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Roldan Muradian
Laura Rival *Editors*

Governing the Provision of Ecosystem Services

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Chapter 1
Introduction: Governing the Provision of Ecosystem Services

Laura Rival and Roldan Muradian

Knowledge of ecological systems, biological diversity, and environmental degradation has progressed substantially over the last three decades and, with it, attempts to integrate ecology with economics. Economists interested in understanding the causes of environmental problems, as well as the economic value of the goods and services provided by ecological systems, have elaborated a range of innovative concepts and methods. Various schools of economic thought have sought to assist the decision-making process by addressing market failures and their negative impact on both the natural world and the welfare of societies. The call to value nature when making development decisions and to treat the world's ecosystems as capital assets in order to prevent their continued degradation and depletion is at the origin of current concern with 'greening' the economy (Panayotou 1993; Daily et al. 2000).

Despite inherent problems in measuring natural capital and assigning a monetary value to biological diversity and the services we may derive from it (market prices do not reflect the full social costs of production, nor do they reveal clearly societal values), 'green markets' have emerged and expanded in response to the ecological crisis. Although a utilitarian framing of ecosystem functions as providing numerous benefits, goods, and services to society is not new (Gómez-Baggethun et al. 2010), the growing consensus that conserving nature enhances human well-being (MA 2005), helps reduce poverty (Sachs et al. 2009), and promotes resilience in the face of climate change (Chapin et al. 2009) has led to new international initiatives such as The Economics of Ecosystems and Biodiversity report (Kumar 2010) and the

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creation of the IPBES (Intergovernmental Platform on Biodiversity and Ecosystem Services). As a result, the interest in market-based policy instruments such as PES (payments for ecosystem services) has spread very quickly, especially in regions rich in biodiversity (Pattanayak et al. 2010; Bateman et al. 2011). The growing popularity of PES has meant that 'what started as a humble metaphor to help us think about our relation to nature has become integral to how we are addressing the future of humanity' (Norgaard 2010: 1227). There is considerable debate as to whether PES amount to a particularly reductionist form of free market fundamentalism, and whether they are causing the unnecessary commodification of ecosystem services (Engel et al. 2008; Muradian et al. 2010; Farley and Costanza 2010; Gómez-Baggethun and Ruiz-Pérez 2011). The latter refers to the incorporation into a trading system of ecosystem services that hitherto were outside the market domain. Though in a matter of few years market-oriented tools have gained considerable leverage in the environmental policy agenda worldwide, market approaches are still far from being the dominant policy strategies for environmental protection and biodiversity conservation. In practice, environmental governance is implemented through a wide variety of models and instruments, and more often than expected, the management of natural resources depends on a combination of governmental command-and-control, market tools, and community-based institutional arrangements, as some of the cases studied in this book illustrate.

Now that the concept of ecosystems services (ES) has been introduced to address the fact that human activities affect earth's life support systems so profoundly as to threaten many of the biological functions of ecosystems, including those that are essential to human survival and key economic processes, its increased use in policy and decision-making reopens many of the value debates that have marked the recent evolution of economic theory (Kosoy and Corbera 2010; O'Neill 2007). More specifically, the ES paradigm has revived the debate about the relationship of industrialised societies with the natural world. It has also renewed the critique by a wide range of social scientists, social theorists, and philosophers of the theoretical and methodological contributions of environmental and ecological economics. Many authors now agree on the need to value nature according to a broