. Additionally, the villages are similar in terms of their population, socio-economic makeup, and distance from major urban centres (Table 6.1). The framed experiment involved measuring participation in planned *ashar* activities for each village once per week over the course of four weeks. Each activity involved the removal of fallen trees, shrubs and large rocks blocking water from flowing through collectively managed irrigation canals. The activity itself did not change throughout the four week period or between villages. The incentives were chosen by the research team to reflect the intersection of 'I', 'We' and 'other regarding' motivations as identified in section 6.2. The cultural acceptability of the incentive treatments were verified in a village meeting with the *Ayil okmotu*, school director, and the WUA organiser within each of the four villages. One of the following treatments was allotted to each of the four villages. The incentive treatments were as follows:

1) Payment is not mentioned nor offered (*control*).[24]

2) Payment is not offered, but it is clearly mentioned that no payments will be offered.

3) US$100 will be donated to the school library if the number of individuals contributing labour continues to increase from the first to the fourth week of the *ashar*.

[24] *Ashar* is defined within the selected communities as voluntary labour with no expectation of remuneration (Kuban Matraimov, *pers. comm*.)

4) Payment is offered as an individual award provided according to effort in the *ashar* work. The individual(s) participating the most and with the greatest effort will be publicly recognised by the *Ayil okmotu* at the end of four week period. In addition, a contribution to a local publication[25] would mention the individual(s) as 'community stewards'.

The choice of which village would be given a particular incentive treatment was randomly determined. The control treatment took place in the western-most village of Temirovka. The second treatment took place in Kashat; the third treatment in Kojoyar and the fourth treatment in Grigorievka which had the school in greatest need of support. The time and dates of each *ashar* activity were also decided upon during the initial village meeting and subsequently disseminated by word of mouth.[26] By linking research questions with already planned collective activities managed by local leaders rather than framed as research experiments, we avoid the experimental biases present in Kerr et al. (2012). While those attending the initial village meeting were fully aware of the research agenda, the information disseminated within each village by these individuals pertained only to the particular incentive treatment offered for that village. For instance, in the village of Kashat where incentive treatment 2 was given, it was announced throughout the village that a series of *ashar* activities would take place and that no payment would be offered. The two incentive-providing treatments (3 and 4) provided a degree of conditionality to the 'payment' in terms of a participation threshold (for treatment 3) and an effort-based component[27] (for treatment 4).

[25] *Ayil Demi*
[26] In the case of inclement weather, the *ashar* activity was postponed to the following day at the same time.
[27] The intensity of participation was also measured through the use of time logging the behaviour of randomly two selected participants every ten minutes for each *ashar* activity. The results are not presented here as these data do not reveal striking differences between treatments. Moreover, repeated contribution rather than intensity of participation was used to consider effort for treatment 4 since one individual attended and contributed to *ashar* more than any others across the four weeks.

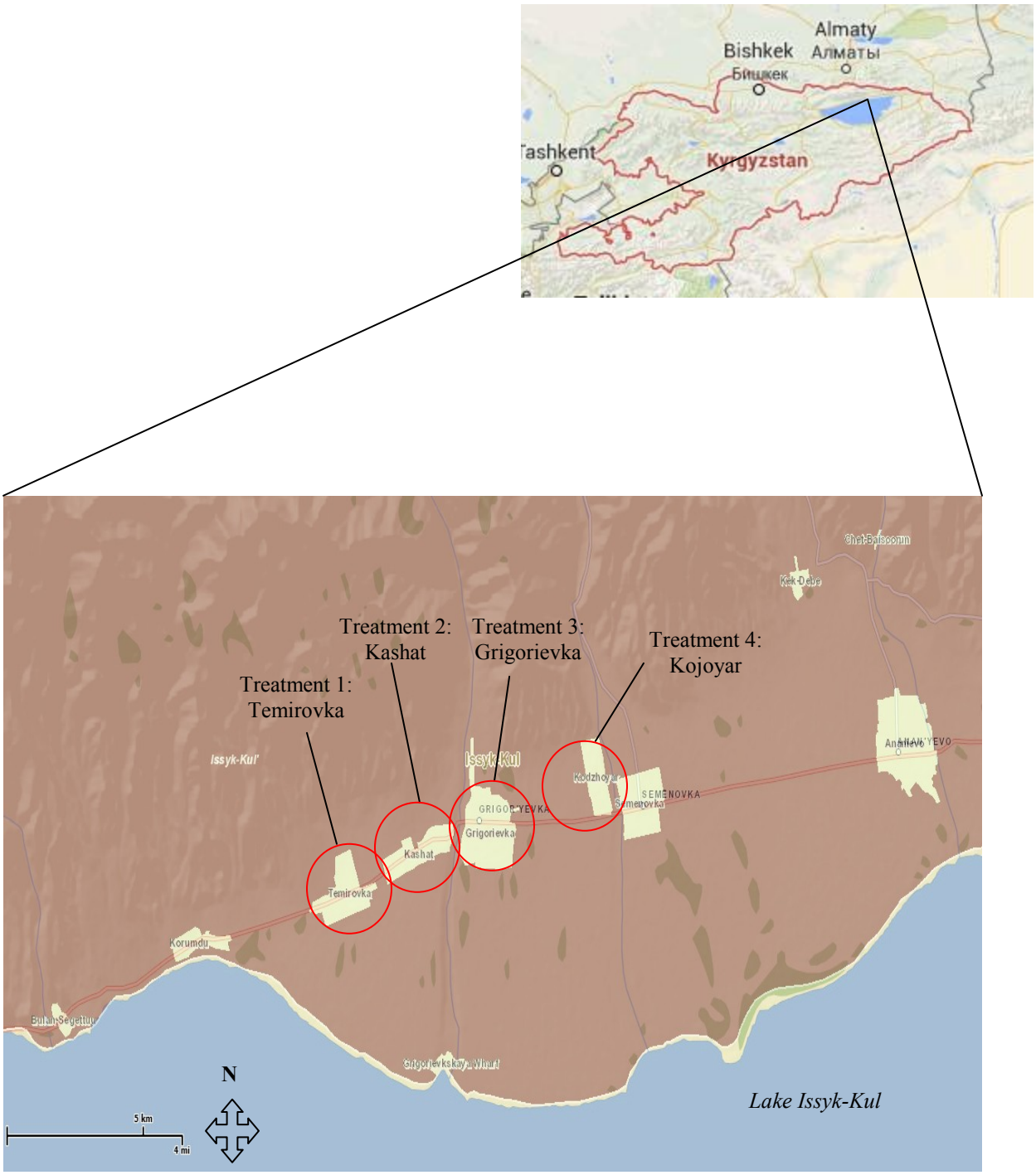

Figure 6.1: The location of villages and incentive and control treatments along the north shore of Lake Issyk-Kul, Kyrgyz Republic. (Images: Google Maps, 2014; USAID Reach Initiative, 2014)

Village	Treatment	Population (2014)[a]	Distance to nearest urban centre[b]	Main livelihoods	Frequency of annual *ashar* activities
Temirovka	Control	4568 (1123)	122km	Agriculture, husbandry	Autumn(1), Spring (2)
Kashat	No payment: expressed	3970 (975)	114km	Agriculture, husbandry	Autumn (1), Spring (2)
Grigorievka	Collective payment; group-payoff	4546 (1040)	106km	Agriculture, husbandry	Autumn (1), Spring (2)
Kojoyar	Individual; image-based	2576 (597)	99km	Agriculture, husbandry	Autumn (1), Spring (2)

Table 6.1: Village characteristics chosen for the framed experiments[b]
[a]Number in parentheses refers to the total adult male population per village, representing the sample of potential *ashar* participants
[b]Proximity to one of the Kyrgyz Republic's administratively independent cities or provincial capitals: Bishkek, Batken, Jalalabad, Naryn, Osh, Talas, or Karakol

Source: Village government (*Ayil okmotu*) census and village statistics (April, 2014) for Temirovka, Saderake, and Semenovka districts

While it was not possible to fully control for communication between the incentive treatments, it is believed that the distance between the villages minimised the risk that differences in the incentives offered for the *ashar* activities would be exposed[28]. Moreover, the research team solicited agreement from those privy to the research objectives (e.g. school directors, *Ayil okmotu* for each village and WUA mobilisers) to refrain from divulging the research objectives until the end of the experiment. Since village leaders were at the time of data collection working in close cooperation with CAREC, the presence of the latter at the initial meeting was instrumental in building trust between the research team and the *Ayil okmotu* of each village.

The location of the *ashar* activities was along an irrigation canal located within each village. At the end of the four week period, it was necessary to understand *why* participants contributed labour to the *ashar*. To better understand the source of motivation, semi-structured interviews

[28] One caveat of the research design is that it was not possible to conduct a census of all villagers to determine whether all had received and understood the message regarding the timing, location, and incentive provided for the *ashar* activities in each treatment. As mentioned in the initial planning meeting, many eligible villagers may have been busy with the apple/pear harvest which was occurring during the time of the planned *ashar* activities. Informal discussions with eligible individuals who were non-participants to the *ashar* corroborated that they were not aware of the *ashar* due to occupation with the harvest. While the overall low turnout in relation to eligible potential participants may be attributed to the concurrent fruit harvest, all villages were equally involved in the harvest. Given a comparable dissemination effort for each village, we feel that the reduced awareness of the planned *ashar* does not detract from the results of participation from the effects of each treatment.

were conducted with randomly selected participants[29]. In total 32 interviews with participants were conducted. Individuals interviewed were asked to express the primary motivating factor for participation, whether they would be inclined to participate if provided some form of payment, how they would feel if they knew *ashar* was taking place but were not contributing labour, and their perceptions towards the village *Ayil okmotu*, school director, and WUA organiser facilitating each *ashar* activity. Information was also collected on the number of *ashar* activities each individual typically contributes to on an annual basis as well as material assets owned as a proxy of household wealth. Finally, four focus groups were carried out (one per village). In addition to exploring the nature of *ashar* traditions for collective action in the village, questions from individual interviews were repeated in order to triangulate the responses in a social setting and to expose further insight underpinning participation.

6.4.4. Analysis

The participation in the *ashar* activities across the four villages and across four weeks was analysed according to: a) what villagers actually *do* when asked to participate in a collective action activity for the public benefit, with and without incentives; b) what villagers *say that they would do* in relation to participation in these activities with incentives and c) what additional insights social *deliberation* on the subject has to offer. The measurement of participation across and within incentive treatments as well as the probability of repeated contributions per treatment was determined through probability regression.

In order to assess the factors influencing behaviour, we carried out logistic regression following Kerr et al. (2012). Of particular interest were those individuals who claimed that their participation was not based on expected compensation or extrinsic reward. Hence the dependent variable was a binary score of '1' for individuals who claimed to participate for non-payment reasons. Independent variables were both continuous (e.g. age, number of annual *ashar* contributions, number of household assets owned); categorical (e.g. feelings of guilt for not

[29] The lowest participation turnout of the incentive treatments was in the village of Kashat (incentive treatment 2) with 11 individuals contributing across the four week period. Only 8 of these individuals were available for interview. To maintain equal representation across the treatments, 8 individuals were thus randomly selected in the other 3 treatment groups for a total of 32 interviewed participants.

attending; perceptions of the mobilisation capacity of the WUA organiser; perceptions of the *Ayil okmotu* and school director), and binary (e.g. repeated or one-off contributions over the four week period). Categorical data ranged from 1 to 5, with '1' being strongly agree and '5' being strongly disagree following a classic Likert scale, adopted for its simplicity in capturing preference in terms easily understandable by respondents.

Content analysis was conducted to examine the information collected from the four focus groups. Data was coded to reflect the themes identified in the literature relating to the interactions of incentive provision on pro-social behaviour: market versus social framing; decentralised norm enforcement, reciprocity; image motivation as well as extant social capital (e.g. relationships and informal institutions) present in each village. In this manner, qualitative analysis was used to reveal in-depth understanding of motivating factors for participating in the *ashar* that could not be revealed through logistic regression alone.

6.5. Results and Discussion

6.5.1. Contributions between the incentivised treatments

Table 6.2 presents descriptive statistics for the participants within each of four incentive treatments across the four villages. Mean and median age of the total number of participants within each incentive treatment is provided as well as a summary of interview responses. In three of the four villages, 90% or more of participants had contributed labour to *ashar* at least once over the past two years. In Kojoyar (incentive treatment 4), just over 50% had participated in *ashar* previously; the others contributed for the first time.

Additionally, it was determined that the age structure of participants differed between the four treatments with the village of Kojoyar (incentive treatment 4) having a significantly younger turnout. This result differs from the findings of Kerr et al. (2012) who found no difference in the age structure between incentive treatments in their framed experiments in Tanzania, yet resonates with their findings that incentives might increase the participation of individuals

previously disengaged in such work. Discussions emerging from the focus group revealed that most of the recent graduates of Kojoyar/Semenovka school had never before contributed to *ashar*. Moreover, they were attracted to the stewardship certificate as a possibility of receiving future leadership positions in the village.

> *"Yes, we came since we heard we would get a certificate from [the] Ayil okmotu.*
>
> *(probing question): "Why is this important?*
>
> *"We just finished school and are looking for jobs. We want to work hard to make a good name for ourselves."* (Focus group-treatment 4)

Table 6.3 provides a probit regression illustrating the probability of participating in each of the three treatments in comparison to the control (village 1). It can be seen that the probability of participating was significantly different between the different treatments; the probability of participating in treatment 2 is lower than the control ($p<0.01$) and higher in comparison to the control for treatments 3 and 4 ($p<0.001$). The probability of repeated contributions is shown in Table 6.4. The results highlight that there was a significantly lower probability of contributing labour repeatedly in the incentive treatments (treatments 3 and 4) in relation to the control. In relation to participation within each treatment across the four week period, significant differences in the number of participants across the weeks was evident for treatments 2 and 3, as shown in Table 6.5.

	1.Temirovka (*control*)	2. Kashat (*no incentive made explicit*)	3. Grigorievka (*collective payment; group payoff*)	4. Kojoyar (*individual image-based payment*)
Number of participants (total)[a]	21 (1.87%)	11 (1.13%)	77 (7.40%)	27 (4.52%)
Average household size[b]	5.8 (2.2)	4.8 (2.8)	5.9 (1.9)	4.1 (2.7)

Mean age	44.6 (13.0)	49.4 (13.4)	43.9 (12.4)	33 (13.5)
Median age	43	53	44	37
Age structure[c] 18-25 26-35 36-45 46-55 56-75	 4.8% 23.8% 33.3% 9.5% 28.6%	 0% 9.1% 27.3% 27.3% 36.4%	 9.1% 18.2% 27.3% 26.0% 19.5%	 40.7% 11.1% 25.9% 18.5% 3.7%
Awareness of payment	N/A	N/A	70 (91%)	22 (82%)
Repeated participation (attending 2 or more *ashar* activities)	10 (33%)	2 (18%)	14 (18%)	3 (11%)
Contributions to *ashar* at least once in the previous two years	20 / 21 (95%)	10 / 11 (91%)	75 / 77 (97%)	15 / 27 (52%)
Satisfaction with activity[d]	Very satisfied	Satisfied	Satisfied	Satisfied
Feel guilty for not contributing to *ashar* having known it was taking place[d]	Very Guilty	No strong feeling	Guilty	No strong feeling
Positive perception of WUA mobiliser as a respectable (leader)[d]	Very positive	No strong feeling	Very positive	No strong feeling
Experience of WUA mobiliser (number of years in this role)	6	5	30	2
Overall perception of current *Ayil okmotu*[d]	Positive	No strong feeling	Positive	No strong feeling
Overall perception of current school director[d]	No strong feeling	No strong feeling	No strong feeling	Positive

Table 6.2: Characteristics of participants in the framed incentive treatments

[a]Number in parenthesis represents percentage of total adult male population per village as of April, 2014.

[b]Number of immediate and extended family members living in a single household

[c]Statistically significant difference in age structure of participants across the treatments (Chi sq. = 28.22, 12df, $p<0.01$). This finding is not an artefact of village population structure as village census data confirmed no significant difference in the age structure of the population between the four villages as of April, 2014.

[d]Presented as the average of a Likert classification: 1= very satisfied, very guilty, or very positively perceived; 3= no strong feeling; 5 = highly unsatisfied, strong lack of guilt, or highly negatively perceived.

The differences in participation between the treatments are displayed graphically in Figure 6.2. The number of participants is compared across the four villages based on the most recent census data of adult males between the ages of 18 and 75 in each village. Incentive treatment 3 (group payoff) attracted the greatest percentage of possible participants. Conversely, it was not possible to attract further contributors after the second week in treatment 2 where no payment was

explicitly stated. It can also be seen that the individual incentives (treatment 4) attracted participants to a greater extent than did the control (treatment 1). While the number of participants remained stable across the weeks in both the control and treatment 4, the number of participants increased each week in treatment 3 as per the conditionality requirement of the group pay-off. These findings resonate with the findings of Yanez-Pagans, 2013; Kerr et al., 2012, and Narloch et al., 2012 suggesting that incentives may increase cooperation in collective resource management.

	Model
Treatment	
2	-0.199363**
	(0.7223)
3	0.635015***
	(1.7650)
4	0.3883557***
	(0.0628)
# of Cases (states)	14940
R^2	0.0598

Table 6.3: A probit regression showing the probability of participation in *ashar* between the different treatments, in which each treatment is compared against the control (village 1).
* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

	Model
Treatment	
2	-0.728446
	(0.5193)
3	-0.634093*
	(0.3190)
4	-1.040628*
	(0.4215)
# of Cases (states)	136
R^2	0.0479

Table 6.4: A probit regression depicting the probability of repeated contributions to *ashar* between the different treatments, in which each treatment is compared in contrast to the control (village 1).

* p < 0.05; ** p < 0.01; *** p < 0.001

	Model
Treatment	
1	0.0046407
	(0.0575)
2	-0.4534769**
	(0.1494)
3	0.0831906*
	(0.0389)
4	0.0148956
	(0.0617)

# of cases (states)		R^2
Treatment 1:	4492	0.000
Treatment 2:	3900	0.097
Treatment 3:	4160	0.005
Treatment 4:	2388	0.000

Table 6.5: A probit regression showing the probability of participation in ashar within a single treatment across the four week time period.
*p<0.05; **p<0.01; ***p<0.001

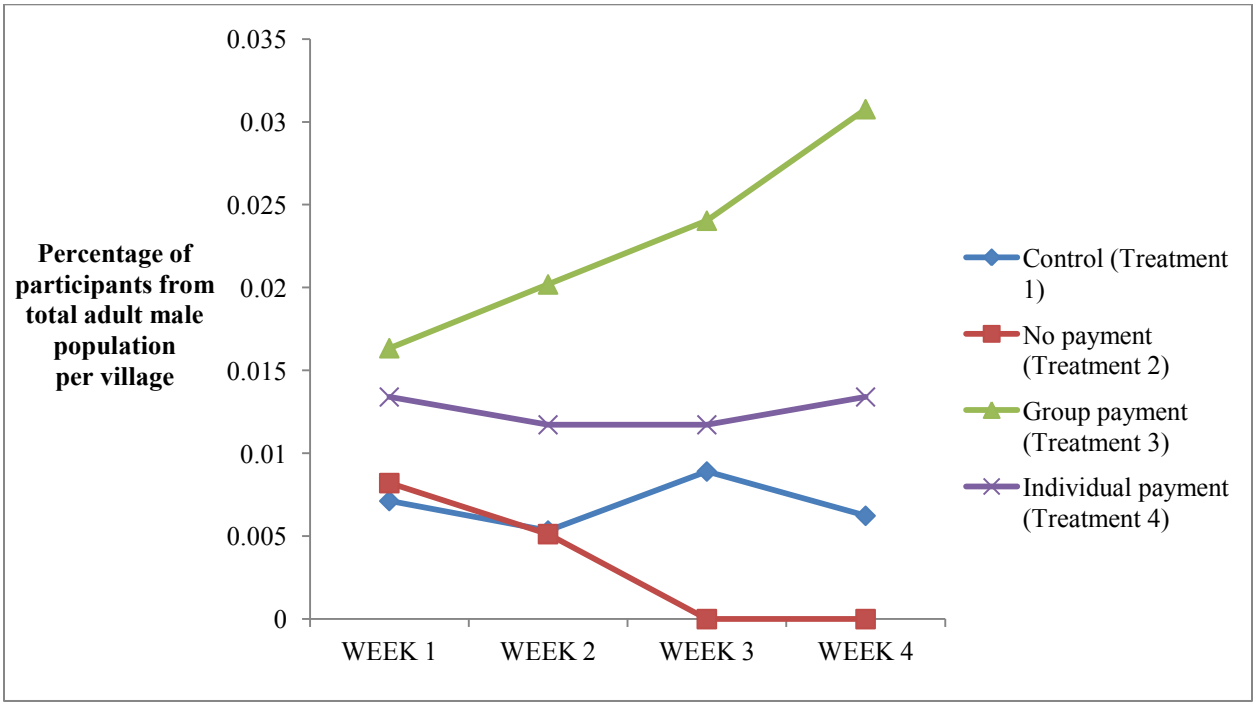

Figure 6.2: Participation in the collective *ashar* activities per village treatment across four weeks.

Despite a lack of significant difference in the number of participants contributing to *ashar* across time within a single incentive treatment, several interesting relationships were evident between those participants that returned for two or more *ashar* activities based on data collected in the post-activity interviews. As Table 6.6 (C) illustrates, those who contributed repeatedly (across any of the treatments) were significantly more satisfied with their participation. Thus, participants in treatment 1 (control) were more likely to be satisfied with their contribution than in the other treatments, since there was the greatest percentage of returning individuals as indicated in Table 6.2. Those who were least interested in receiving compensation for their participation were also more likely to be satisfied with their experience (A). Moreover, those who were most satisfied with their contribution were also those who had more faith in the WUA organiser as a strong motivator (B); likewise, those who believed the WUA organiser to be a strong motivator were also more likely to contribute repeatedly (D). There was no significance between preference for payment and likelihood for repeated contributions. This suggests that expectation of payment plays less of a role in attracting people who contribute labour repeatedly. The relatively poor perception of the leader organising and directing the activities in treatment 4 as shown in Table 6.2, combined with the lowest repeated contribution (despite the conditionality basis of the incentive towards consistent effort across the four weeks) offers evidence to this claim.

A)	Satisfied with contribution	No strong feeling	Unsatisfied with contribution
1. Compensation strongly discouraged	4	0	0
2. Compensation discouraged	9	1	1
3. Indifferent to compensation	3	3	0
4. Compensation preferred	4	3	2
5. Compensation strongly preferred	0	0	2
TOTAL	20	7	5
CHI-SQ: 19.4967, df=8, $p<0.05$*			
B)	Satisfied with contribution	No strong feeling	Unsatisfied with contribution
Strong motivator	15	1	0

Adequate motivator	1	3	2
No strong feeling	3	2	2
Weak motivator	1	1	1
TOTAL	20	7	5

C)	Repeated contribution	One-shot
Satisfied with participation	14	6
No strong feeling	0	7
Unsatisfied with participation	0	5
TOTAL	15	17
CHI-SQ: 14.6932, df=6, p<0.05*		

D)	Repeated contribution	One-shot
Strong motivator	13	3
Adequate motivator	0	6
No strong feeling	1	6
Weak motivator	0	3
TOTAL	14	18
CHI-SQ: 18.6122, df=3, p<0.01**		

Table 6.6: Results of post-activity interviews revealing satisfaction with *ashar* contributions; repeated contributions over the four-week period; perceptions of the *ashar* mobiliser; and preference for compensation for contributions to *ashar*.
Source: Post-activity interviews, N = 32.
Both preference for incentive and repeated contributions as well as preference for incentive and perception of the *ashar* mobiliser yielded no significant difference between actual and expected distributions.

While we assume that rational self-interest would explain why people would prefer to be compensated for their efforts, it is relevant here to understand what factors may influence why people may prefer not to be compensated. Table 6.7 presents an ordinal logistic regression explaining preference for uncompensated contributed labour and links these directly to the independent variables directly associated with *ashar* activities more generally and described in section 6.4.3.

	Model
Variables	
Repeated contribution	2.728475* (1.1235)
Ayil okmotu	1.992217** (0.6512)
Guilt	1.180326* (0.4866)
School director	3.337935** (1.1263)
Ashar Leader	1.376458* (0.5951)
Age Class	0.3272249 (0.5636)
Assets	0.8228769 (0.4513)
Ashar participation	0.5633838 (0.4513)
Household #	0.4389852 (0.2867)
# of Cases (states)	32
R^2	0.4032

Table 6.7: Ordinal logistic regression explaining contribution to *ashar* without preference for compensation. The dependent variable is stated preference for no compensation for participation in *ashar.*
* p < 0.05; ** p < 0.01; *** p < 0.001

Of these, perceptions of the village government (p<0.01), satisfaction with participation (p<0.01), repeated contribution (p<0.01); leadership perception of the WUA organiser (p<0.05), perception of the school director (p<0.05), and admission of guilt in failing to attend *ashar* (p<0.01) significantly explained strong opposition to compensation.

It is necessary to emphasize the limitations of these findings given the inherent variability in social capital between treatments in the experimental setting. Underlying variability in relationships between actors would naturally explain difference in contributions to collective action between treatments. The identification of a 'control' treatment reflects population, socio-economic and cultural characteristics which can be controlled for in a real-life setting, but not the perceptions of actors which is characterised by a dynamic interplay of both historical and

ongoing interactions between individuals. Accordingly, closer engagement with participants is needed which considers each treatment individually in order to better understand processes of extant social capital unique to each village. Moreover, the relatively small number of participants limits the extent to which robust conclusions can be drawn, thus highlighting the value of understanding these phenomena through a more exploratory approach. Greater contextualisation through deliberation of unobserved social processes also serves to expose more tacit understandings of the circumstances in which the provision of incentives under existing social norms can be positively synergistic.

6.5.2. Lessons from the different incentive treatments

Focus groups with participants from each incentive treatment revealed more nuanced insights in appreciating the rationale behind the decision to contribute efforts for the *ashar*.[30] In the case of the control (treatment 1), participants enjoyed contributing labour not only because of a feeling of responsibility for maintaining what is collectively 'theirs', but also as an opportunity to socialise with their neighbours as well as respect for the WUA organiser who called them out to participate. The latter aspect was raised both in the discussion and observed in the field. For example, after each *ashar* session was completed, all participants together with the WUA organiser spent between 30 minutes to one hour sharing refreshments and socialising. This 'social session' following each *ashar* was unique to the control treatment, and did not take place in the other treatments. When asked about payment, the response defended the tradition of *ashar* as a traditional duty. Concerns were raised that the introduction of payment could disrupt this duty.

"We all benefit from the canals and so have to put the efforts to maintain them."

[30] Individuals in the focus groups included only *ashar* participants. WUA organisers, representatives of the *Ayil okmotu*, the school or other village leaders were not invited to these discussions to facilitate unrestrained and unbiased opinion.

> *"If you start to pay people to do ashar, they will make excuses when they are called to work. They won't feel like it is their duty to maintain [the canals]. Instead, it will become a salary for them."* (Focus group-treatment 1)

These quotes suggest that benefits to cooperation are relevant social goals. There is also an allusion to the existing of dual norms: a civic-minded frame and one centred on remuneration for duly completed work. These expressions reflect the concerns raised in the literature in which decentralised norm enforcement is destabilised by the provision of incentives whereby free-riders feel justified for not contributing effort when those that do contribute receive an incentive that free-riders would not (Ostrom, 1990; Fehr and Falk, 2002; Carpenter et al., 2004; Fuster and Meier, 2010). The following quotes illustrate the salience of reciprocity and norm enforcement which may lead to social exclusion if contributions to *ashar* are not made.

> *"If we do not contribute, how can we take extract water to irrigate our fields when our neighbours who did contribute see that we are taking water without having contributed anything?"* (Focus group-treatment 3)

> *"*(Name of WUA organiser)*...will not have a good opinion of us and then we cannot obtain water for our crops when we propose making an extraction [of water] for our fields."* (Focus group-treatment 1)

The self-regulation process of enforcement associated with the social stigmatisation of free-riders is evident in these sentiments (Fuster and Meier, 2010). The second quotation makes evident the reputational effect of the farmers in relation to the WUA organiser and their eventual opportunity to extract water for household benefit.

Where the control was replicated with the explicit mention that remuneration for contributions would not be provided, responses for contribution (or the lack of contribution) were markedly different. In the focus group for treatment 2, participants noted that they were told that payment would not be provided for their participation. Participants who did attend expressed in the focus group that in recent years, foreign aid agencies have promoted community development by

compensating people for their time. It was argued that by explicitly mentioning no payment, some farmers may have confused the activity with a project organised by one of these agencies since association with payment is otherwise not affiliated with *ashar* work:

> *"It is strange that* [name of WUA organiser] *told us that we would not be paid. I think some may have thought it was an employment opportunity offered by Peace Corps or JICA* [Japan International Cooperation Agency]. *They have come here in the past and offered payment for improvements in the village."* (Focus group-treatment 2)

This understanding clearly echoes the results of Heyman and Ariely (2004) indicating that the 'framing' of an activity may be shaped according to either social or market norms. The possible perceived affiliation of foreign development agencies with employment also suggests a foreboding implication for PES, in which uncompensated labour according to community norms become replaced with expectations of payment associated with externally-driven projects. However, we do not have enough evidence to substantiate this claim.

For those who did contribute, the social norm of *ashar* was defended:

> *"One time we did ashar work and one neighbour didn't come, then at that point all the neighbours approached him and told him he cannot get water because he didn't show up for the ashar work. It does not look good if we do not attend."* (Focus group-treatment 2)

Treatment 3 (Grigorievka) resulted in the highest turnout of any treatment, yet revealed somewhat mixed responses in relation to the group pay-off. There was a consistent response in both individual interviews and the focus group discussion that support and respect for the WUA organiser was the rationale for their participation. In relation to the collective payment, participants in the focus group agreed that they would participate in *ashar* even without compensation as they have done previously.

This viewpoint was endorsed by a participant who expressed distrust in the school director in distributing the funding for its destined cause. While respect for the WUA organiser may have

influenced participation, the communal payment itself is less likely to have had an effect in a setting of institutional corruption or mistrust in those responsible for managing the payment (Kerr et al., 2012). However, for some individuals, a clear positive synergy was set in motion as a result of the community payment, particularly for individuals with children attending the school as one participant stated.

> *"...and now that there is payment to school for ashar, I know I must participate since my daughter goes to the school and I also I get water from the canal"* (Focus group-treatment 3)

At the same time, the communal payment was also seen as unfair for those individuals who do not have children attending the school as another participant argued:

> *"Providing money to the school is good but will cause conflict for those people who don't have children in the school. This may be seen as unfair if some cannot benefit from the payment in some way also."* (Focus group-treatment 3)

Despite these divisions of opinion related to the introduction of a communal payment, agreement was unanimous amongst participants that contributions to *ashar* are required when the organiser calls on the community to participate. The personal agenda of a well-respected individual, in this case to mobilise the necessary commitment to receive the communal payment, has a powerful influence in directing outcomes. Thus strong social norms can leverage the use of group payoff incentives to achieve greater collective benefits.

Incentive treatment 4 (Kojoyar) presented the converse of this potentially synergistic effect of incentives. It also brought to light the differential influence of incentives between older and younger generations to contribute to *ashar*. It was argued in the focus group of treatment 1 (control) that younger people require compensation since they do not fully understand the value of collective resources and accordingly do not feel affected by not maintaining the canals. This divergence between the perceptions of the young and the older generation was emphasised still further in the focus group for incentive treatment 4, where one participant explained that the

'community steward' certificate influenced his decision to contribute effort. He argued that for himself, the opportunity cost of time cannot justify providing labour without compensation:

> *"If I had a job, I wouldn't feel guilty for not attending ashar since I wouldn't have time. Many of us are busy in trying to make ends meet and cannot be everywhere at once."* (Focus group-treatment 4)

However, an older participant warned that awareness-raising amongst the younger generation regarding *ashar* traditions is crucial to avoid the degeneration of the tradition as a result of incentivisation:

> "*If you give money, people might not want to do ashar work without money. This can be dangerous, it will mean there will not be anymore ashar works, because there will never be enough money to fund everyone for their demands for the work. This will not happen immediately, but after a few times the ashar tradition will die completely."* (Focus group-treatment 4)

Ironically, what has attracted younger participants is the recognition of being seen to be 'doing good', resonating with the signaling of image motivation geared to self-interest but manifested through collective benefit. As such, image enhancement is not a perfect signal of utility maximization as monetary incentives would be. Moreover, it is noteworthy to highlight that image motivation will only be successful in encouraging certain behaviours insofar as respect and trust in the perceptions and preferences of a social group or specific individuals exists to reciprocate the 'good image' effect (Fehr and Falk, 2002).

As mentioned, despite being a key condition to receiving recognition as a community steward, the percentage of repeated contributions was the lowest in treatment 4. One reason for the poor adherence to the condition of the incentive is perhaps related to the relatively poor perception of the WUA organiser in this village.

"(participant 1): (He) doesn't seem to know how to manage the canals consistently. We've had a lot of problems and think it is time for someone else to take charge.

"(participate 4): yes, I have not been able to take water some days due to obstructions in the canal. When I complained to him about this, there was no action taken."

Thus, the provision of an incentive alone, even if conditional to a particular collectively desired objective, may not be sufficient in this case in the absence of strong leadership. Indeed, it was the incentive of community recognition which encouraged younger participants, yet it was unable to encourage their commitment to actually attain the award. Indeed, of those who attended more than one of the activities across the four weeks, all were among older participants and had stated that their participation was based on being called to participate by the WUA organiser. Moreover, none of these repeated contributors stated in post-hoc interviews that they would be more inclined to contribute to *ashar* if they received compensation. Hence, what these individuals have said and what they do in practice appears to be consistent.

6.5.3. A plurality of rationalities

The lesson from these experiments serves to illustrate that each context represents a specific configuration of institutional history as well as dynamic patterns of reciprocity and trust characteristic of social capital, and will thus respond to incentives in different ways. Each context is held together by a particular interplay of 'I', 'We', and other regarding sentiments which sustain, enhance or deter collective action potential. In relation to participation in the absence of incentives, decentralised norm enforcement, and historical interactions based on positive reciprocity have been illustrated to be clear motivators for contribution. These factors are internal drivers of behaviour as they refer to established patterns of interaction influencing collective resources. Moreover, they are related to a fiercely hierarchical system of leadership present in these and many other rural communities in the Kyrgyz Republic and other parts of Central Asia.

The plurality of rationalities guiding behaviour coalesces in interesting ways. Individuals attend *ashar* for the purely self-interested reason that if they do not contribute, they will not have the social 'permission' to remove water for their crops. However, they also attend based on a historically embedded tradition of cooperation for collective benefit as well as to socialise with their neighbours. Additionally, respect and reputation associated with relationships with village-appointed leaders reflects an 'other-regarding' motivation that straddles both 'I' and 'We' rationalities. The interaction of these rationalities is uniquely crafted yet constantly shifting as a result of relationships between individuals, the reputation and respect of leaders, and external circumstances. The ephemeral configuration of these rationalities shifts state when new signals are introduced which stimulate the 'I' and the 'We' rationalities in subtle ways. For example, village-level payments such as donations to the school library fund add an additional 'I' dimension, particularly for those whose students attend the school. At the same time, group-payoff is contingent upon a certain level of participation by the end of the four week period which re-emphasizes the 'We' in cooperating for collective benefit. Leveraging upon the 'we' relationships may have positive implications for building conditionality into PES agreements. Thus, the shifts induced by incentives can be positively synergistic (incentive treatment 3), antagonistic (incentive treatment 2), or result in unexpected outcomes that harness multiple rationalities (incentive treatment 4).

The results of this study identify clear differences in the proclivity to participate between the incentive treatments, but suggest that incentives alone did not provide sufficient evidence to explain repeated contributions of labour for collective action. While it may be argued that the incentives provided were not large enough to result in repeated contributions (e.g. a critical component of the conditionality of the incentives), the more important implication is that 'other regarding' motivations can trump an imposition of the 'I' narrative in certain contexts. The significant chi-squared relationship between trust and respect in leaders and incidence of repeated contributions stands testament to this point. Further, in the case of treatment 3, conditionality requirements were met due to a combination of incentive and 'other regarding' factors such as reputational norms associated with trust in leaders which were strongly emphasized in the focus group discussions. In this case, conditionality may be more closely tied to a "We" versus an 'I' rationality, yet could possibly change given greater emphasis on the 'I'

rationality. Future research might explore the effect of incremental incentives on repeated contributions to examine the extent to which existing 'We' regarding sentiments are affected through increasing replacement by an 'I' framing. Depending on the emphasis of latent 'I' and 'We' framings in a particular community or context, loyalty to incentive conditions may vary. Such insight would have important implications for determining the behavioural motivations which promote for conditionality in PES schemes in a given setting.

Hence, a 'muddling' of both 'I' and 'We' narratives becomes apparent. In the case where the original "We" narrative of *ashar* shifts to an 'I' logic in framing collective action, contribution may be encouraged, such as treatment 4 which brought in additional volunteers interested in obtaining the certificate. However, without the presence of the countering 'We' narrative recognised through strongly positive reciprocity and traditional modes of socio-political organisation, there may be a continued dependence on incentives to attain a desired level of contribution. Alternatively, the lack of a 'We' narrative may fail to generate the requisite degree of commitment for a task which is inherently in the collective interest. This is much in line with comments invoked in the focus groups warning of the unravelling of *ashar* traditions should participation be dependent on incentives. It seems clear then that any introduction of incentives would need to be tailored carefully to appeal to different rationalities of 'I' and 'We' at the same time.

6.5.4. Role of leaders and the interplay of rationalities: implications for PES

The strong influence of local leaders in mobilising people in this context raises several thorny concerns. As mentioned, the nomadic tradition of collective support was reinforced by the Leninist *subbotniks* in which a powerful leader calls on people to contribute labour, in a highly top-down manner (Earle et al., 2004). Historical patterns of leadership have traditionally permitted informal institutions to function for collective benefit, since it is based on repeated positive interactions over time. When an intervention is premised on an 'I' rationality, leaders historically respected in the community can become victims of local power struggles. It is thus pertinent to identify how PES can inadvertently predate upon local leaders to induce a shift in the interplay of rationalities guiding collective action.

There exists a mismatch between dynamic relationships of trust and reciprocity at the local scale and the introduction of payments which harness the influence of powerful individuals to advance the 'I' agenda. An example can be drawn from the PES pilot. The intermediary organisation (CAREC) identified a specific local leader, who was previously the *Ayil okmotu* of the village of Temirovka, to become the 'champion' or spokesperson of the PES initiative. Bremer et al. (2014) identify such charismatic individuals as 'change agents' instrumental for legitimizing PES interventions amongst relevant social actors. Following this approach, this chosen leader was paid by CAREC to mobilise service beneficiaries to provide payment in the form of labour for upstream land-use management. Additionally, CAREC received funding to send this individual to observe successful implementation of PES in Vietnam in order to transfer such success to the Kyrgyz Republic where PES had not yet been experimented with. Upon discussing perceptions of the *Ayil okmotu* and other local leaders in the treatment 1 focus group, a comment was raised with respect to the PES 'champion' as a respected leader in the community. Others acknowledged and expressed agreement with the sentiment raised:

> *"I do not know why [individual] is being paid for this position. [Individual] is known for having stolen money that was meant to be used to repair a bridge in Temirovka. They should have chosen someone else. Now they are sending [individual] to travel to different places and paying him, but it is not right"*

Lastly, it is relevant to note that *ashar* traditions involve little input from community members and hence offer minimal opportunity for initiating deliberative processes for the expression of plural values linked to 'I' and 'We' rationalities. Influence in organising *ashar* lies entirely with the orders of a village leader or WUA organiser; though, as has been observed in the aforementioned example, social norms of reciprocity play a self-reflexive role in tempering the power of such leaders to act against the collective interest.

Understanding how the behaviour of village leaders responds to social norms and expectations of villagers is the delicate 'I'-'We' balance which PES interventions will need to be attuned with (Vatn, 2009). The implication here is that the introduction of incentives for managing ecosystem

services is a delicate process of co-creation between individual needs and wants as well as social norms which maintain processes of collective action. In the wake of calls to 'scale up' PES across the Kyrgyz Republic, we warn that formulaic recipes of motivating behaviour through incentives will at best underestimate social dynamics and at worst reinforce social inequities or perceived injustices.

6.6. Conclusion

This research adds to a body of literature linking experiments of contributions to a common good, in this case to the maintenance of collectively owned irrigation canals, with useful insights for the design and implementation of PES. Where ecosystem service contracts between discrete beneficiaries and service providers (with no risk of free-riders) under the assumption that specific land-use behaviour will always lead to unequivocal and quantifiable outcomes become costly or challenging to specify, then collective action arrangements may be more appropriate for promoting ecosystem services, particularly in the case of biodiversity protection and water quality improvements. Thus, there is value in bridging the literature on PES with that of collective resource management, particularly in understanding the design and success of incentive strategies.

In the case study presented, the 'I'-'We-other regarding' rationalities are reflected through the household dependence on water for irrigation and the collective ownership of irrigation infrastructure. The interplay between these rationalities provides clear implications for PES implementation. In particular, it raises a question on the appropriate role of intermediaries for PES implementation given the significance of inherently social norms and informal institutions in crafting incentive-based agreements for collective resource management. In this regard, it is useful to distinguish between intermediaries who facilitate the bringing together of actors to implement a pre-determined or pre-funded PES project, and intermediaries who mobilise actors by leveraging upon social norms, informal institutions and local customs in order to derive a set of rules which may result in a set of positive or negative incentives. Intermediaries as 'facilitators' of externally determined rules are not politically neutral, as the above example of the PES champion illustrate; yet the established trust and reciprocity intermediaries can offer as embedded in agreed upon rules and norms can serve to mobilise others in desired ways.

Additional research may investigate how deliberative mechanisms of communication can serve to identify particular sets of incentives that reflect both 'I' and 'We' objectives. Future research might also explore the impact of introducing incentives and later removing them. This could take place under two scenarios; one in which incentives are externally introduced as in-kind or individually targeted and another where incentives align with pre-existing social norms influencing natural resource management in specific ways.

Finally, this study has recognised the limitations of quantitatively assessing how collective action behaviour responds to incentives under a context of dynamic social relationships. We should not assume that it is the goal of the researcher to uncover discrete relationships between incentive provision and behaviour. Overemphasis on the significance of this relationship leads us back to the trap of relying on universal theories of human behaviour. Given the unique historical context and socio-political transitioning occurring in the Kyrgyz Republic and elsewhere in Central Asia, the scaling up of PES interventions requires closer scrutiny to this end.

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6.8. References

Agresti, A., 2002. *Categorical Data Analysis*. John Wiley, Hoboken.

Akçay, E., Van Cleve, J., Feldman, M.W., and Roughgarden, J., 2009. A theory for the evolution of other-regard integrating proximate and ultimate perspectives. Proceedings of the National Academy of Sciences of the United States of America 106 (45), 19061-19066.

Ariely, D., Bracha, A., and Meier, S., 2009. Doing good or doing well? Image motivation and monetary incentives in behaving prosocially. The American Economic Review 99 (1), 544-555.

Bénabou, R., and Tirole, J., 2006. Incentives and Prosocial Behaviour. American Economic Review 96 (5), 1652-1678.

Bowles, S., 2008. Policies designed for self-interested citizens may undermine "the moral sentiments": Evidence from economic experiments. Science 320 (5883), 1605-1609.

Bremer, L.L., Farley, K.A., and Lopez-Carr, D., 2014. What factors influence participation in payment for ecosystem services programs? An evaluation of Ecuador's SocioPáramo program. Land Use Policy 36, 122-133.

Cardenas, J. C., 2000. How do groups solve local commons dilemmas? Lessons from experimental economics in the field. Environment, Development and Sustainability 2 (3-4), 305-322.

Cardenas, J.C., Rodriguez, L.A., and Johnson, N., 2011. Collective action for watershed management: Field experiments in Colombia and Kenya. Environment and Development Economics 16, 275-303.

Carpenter, J.P., Daniere, A.G., and Takahashi, L.M., 2004. Cooperation, trust, and social capital in Southeast Asian urban slums. Journal of Economic Behaviour and Organization 55, 533-551.

Clements, T., John, A., Nielsen, K., An, D., Tan, S., and Milner-Gulland, E. J. 2010. Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecological Economics 69 (6), 1283-1291.

Corbera, E., Kosoy, N., and Martinez Tuna, M. 2007. Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. Global Environmental Change 17 (3), 365-380.

Earle, L., Fozilhujaev, B., Tashbaeva, C., Djamankulova, K., 2004. Community Development in Kazakhstan, Kyrgyzstan, and Uzbekistan. Lessons Learnt from Recent Experience. INTRAC. Occasional Paper Series Number 40.

Engel, S., Pagiola, S., and Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65 (4), 663-674.

Escobar, M.M., Hollaender, R., Weffer, C.P., 2013. Institutional durability of payments for watershed ecosystem services: Lessons from two case studies from Colombia and Germany. Ecosystem Services 6, 46-53.

Etzioni, A., 1989. Toward an I & We Paradigm. Contemporary Sociology 18 (2), 171-176.

Fehr, E., and Falk, A. 2002. Psychological foundations of incentives. European Economic Review 46 (4), 687-724.

Fiske, A. P., 1992. The four elementary forms of sociality: framework for a unified theory of social relations. Psychological Review 99 (4), 689.

Fuhrmann, M., 2006. A tale of two social capitals: Revolutionary collective action in Kyrgyzstan. Problems of Post-Communism 53 (6), 16-29.

Fuster, A., and Meier, S., 2010. Another Hidden Cost of Incentives: The Detrimental Effect on Norm Enforcement. Management Science 56 (1), 57-70.

Garmendia, E., Pascual, U., 2013. Chapter 8: A justice critique of environmental valuation for ecosystem governance. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Gintis, H., Henrich, J., Bowles, S., Boyd, R., Fehr, E., 2008. Strong Reciprocity and the Roots of Human Morality. Social Justice Research 21 (2), 241-253.

Heyman, J., and Ariely, D., 2004. Effort for Payment: A Tale of Two Markets. Psychological Science 15 (11), 787-793.

Hopfensitz, A., and Reuben, E., 2009. The importance of emotions for the effectiveness of social punishment. The Economic Journal 119 (540), 1534-1559.

Henrich, J., Boyd, R., Bowles, S., Camerer, C., Fehr, E., Gintis, H., ... and Tracer, D., 2005. "Economic man" in cross-cultural perspective: Behavioral experiments in 15 small-scale societies. Behavioral and Brain Sciences 28 (6), 795-815.

Kerr, J., Vardhan, M., and Jindal, R., 2012. Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. Ecological Economics 73, 220-227.

Kosoy, N., and Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69 (6), 1228-1236.

Masclet, D., Noussair, C., Tucker, S., Villeval, M.C., 2003. Monetary and non-monetary punishment in the voluntary contributions mechanism. American Economic Review 93, 366-380.

Matulis, B.S., 2013. The narrowing gap between vision and execution: Neoliberalization of PES in Costa Rica. Geoforum 44, 253-260.

McAfee, K., 2012. The Contradictory Logic of Global Ecosystem Services Markets. Development and Change 43 (1), 105-131.

McDermott, M., Mahanty, S., and Schreckenberg, K., 2013. Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. Environmental Science & Policy 33, 416-427.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202-1208.

Narloch, U., Pascual, U., Drucker, A.G., 2012. Collective Action Dynamics under External Rewards: Experimental Insights from Andean Farming Communities. World Development 40, 2096-2107.

Nyborg, K., 2000. Homo economicus and homo politicus: interpretation and aggregation of environmental values. Journal of Economic Behavior & Organization 42 (3), 305-322.

Ostrom, E., 1990. *Governing the commons: The Evolution of institutions for collective action*. Cambridge University Press, Cambridge.

Pagiola, S., and Platais, G., 2007. Payments for Environmental Services: From Theory to Practice. World Bank, Washington.

Pfaff, A., Robalino, J.A., Sanchez-Azofeifa, G.A., 2006. Payments for Environmental Services: Empirical analysis for Costa Rica. Columbia University, New York.

Pirard, R., and Billé, R., 2010. Payments for environmental services (PES): a reality check (stories from Indonesia). Number 3. Institut du développement durable et des relations internationales (IDDRI). Paris, France.

Rico García-Amado, L., Ruiz Pérez, M., and Barrasa García, S., 2013. Motivation for conservation: Assessing integrated conservation and development projects and payments for environmental services in La Sepultura Biosphere Reserve, Chiapas, Mexico. Ecological Economics 89, 92-100.

Rodriguez-Sickert, C., Guzmán, R. A., and Cárdenas, J. C., 2008. Institutions influence preferences: Evidence from a common pool resource experiment. Journal of Economic Behavior & Organization 67(1), 215-227.

Salzman, J., 2005. Creating markets for ecosystem services: notes from the field. New York University Law Review 80 (6), 870-961.

Singh, S., 2013. Chapter 7. Local Governance and Environment Investments in Hiware Bazar, India. *In*: Healy, H., Martínez-Alier, J., Temper, L., Walter, M., Gerber, J-F. (Eds.). *Ecological Economics from the Ground Up*. Routledge, London.

Sommerville, M., Milner-Gulland, E. J., Rahajaharison, M., and Jones, J. P., 2010. Impact of a Community-Based Payment for Environmental Services Intervention on Forest Use in Menabe, Madagascar. Conservation Biology 24 (6), 1488-1498.

Tomasello, M., Carpenter, M., Call, J., Behne, T., and Moll, H., 2005. Understanding and sharing intentions: The origins of cultural cognition. Behavioral and Brain Sciences 28 (5), 675-691.

Turpie, J.K., Marais, C., and Blignaut, J.N., 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics 65 (4), 788-798.

Van Hecken, G., Bastiaensen, J., Vásquez, W.F., 2012. The viability of local payments for watershed services. Empirical evidence from Matiguás, Nicaragua. Ecological Economics 74, 169-176.

Vatn, A., 2005. Rationality, institutions and environmental policy. Ecological Economics 55 (2), 203-217.

Vatn, A., 2009. An institutional analysis of methods for environmental appraisal. Ecological Economics 68, 2207-2215.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69 (6), 1245-1252.

Vollan, B., 2008. Socio-ecological explanations for crowding-out effects from economic field experiments in southern Africa. Ecological Economics 67 (4), 560-573.

Wunder, S., 2008. Payments for environmental services and the poor: concepts and preliminary evidence. Environment and Development Economics 13 (3), 279-297.

Yanez-Pagans, P., 2013. Cash for cooperation? Payments for Ecosystem Services and common property management in Mexico. CIDE International Seminar, October 2013. Mexico City, Mexico.

Chapter 7

GENERAL SUMMARY AND CONCLUSIONS

This research has advanced the scholarship on PES by exposing the myths of applying the policy tool for water resource management from an economic efficiency perspective. PES has been heralded as a means to promote more flexible and socially acceptable conservation over legal restrictions, punitive taxation or integrated community-based initiatives since it arguably links direct and conditional incentives to voluntary land-use stewardship (Wunder, 2005; Pattanayak et al., 2010). To date, developed countries have invested over $170 billion in support of PES approaches[31] internationally, signifying that directing financial flows to increase natural resource management and ecosystem service delivery is becoming an increasingly important development priority (Kerr et al., 2014). While PES initiatives that reflect the theoretical definition do exist (Wunder, 2008), the vast majority of mechanisms are PES-like, having evolved out of a context of political relationships between actors, perceptions of equity and justice, and substantial information asymmetries and ecological uncertainties. The research presented here as illustrated why PES-like approaches may be a more pragmatic strategy than strict adherence to a theory which might work under very narrow circumstances.

The underpinning current threading each of the theoretical and empirical investigations in this research has centred on the assumption that incentive structures are presumed to be the most economically efficient, environmentally effective, or socially equitable. As specified in Chapter 3, it is possible to distinguish between ecosystem functions and processes and the inherent and relational components of the services that are rendered from ecological structure for human well-being. Through the nested institutional framework offered in Chapter 3, it becomes clear that ecosystem services are not merely 'free gifts of nature' but are fundamentally social in terms of

[31] Namely REDD ('Reducing Emissions from Deforestation and Forest Degradation') initiatives

how they are perceived, valued, and appropriated (Spangenberg et al., 2014; Sikor, 2013). Accordingly, policies which seek to optimise ecosystem services through wholesale commodification (i.e. assuming they are all rival and excludable) falsely disaggregates their provisioning from a socio-cultural and political context (Spangenberg et al., 2014).

Given the non-rival nature of most watershed services (e.g. flood protection, water quality, aesthetic or cultural values), management efforts are needed to examine how PES can be integrated to promote collective action in striving for the aspirations of an integrated and adaptive water resource management effort. The literature on this subject has devoted very little attention to understanding the role of incentives in collective resource management. As the empirical analysis presented in Chapter 6 has illustrated, if social relationships and norms drive how commons resources are managed, there is substantial value in investigating how the application of incentives may foster or deter collective action by changing the interplay of 'I' and 'We' rationalities driving land-use behaviour for public goods. Consequently, through consideration of case studies of proposed PES schemes in Nepal and the Kyrgyz Republic, this research has also identified the conditions in which incentives can improve collective action for water resource management and conversely, when incentive provision may be detrimental to this objective.

More specifically in Chapter 5, it was identified through both technical and social evaluations using social multi-criteria evaluation in the Sundarijal PES proposal in Nepal that stylised incentives which are geared towards perceived deprivations in capability sets of upstream land-users was the least likely to result in social conflict while performing moderately in responding to the criteria set of proposed management alternatives for the catchment. Meanwhile in the PES pilot scheme in the Chon-Aksuu catchment in the Kyrgyz Republic described in Chapter 6, it was revealed that trust in leaders, decentralised norm enforcement, framing of collective action and image motivation all influenced an individual's propensity to participate in collectively owned irrigation canal management and to engage in repeated contributions over time. Depending on the interplay of social norms and individual expectations (e.g. private payoffs) in each village, conditional incentive provision can be synergistic or detrimental in stimulating participation to achieve specified goals. For instance, where trust in village leaders and

associated norm enforcement was the greatest, incentives provided a supplementary motivation for participation (e.g. treatment 3). Conversely, where an 'I' rationality framing was imposed, established patterns of interaction or endogenous institutions that follow a 'We' rationality became severed (e.g. treatment 2). The lack of 'other regarding' motivations such as trust in leaders and norm enforcement of existing institutions may have contributed to this effect.

This research has emphasized that the influence of outsiders and externally-determined rules which often govern PES schemes, renders the integral consideration of informal institutions, cultural practices, historical political relationships, and self-determination of local communities to be secondary to addressing the efficiency and optimality objectives of beneficiaries or those providing loans for PES initiatives. Indeed, establishing the necessary local-level institutions, trust, and negotiating mutually acceptable yet evolving management rules through the use of incentives for good land-use stewardship to provide ecosystem services will not necessarily be as expedient or more economically efficient as theoretically argued. Moreover, given that it is largely economic drivers at various scales impacting land-use patterns, it is unclear how getting the 'price right' for services which are not excludable and rival can ever be more economically efficient than those goods that are amenable to commodification. Simplistic attempts to equate private pay-offs to the broader milieu of social, political and economic drivers causing water resource degradation grossly underestimates socio-ecological complexity and may result in greater inequality and environmental degradation in the long-term.

As a result of constantly evolving economic drivers modifying opportunity costs of land-users combined with the inertia of existing and expected disruption of hydrological processes in a watershed, the design of the 'optimum' PES scheme is a frequently moving target. Climate change impacts contribute to the uncertainty of ecological consequences as well as to the volatility of socio-economic and cultural systems. As Chapter 4 has argued and illustrated, stylistic attempts of varying degrees of sophistication to target PES to be cost-effective, environmentally effective, economically efficient and/or poverty-reducing are ultimately reductionist strategies for understanding the underlying relationships between society and nature. In addition to the uncertain ecological and economic impacts at play, poverty is a multidimensional and subjective state of well-being requiring closer attention to the various

dimensions of social justice: recognition of knowledge, norms and values; procedural inclusiveness, and distributional equity (McDermott et al., 2013). The empirical investigation presented in Chapter 4 concluded that the identification of salient political objectives for PES as externally prioritised considered distributional aspects but failed to consider neither the recognition of informal institutions nor the procedural justice of devising management solutions. As such, targeted payment designs were unfavourable for upstream service providers of valued water quality services despite attempts to achieve multiple objectives in an efficient manner. As shown in Chapter 5, this research has advanced efforts to disaggregate equity concerns from the more political and normative characterisations of *injustice*, including the possibility to compare PES with other environmental management alternatives. Moving forward, it is imperative to identify how PES interventions can advance environmental justice according to certain value ethics, yet generate injustice from other another set of ethics- with each set being equally legitimate.

Based on the results of this body of research, it can be concluded that PES can play an important role for integrated and adaptive water resource management if incentive-based negotiations are integrated within collective resource management for regulating and cultural watershed services that are inherently non-rival. This is in contrast to the current thrust of scholarship on PES which continues to frame the provision of incentives as a market-based mechanism. Understanding how incentives can facilitate or detract collective action by leveraging upon plural values in light of substantial ontological and epistemological uncertainty can aid in improving socio-ecological resiliency.

The specific conclusions drawn from this body of research and which address the specific objectives identified in Section 1.2.2 are as follows:

i) A conceptual framework for characterising ecosystem services was developed and discussed in relation to IWRM. PES arrangements are best suited for negotiating the management of watershed services that are non-rival yet whose delivery is contingent upon the actions of land-use managers who either formally or informally have land-rights impacting the delivery of services such as flood protection and water quality to service beneficiaries downstream. In this

manner the social understandings deeply embedded in ecological outcomes can be deliberated to better link changes in ecological functions to the social mobilisation and attribution of use-value to ecological services (objectives 1 and 2);

ii) A spatially explicit typology was developed for revealing the trade-offs between various politically-salient objectives in the design of PES payments, as identified from the theoretical literature on PES. The application of this typology matrix in a given case study of PES consideration in Nepal revealed that none of the targeted designs overlapping the various objectives together *recognised* local informal institutions, cultural identities, nor the political-historical context of power asymmetries as well as ingrained perceptions and relationships between actors. Moreover, the *procedural justice* of determining how and why PES design objectives become politically important was also absent. As a consequence, attempts to be holistic in meeting efficiency and equity goals for PES were not rooted in local legitimacy (objectives 3 and 4);

iii) Social multi-criteria evaluation (SMCE) was applied to consider different PES alternatives for watershed management in a given catchment in Nepal which is characterised by intense socio-economic and ecological uncertainty. This valuation methodology explicitly recognised plural values (measured as criteria) in breaking down the implications of various management strategies without allowing for compensation of one value-set over another (i.e. no use of weights, unlike other compensatory multi-criteria methods). The SMCE identified highest ranking management alternatives according to the full set of valued criteria and revealed the trade-offs between a technical evaluation and the social equity implications in comparing alternatives among separate coalitions of social actors and their interests. Despite strong concordance for alternatives which build upon buffer zone management and decentralised decision-making for devising resource management rules, several of the PES payment alternatives were illustrated to be more socially acceptable once the risk of social conflict over a divergence of interests was revealed (objective 5);

iv) The elicitation of criteria in the SMCE was framed in defining payments according to a justice perspective known as the capabilities approach. Through this approach, PES payments

(e.g. SMCE management alternatives) were designed according to recognised deprivations in 'doing and being' what is of value in life. In this manner, the choice of management strategy best reflects justice according to individual needs and perspectives as opposed to an external assessment of a 'just state of affairs'. Thus, the payment designs are scored in the SMCE in terms of the extent to which they satisfy capability sets that are recognised as deprived by potential service providers. In this way, the management of ecosystem services and improvement of human well-being were explicitly linked (objective 6);

v) Through a behavioural economics analysis involving small-scale economic experiments in an empirical setting in the Kyrgyz Republic where PES is being piloted for water quality management, it was determined that the framing of market versus social norms influenced participation in a collective resource management activity across time. Specifically, decentralised norm enforcement, trust in local leaders, as well as image motivation had a substantial influence on behavioural outcomes. The introduction of private and monetary incentives for enhancing participation resulted in variable outcomes depending on the strength of market versus social norms in a given context. The design of PES payments for collective resource management requires identifying the conditions in which pro-social norms and individual gain can become synergistic in a given setting to improve the likelihood that conditionality requirements associated with payment negotiation are achieved. Thus, when workable local institutions exist and are cemented by strong social norms and trust, external incentives can serve to improve collective action. The key lies in recognising that local existing institutions matter for environmental management and these may already promote or preclude the feasibility of introducing private incentives (or 'I' rationality) in a particular context. However, it is emphasized that where strong functional institutions and norms exist for promoting collective action, there may be little to gain (and perhaps more to lose) by introducing external incentives, unless it can be determined that incentives can build upon those existing institutions to facilitate additional collective action.

vi) In responding to the general objectives stated in Section 1.2.1, this body of research has conceptually and empirically argued for more deliberative negotiations in the access, management and valuation of watershed resources and the services that derive from them.

Framing value as economic benefit alone is an exercise of power which precludes the consideration of legitimate knowledge systems involving long-established relationships between people and land as defined through cultural processes. Accordingly, failure to incorporate justice in PES will destabilize local self-determination and risk unravelling well-functioning institutions and social processes for long-term sustainable solutions.

Chapter 8

CLAIMS OF ORIGINALITY AND RECOMMENDATIONS FOR FURTHER RESEARCH

8.1. Contributions to Knowledge

The work presented in this manuscript provides original contributions to the scholarship on PES, which is currently among the most championed policy mechanisms for addressing natural resource management conflicts, with specific emphasis on watershed ecosystem services. The main contributions of this dissertation are as follows:

1. The development of a conceptual framework which recognises the contribution of inherent natural and relational social practices that characterise ecosystem services. The precise role of PES as a management or policy technique for realising IAWRM is specified. This conceptual framework will permit practitioners and donor agencies to better understand the conditions for which PES and other policy mechanisms are best placed to sustain or improve the management of a range of watershed ecosystem services.

2. This research has developed a targeting typology for assessing the trade-offs between objectives in designing PES payments for watershed management. The typology presented herein is presented to serve the argument that externally defined objectives that fail to integrate multiple knowledge systems will fail to attract local legitimacy. This typology can also be applied through more participatory mapping exercises which do integrate plural values in order to deliberate upon the most acceptable payment strategy and how this might evolve over time given dynamic ecological and socio-economic drivers of change.

3. This research is the first study to use social multi-criteria evaluation as an alternative valuation method to compare PES and non-PES management alternatives to address a socio-ecological conflict over water quality services. Unlike other multi-criteria methods, cost-benefit analysis,

systems dynamics modelling or other policy evaluation techniques, SMCE accounts for the diversity of value systems under weak-comparability, that is, without reducing the contribution of diverse values through measurement standardisation. Accordingly, it is best suited in conflicts of high uncertainty, power differentials between actors, inequality in purchasing power, and rapidly evolving underlying drivers of system change. Such contexts often characterise socio-ecological conflicts where PES incentives are proposed to improve social and environmental outcomes, yet mechanisms to evaluate management alternatives, including PES, under such conditions have been limited. The examination of PES feasibility in the Sundarijal catchment of Nepal, contributes to filling this research gap.

4. In the empirical study examined, criteria for evaluating PES designs were defined with the objective of linking payment negotiation and distribution to fundamental injustices perceived at the household level. Hence, this is the first study to consider PES payments as 'capabilities' for improving well-being, rather than as purely economic benefits. In this manner, practitioners of PES implementation can better link negotiated incentives to the inseparable social and natural processes which co-produce ecosystem services of value.

5. This research contributes to the empirical body of knowledge on the behavioural implications of providing incentives for public good provisioning, and specifically in relation to PES incentives for watershed management. This is a nascent area of scholarship with few and relatively recent studies conducted in relation to the feasibility of PES implementation. The economic experiments carried out here integrate with existing institutional practices which characterise collective resource management in the case study context in order to obtain a more realistic depiction of the influence of different incentives or incentive-framings on behaviour. This aspect distinguishes the present research from previous similar studies (e.g. Kerr et al., 2012). Furthermore, the study incorporates repeated interactions across four weeks to assess whether differences exist both between and within incentive and non-incentive treatments in relation to management of the collective commons.

8.2. Overall Study Limitations

The limitations bounding the conclusions of the research carried out can be characterised as the following:

a) The theoretical re-framing of PES within a nested institutional governance framework has yet to be empirically tested, and thus cannot be effectively made as an argument to influence PES theory. Future research should identify how elements of 'nesting' ecosystem services according to their economic characteristics could occur through resource management arrangements in a particular context. Incentive-based negotiations for regulating and cultural services can be promoted where norms of reciprocity and trust exist and positively support such arrangements.

b) The targeting matrix presented in Chapter 4 is not meant to serve as a tool to be applied in other contexts. It is merely an analytical and illustrative argument to indicate the trade-offs in PES objectives between external actors and local communities. The subjectivity inherent in the choice of delimiters and assumptions made in the interpolation exercise are high. However, as mentioned earlier, the emphasis on the approach taken is on the consequences of payment targeting which excludes the multiple ways by which diverse social actors make sense of their natural and social environments.

c) While engagement with social actors and transparency of scoring proposed management alternatives against the criteria set was carried out, the SMCE process remains highly uncertain. The uncertainty inherent in the process reflects the difficulty of social actors to predict the extent to which proposed alternatives would attain the identified criteria set. The use of a sensitivity analysis, both in the levels of compensation allowed in scoring the alternatives and in the thresholds of preference ascribed in pairwise comparisons, served to minimise this uncertainty. While several of the management alternatives are no longer considered applicable for the management of Shivapuri-Nagarjun National Park (particularly the PES alternatives), the exercise serves to highlight how a complex socio-ecological problem can be deconstructed to better deliberate upon the ramifications of different courses of action and the potential social conflicts these may create.

d) The design of the experimental setting of the incentive treatments in Chapter 6 is based on the assumption that the villages are similar in several aspects (e.g. socio-economic, population size,

distance to markets), but not to the levels of social capital that exist. As such, it is difficult to justify a 'control' treatment, since the social characteristics of each village render a comparison across them impossible. It is for this reason that the exploratory nature of the study presented is emphasized. It is hoped that further study which can seek to better control for the variation between treatments in an empirical setting can take these findings to draw more robust conclusions. The relatively small number of participants and interviews conducted to understand motivation for participation also limits the validity of findings.

8.3. Recommendations for Further Research

1. It would be useful to apply the nested institutional framework as well as the targeting typology matrix in understanding how natural resources are best managed to better understand when PES schemes are more or less likely to be successful. This would involve identifying key aspects of PES initiatives which explain which objectives between value systems were excluded in decision-making. More fundamentally, a meta-study of this nature would serve to buttress the evidence in arguing against the use of sheer-market design principles for ecosystem services which are neither excludable nor rival.

2. Examining how ecosystem processes or functions provide the opportunity or context for various social actors to derive benefits or values based upon unique knowledge systems linked to ecological understandings. The diversity of benefits that emerge from a set of ecological functions is conducive to varying shades of appropriation (e.g. excludability) depending on the social conceptualisation of the benefit. Taking such an approach can highlight the range of social and ecological implications of a particular management proposal and reveal complementarities between objectives between different social groups to better identify when and how negotiations such as PES can be successful in meeting the needs and aspirations of different groups, given the diversity of benefits that different groups perceive from their environment.

3. As alluded to in Section 8.1. above, there is substantial opportunity for promoting a more deliberative and self-reflexive process of spatially targeting PES designs. Such a process could be facilitated through collaborative group workshops, in which priority objectives are internally

negotiated involving a range of social actors rather than externally defined by PES proponents or donors alone. This process could result in a participatory mapping exercise which indicates, for example, the minimum-level environmental flows of a catchment that are collectively decided to be necessary for both ecological resiliency and human well-being needs as defined by a plurality of voices. The trade-offs between different participatory maps can then be collectively deliberated upon to arrive at a possible management strategy. SMCE combined within this participatory mapping of targeted objectives could be a salient methodology to this end.

4. Taking a political ecology dialectical perspective in examining the extent to which a capabilities-focused approach to social equity has been or potentially could result in greater recognition, procedural and distributive justice outcomes for eco-social conflicts surrounding resource extractive frontiers. Such an approach would offer an indication as to the sets of organisations, rules and social norms that have developed over the equitable management of water resources. Further, alternative political ecologies would highlight the extent to which greater decentralisation of resource management to the community level may enhance justice-oriented outcomes.

5. Further behavioural economic experiments which assess how introducing different incentive types and then removing them at some point in time influences long-term permanence of collective resource management or positive environmental stewardship. These experiments could be tested in communities with stronger social norms and local institutions for collective management versus other communities that are more market-integrated or have less well functioning institutions for managing collective resources at the community level.

6. Additional economic experiments could assess the gradual increase or reduction of incentives over time to assess implications for maintaining or increasing contributions of collective action over time across comparable settings that may differ only in terms of the strength of social norms or institutions for managing collective resources. Attaching conditionality requirements of specified effort in contributions to increasing or declining incentives in contexts of weaker and stronger social norms may provide insight into the interplay of ‘I’ and ‘We’ rationalities that sustain ecological management over time.

References

Abson, D.J., von Wehrden, H., Baumgärtner, S., Fischer, J., Hanspach, J., Härdtle, W., Heinrichs, H., Klein, A.M., Lang, D.J., Martens, P., and Walmsley, D., 2014. Ecosystem services as a boundary object for sustainability. Ecological Economics 103, 29-37.

Ahmad, Q.K., 2003. Towards Poverty Alleviation: The Water Sector Perspectives. Water Resources Development 19 (2), 263-277.

Alix-Garcia, J., de Janvry, A., Sadoulet, E., 2008. The role of deforestation risk and calibrated compensation in designing payments for environmental services. Environment and Development Economics 13, 375-394.

Alkire, S., Santos, M.E., 2010. Acute Multidimensional Poverty: A New Index for Developing Countries. Oxford Poverty and Human Development Initiative Working Paper No. 38. Oxford.

Ariely, D., Bracha, A., and Meier, S., 2009. Doing good or doing well? Image motivation and monetary incentives in behaving prosocially. The American Economic Review 99 (1), 544-555.

Arrow, K. and Dasgupta, P., 2004. Are we consuming too much? Journal of Economic Perspectives 18, 147-172.

Ballet, J., Koffi, J-M., Pelenc, J., 2013. Environment, justice and the capability approach. Ecological Economics 85, 28-34.

Bénabou, R. and Tirole, J., 2006. Incentives and Prosocial Behaviour. American Economic

Review 96 (5), 1652-1678.

Biswas, A. K., 2004. Integrated water resources management: a reassessment: a water forum contribution. Water International 29 (2), 248-256.

Bidaud, C., Méral, P., Andriamahefazafy, F., Serpantié, G., Cahen-Fourot, L. and Tollier, A., 2013. Institutional and Historical Analysis of Payments for Ecosystem Services in Madagascar. *In*: Muradian, R., and Rival, L. (Eds.). *Governing the Provision of Ecosystem Services*. Springer Netherlands. pp 207-233.

Bockstaller, C., Girardin, P. and van der Werf, H.M.G., 1997. Use of agro-ecological indicators for the evaluation of farming systems. European Journal of Agronomy 7, 261-270.

Bonn, A., Reed, M.S., Evans, C.D., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M-A., Birnie, D., 2014. Investing in nature: Developing ecosystem service markets for peatland restoration. Ecosystem Services 9, 54-65.

Boody, G., Vondracek, B., Andow, D.A., Krinke, M., Westra, J., Zimmerman, J. and Welle, P., 2005. Multifunctional agriculture in the United States. Bioscience 55 (1), 27-38.

Bowles, S., 2008. Policies designed for self-interested citizens may undermine "the moral sentiments": Evidence from economic experiments. Science 320 (5883), 1605-1609.

Brondizio, E. S., Ostrom, E. and Young, O. R., 2009. Connectivity and the Governance of Multilevel Social-Ecological Systems: The Role of Social Capital. Annual Review of Environment and Resources 34, 253-278.

Campbell, B., Sayer, J. A., Frost, P., Vermeulen, S., Perez, M. R., Cunningham, A. and Prabhu, R., 2001. Assessing the performance of natural resource systems. Conservation Ecology 5 (2), 22.

Cardenas, J.C. and Carpenter, J.P., 2008. Behavioural development economics: Lessons from field labs in the developing world. Journal of Development Studies 44, 337-364.

Cardwell, H. E., Cole, R. A., Cartwright, L. A. and Martin, L. A., 2006. Integrated Water Resources Management: Definitions and Conceptual Musings. Journal of Contemporary Water Research & Education 135, 8-18.

Castree, N., 2008. Neoliberalising nature: the logics of deregulation and regulation. Environment and Planning A 40 (1), 131-152.

Chomitz, K.M. and Gray, D.A., 1996. Roads, Land Use, and Deforestation: A Spatial Model Applied to Belize. World Bank Economic Review 10 (3), 487-512.

Clark, D.A., 2009. Adaptation, Poverty and Well-Being: Some Issues and Observations with Special Reference to the Capability Approach and Development Studies 1. Journal of Human Development and Capabilities 10 (1), 21-42.

Clements, T., John, A., Nielsen, K., An, D., Tan, S., and Milner-Gulland, E. J. 2010. Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecological Economics 69 (6), 1283-1291.

Coase, R., 1960. The Problem of Social Cost. The Journal of Law and Economics 3, 1-44.

Cook, B.R. and Spray, C.J., 2012. Ecosystem services and integrated water resource management: Different paths to the same end?" Journal of Environmental Management 109, 93-100.

Corbera, E., Kosoy, N., and Martinez Tuna, M., 2007. Equity implications of marketing

ecosystem services in protected areas and rural communities: Case studies from Meso-America. Global Environmental Change 17 (3), 365-380.

Daily, G.C., 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press, Washington, DC.

Dale, V.H. and Polasky, S., 2007. Measures of the effects of agricultural practices on ecosystem services. Ecological Economics 64, 286-296.

Deci, E.L., 1971. Effects of externally mediated rewards on intrinsic motivation. Journal of Personality and Social Psychology 18, 105-115.

de Groot, R.S., Wilson, M.A. and Boumans, R.M.J., 2002. A typology for the classification, description, and valuation of ecosystem functions, goods and services. Ecological Economics 41, 393-408.

Duarte, R., Sanchez-Choliz, J. and Bielsa, J., 2002. Water use in the Spanish economy: an input-output approach. Ecological Economics 43, 71-85.

Duraiappah, A. K., 2004. Exploring the links: human well-being, poverty, and ecosystem services. The United Nations Environment Programme and the International Institute for Sustainable Development.

Duraiappah, A. K., 2011. Ecosystem Services and Human Well-being: Do Global Findings Make Any Sense? BioScience 61 (1), 7-8.

Engel, S., Pagiola, S., and Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65 (4), 663-674.

Engle, N.L., Johns, O.R., Lemos, M-C., and Nelson, D.R., 2011. Integrated and Adaptive

Management for Water Resources: Tensions, Legacies, and the Next Best Thing. Ecology and Society 16 (1), 19-29.

Farley, J., 2010. Conservation through the Economics Lens. Environmental Management 45, 26-38.

Farley, J. and Costanza, R., 2010. Payments for ecosystem services: from local to global. Ecological Economics 69 (11), 2060-2068.

Fehr, E. and Gächter, S., 2002. Altruistic punishment in humans. Nature 415 (6868), 137-140.

Fehr, E. and Falk, A., 2002. Psychological foundations of incentives. European Economic Review 46 (4), 687-724.

Falk, A. and Kosfeld, M., 2006. The hidden costs of control. The American Economic Review 96 (5), 1611-1630.

Ferraro, P.J., Kiss, A., 2002. Direct payments to conserve biodiversity. Science 298, 1718-1719.

Ferraro, P.J. and Simpson, R.D., 2002. The cost effectiveness of conservation payments. Land Economics 78, 339-353.

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K. and Burgess, N.D., 2010. Common pool resource management and PES: lessons and constraints for water PES in Tanzania. Ecological Economics 69 (6), 1253-1261.

Folke, C., Hahn, T., Olsson, P. and Norberg, J., 2005. Adaptive governance of socio-ecological systems. Annual Review of Environmental Resources 30, 441-473.

Frey, B., 1994. How intrinsic motivation is crowded out and in. Rationality and Society 6 (3), 334-352.

Functowicz, S.O., Ravetz, J., 1994. Uncertainty, complexity and post-normal science. Environmental Toxicology and Chemistry 13, 1881-1885.

Gamboa, G., 2006. Social Multi-Criteria Evaluation in practice: Two real-world case studies. Doctoral Thesis. Universitat Autònama de Barcelona. Barcelona, Spain.

Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. Environmental Management 45, 488-501.

Geist, H.J. and Lambin, E.F., 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. BioScience 52 (2), 143-150.

Ghazoul, J., Garcia, C. and Kushalappa, C., 2011. Landscape Labelling Approaches to PES: Bundling Services, Products and Stewards. *In* Payments for Ecosystem Services and Food Security. Food and Agriculture Organization of the United Nations, Rome.

Gintis, H., Henrich, J., Bowles, S., Boyd, R., Fehr, E., 2008. Strong Reciprocity and the Roots of Human Morality. Social Justice Research 21 (2), 241-253.

Gneezy, U., Meier, S., and Rey-Biel, P., 2011. When and Why Incentives (Don't) Work to Modify Behavior. Journal of Economic Perspectives 25 (4), 191-210.

Gómez-Baggethun, E., deGroot, R., Lomas, P.L., and Montes, C., 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological Economics 69 (6), 1209-1218.

Grafton, Q.R. and Hussey, K., 2011. *Water Resource Planning and Management*. Cambridge University Press, Cambridge.

Global Water Partnership (GWP), 2000. Integrated Water Resources Management Technical Committee Background Paper 4. Stockholm.

Halbe, J., Pahl-Wostl., C., Sendzimir, J., and Adamowski, J., 2013. Towards adaptive and integrated management paradigms to meet the challenges of water governance. Water Science and Technology 67 (11), 2651-2660.

Henrich, J., Boyd, R., Bowles, S., Camerer, C., Fehr, E., Gintis, H., ... and Tracer, D., 2005. "Economic man" in cross-cultural perspective: Behavioral experiments in 15 small-scale societies. Behavioral and Brain Sciences 28 (6), 795-815.

Houser, J.N., Mulholland, P.J. and Maloney, K., 2006. Stream chemistry indicators of disturbance on military reservations. Journal of Environmental Quality 35, 352-365.

International Water Association (IWA), 2002. Industry as a Partner for Sustainable Development: Water Management. IWA/UNEP. London, UK. Beacon Press. Available online: http://new.unep.org/publications/search/pub_details_s.asp?ID=180. [Accessed: 21 September, 2011].

Jax, K., 2010. *Ecosystem Functioning*. Cambridge University Press, Cambridge.

Jeffrey, P. and Gearey, M., 2006. Integrated water resource management: lost on the road from ambition to realisation? Water Science and Technology 53 (1), 1-8.

Kallis, G., 2011. In defence of degrowth. Ecological Economics 70 (5), 873-880.

Kallis, G., Gómez-Baggethun, E. and Zografos, C., 2013. To value or not to value? That is not the question. Ecological Economics 94, 97-105.

Kamenica, E., 2012. Behavioral Economics and Psychology of Incentives. Annual Review of Economics 4 (1), 427-452.

Kerr, J., Vardhan, M., and Jindal, R., 2012. Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. Ecological Economics 73, 220-227.

Kerr, J., Vardhan, M., and Jindal, R., 2014. Incentives, conditionality and collective action in payment for environmental services. International Journal of the Commons 8 (2), 595-616.

Kiersch, B., Hermans, L. and Van Halsema, G., 2005. Payment schemes for water-related environmental services: a financial mechanism for natural resources management experience from Latin America and the Caribbean. Food and Agriculture Organization of the United Nations. Rome, Italy. Available online: www.unece.org/env/water/meetings/payment_ecosystems/discpapers/fao.pdf. [Accessed: 2 February, 2012].

Kolinjivadi, V.K. and Sunderland, T., 2012. A review of two payment schemes for watershed services from China and Vietnam: The interface of government control and PES theory. Ecology and Society 17 (4), 10-24.

Kosoy, N., Martinez Tuna, M., Muradian, R. and Martinez-Alier, J., 2007. Payments for environmental services in watersheds: insights from a comparative study of three cases in Central America. Ecological Economics 61, 446-455.

Kosoy, N. and Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecological Economics 69 (6), 1228-1236.

Landell-Mills, N. and Porras, I., 2002. Silver Bullet or Fools' Gold? A Global Review of Markets for Forest Environmental Services and Their Impacts on the Poor. International Institute for Environment and Development. London.

Leimona, B. and de Groot, R., 2010. Payments for Environmental Services: The Need for

Redefinition? Mountain Forum Bulletin, (January), 9-10.

Leimona, B., van Noordwijk, M., de Groot, R. and Leemans, R., 2015. Fairly efficient, efficiently fair: Lessons from designing and testing payment schemes for ecosystem services in Asia. Ecosystem Services 12, 16-28.

Liu, S., Crossman, N.D., Nolan, M. and Ghirmay, H., 2013. Bringing ecosystem services into integrated water resources management. Journal of Environmental Management 129, 92-102.

Lockie, S., 2013. Market instruments, ecosystem services, and property rights: assumptions and conditions for sustained social and ecological benefits. Land Use Policy 31, 90-98.

Millennium Ecosystem Assessment (MA), 2005. *In*: Hassan, R., Scholes, R., Ash, N. (Eds.). Ecosystems and Human Well-Being: Current state and Trends, vol. 1. Island Press, Washington, D.C.

Maloney, K.O., Mulholland, P.J. and Feminella J.W., 2005. Influence of catchment-scale military land use on physiochemical conditions in small Southeastern Plains streams (USA). Environmental Management 35, 677-691.

Martínez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecological Economics 26, 277-286.

Martínez Tuna, M. and Kosoy, N., 2007. Compensaciones monetarias y conservación de bosques Pagos por servicios ambientales y pobreza en una comunidad rural en Honduras. Revista Iberoamericana de Economía Ecológica 6, 55-66.

Matulis, B., 2014. The economic valuation of nature: A question of justice? Ecological Economics 104, 155-157.

McAfee, K., 2012. The Contradictory Logic of Global Ecosystem Services Markets. Development and Change 43 (1), 105-131.

McAfee, K. and Shapiro, E., 2012. Payments for Ecosystem Services in Mexico: Nature, Neoliberalism, Social Movements, and the State. Annals of the Association of American Geographers 100 (3), 579-599.

McDermott, M., Mahanty, S., and Schreckenberg, K., 2013. Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. Environmental Science & Policy 33, 416-427.

McDonnell, R.A., 2008. Challenges for integrated water resources management: how do we provide the knowledge to support truly integrated thinking? Water Resources Development 24 (1), 131-143.

Medema, W., McIntosh, B. S. and Jeffrey, P. J., 2008. From Premise to Practice: a Critical Assessment of Integrated Water Resources Management and Adaptive Management Approaches in the Water Sector. Ecology & Society 13 (2), 29-46.

Meier, S., 2007. Do subsidies increase charitable giving in the long run? Matching donations in a field experiment. Journal of the European Economic Association 5 (6), 1203-1222.

Merrey, D.J., Drechsel, P., Penning de Vries, F.W.T. and Sally, H., 2005. Integrating "livelihoods" into integrated water resources management: taking the integration paradigm to its logical next step for developing countries. Regional Environmental Change 5, 197-204.

Mitchell, R.K., Agle, B.R., and Wood, D.J., 1997. Toward a Theory of Stakeholder Identification and Salience: Defining the Principle of Who and What Really Counts. The Academy of Management Review, 22 (4), 853-886.

Moench, M., Dixit, A., Janakarajan, S., Rathore, M.S. and Mudrakartha, S., 2003. The fluid

mosaic: water governance in the context of variability, uncertainty and change. Nepal Water Conservation Foundation. Kathmandu, Nepal. Available online: http://idrc.org/uploads/user-S/10492953541Fluid_Mosaic21.pdf. [Accessed 18 May, 2012].

Munda, G., 2004. Social multi-criteria evaluation (SMCE): methodological foundations and operational consequences. European Journal of Operational Research 158 (3), 662-677.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202-1208.

Muradian, R., Arsel, M., Pellegrini, L., ... and Adaman, F., 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. Conservation Letters, 6 (4), 274-279.

Narloch, U., Pascual, U., Drucker, A.G., 2012. Collective Action Dynamics under External Rewards: Experimental Insights from Andean Farming Communities. World Development 40, 2096-2107.

Neumeyer, E., 2007. *Weak versus Strong Sustainability: Exploring the Limits of Two Opposing Paradigms*. Edward Elgar Publishing, Inc., Cheltenham.

Nguyen, T.B.T., Nguyen, C.T., Nguyen, P.T., Pham, T.N., Bonnardeaux, D., & Riedel, D.C., 2011. Payments for Forest Environmental Services: A Case Study on Pilot Implementation in Lam Dong Province Vietnam from 2006-2010. Winrock International. Available online: http://www.arbcp.com/data/pdf/CasestudyEN.pdf. [Accessed 10 March, 2011].

Norgaard, R.B., 2010. Ecosystem services: from eye-opening metaphor to complexity blinder. Ecological Economics 69 (6), 1219-1227.

Norman, E.S. and Bakker, K., 2009. Transgressing scales: water governance across the Canada-US borderland. Annals of the Association of American Geographers 99 (1), 99-117.

Nussbaum, M.C., 2003. Capabilities as Fundamental Entitlements: Sen and Social Justice. Feminist Economics 9 (2-3), 33-59.

Olsson, P., Folke, C. and Berkes, F., 2004. Adaptive co-management for building resilience in social-ecological systems. Environmental Management 34 (1), 75-90.

Ostrom, E., 1990. *Governing the commons: The Evolution of institutions for collective action*. Cambridge University Press, Cambridge.

Ostrom, E., 2005. *Understanding institutional diversity.* Princeton University Press, Princeton.

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

Pagiola, S., Arcenas, A. and Platais, G., 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33 (2), 237-253.

Pagiola, S. and Platais, G., 2007. Payments for Environmental Services: From Theory to Practice. World Bank. Washington, DC.

Pahl-Wostl, C., 2009. A conceptual framework for analysing adaptive capacity and multi-level learning processes in resource governance regimes. Global Environmental Change 18, 354-365.

Pahl-Wostl, C., Sendzimir, J., Jeffrey, P., Aerts, J., Berkamp, G. and Cross, K., 2007. Managing

change toward adaptive water management through social learning. Ecology and Society 12 (2), 30.

Pahl-Wostl, C., Jeffrey, P., Isendahl, N. and Brugnach, M., 2011. Maturing the New Water Management Paradigm: Progressing from Aspiration to Practice. Water Resource Management 25, 837-856.

Parkes, M.W., Morrison, K.E., Bunch, M.J., ... and Waltner-Toews, D., 2010. Towards integrated governance for water, health and social-ecological systems: The watershed governance prism. Global Environmental Change 20, 693-704.

Pascual, U., Muradian, R., Rodriguez, L. C. and Duraiappah, A., 2010. Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69, 1237-1244.

Pascual, U., Phelps, J., Garmendia, E., ... and Muradian, R., 2014. Social equity matters in payments for ecosystem services. BioScience. *In Press,* 1-10.

Pattanayak, S.K., Wunder, S., Ferraro, P.J., 2010. Show me the money: Do payments supply environmental services in developing countries? Review of Environmental Economics and Policy 4, 254-274.

Perrot-Maître, D., 2006. The Vittel payments for ecosystem services: a "perfect" PES case? International Institute for Environment and Development. London. Available online: http://pubs.iied.org/pdfs/G00388.pdf.

Peterson, M.J., Hall, D.M., Feldpausch-Parker, A.M. and Peterson, T.R., 2010. Obscuring Ecosystem Function with Application of the Ecosystem Services Concept. Conservation Biology 24 (1), 113-119.

Pirard, R., 2012. Market-based instruments for biodiversity and ecosystem services: A lexicon.

Environmental Science and Policy 19, 59-68.

Polishchuk, Y., Rauschmayer, F., 2012. Beyond "benefits"? Looking at ecosystem services through the capability approach. Ecological Economics 81, 103-111.

Purushothaman, S., Thomas, B.K., Abraham, R. and Dhar, U., 2013. Beyond money metrics: Alternative approaches to conceptualising and assessing ecosystem services. Conservation and Society 11 (4), 321-325.

Rahaman, M.M. and Varis, O., 2005. Integrated water resources management: evolution, prospects and future challenges. Sustainability: Science, Practice, & Policy 1 (1), 15-21.

Rawlins, M.A. and Westby, L., 2013. Community participation in payment for ecosystem services design and implementation: An example from Trinidad. Ecosystem Services 6, 117-121.

Reddy, V.R., Reddy, M.G., Galab, S., Soussan, J. and Springate-Baginski, O., 2004. Participatory Watershed Development in India: Can it Sustain Rural Livelihoods? Development and Change 35 (2), 297-326.

Rennie, J. K. and Singh, N., 1996. Participatory Research for Sustainable Livelihoods: A Guidebook for Field Projects. International Institute for Sustainable Development, Winnipeg. Available online: http://www.iisd.org/publications/pub.aspx?id=308. [Accessed 3 March, 2012].

Robeyns, I., 2006. The Capability Approach in Practice. Journal of Political Philosophy 14(3), 351-376.

Rockström, J., Steffen, W., Noone, K., ... and Foley, J., 2009. A safe operating space for humanity. Nature 461, 472-475.

Rodriguez, L. C., Pascual, U., Muradian, R., Pazmino, N. and Whitten, S., 2011. Towards a unified scheme for environmental and social protection: Learning from PES and CCT experiences in developing countries. Ecological Economics 70, 2163-2174.

Santelmann, M., White, D., Freemark, K., ... and Debinski, D., 2004. Assessing alternative futures for agriculture in the U.S. Corn Belt. Landscape Ecology 19 (4), 357-374.

Scerri, A., 2012. Ends in view: The capabilities approach in ecological/sustainability economics. Ecological Economics 77, 7-10.

Sen, A., 1999. *Development as freedom*. Oxford University Press, Oxford.

Sen, A., 2010. *The Idea of Justice*. Harvard University Press, Cambridge.

Sikor, T., 2013. Chapter 1: Introduction: Linking ecosystem services with environmental justice. *In*: Sikor, T. (Ed.). *The Justices and Injustices of Ecosystem Services*. Routledge, London.

Sommerville, M., Jones, J.P.G., Rahajaharison, M. and Milner-Gulland, E.J., 2010. The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. Ecological Economics 69, 1262-1271.

Spangenberg, J.H., Görg, C., Truong, D.T., Tekken, V., Bustamente, J.V. and Settele, J., 2014. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. International Journal of Biodiversity Science, Ecosystem Services and Management 10 (1), 40-53.

Spash, C.L., 2011. Social ecological economics: Understanding the past to see the future. American Journal of Economics and Sociology 70 (2), 340-375.

Swallow, S.K., 2013. Demand-side Value for Ecosystem Services and Implications for

Innovative Markets: Experimental Perspectives on the Possibility of Private Markets for Public Goods. Agricultural and Resource Economics Review 42 (1), 33-56.

Tacconi, L., 2012. Redefining payments for environmental services. Ecological Economics 73, 29-36.

Tilman, D., Cassman, K., Matson, P., Naylor, R. and Polasky, S., 2002. Agricultural sustainability and the costs and benefits of intensive production practices. Nature 418, 671-677.

Travers, H., Clements, T., Keane, A., and Milner-Gulland, E.J., 2011. Incentives for cooperation: The effects of institutional controls on common pool resource extraction in Cambodia. Ecological Economics 71, 151-161.

United Nations Economic Commission for Europe (UNECE), 2007. Recommendations on Payments for Ecosystem Services in Integrated Water Resources Management. Geneva. Available online: http://www.unece.org/fileadmin/DAM/trans/osce/osceunece/Misc_Zaragoza_Recommendations.pdf [Accessed 28 November, 2011].

Unruh, G.C., 2002. Escaping carbon lock-in. Energy Policy 30 (4), 317-325.

Van Hecken, G., Bastiaensen, J., Vásquez, W.F., 2012. The viability of local payments for watershed services. Empirical evidence from Matiguás, Nicaragua. Ecological Economics 74, 169-176.

Vatn, A., 2009. An institutional analysis of methods for environmental appraisal. Ecological Economics 68, 2207-2215.

Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics 69 (6), 1245-1252.

Vignola, R., Koellner, T., Scholz, R.W., McDaniels, T.L., 2010. Decision making by farmers regarding ecosystem services: factors affecting soil conservation efforts in Costa Rica. Land Use Policy 27 (4), 1132-1142.

Vignola, R., McDaniels, T.L. and Scholz, R.W., 2012. Negotiation analysis for mechanisms to deliver ecosystem services: The case of soil conservation in Costa Rica. Ecological Economics 75, 22-31.

Vollan, B., 2008. Socio-ecological explanations for crowding-out effects from economic field experiments in southern Africa. Ecological Economics 67 (4), 560-573.

Wester, P., Merrey, D.J. and de Lange, M., 2003. Boundaries of consent: representation in River Basin management in Mexico and South Africa. World Development 31 (5), 797-812.

World Bank, 2003. Water Resources Sector Strategy: Strategic Directions for World Bank Engagement, Washington, DC. Available online: http://wwwwds.worldbank.org/servlet/WDSContentServer/WDSP/IB/2004/06/01/000090341_20040601150257/Rendered/PDF/28114.pdf. [Accessed 22 January, 2012].

Wunder, S., 2005. Payments for Environmental Services: Some Nuts and Bolts. CIFOR Occasional Paper No. 42. Center for International Forestry Research, Bogor, Indonesia.

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. Conservation Biology 21, 48-58.

Wunder, S., 2008. Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. Ecological Economics 65, 834-852.

Wunscher, T., Engel, S. and Pagiola, S., 2008. Taking stock: a comparative analysis of payments

for environmental services programs in developed and developing countries. Ecological Economics 65 (4), 822-833.

Yanez-Pagans, P., 2013. Cash for cooperation? Payments for Ecosystem Services and common property management in Mexico. CIDE International Seminar, October 2013. Mexico City, Mexico.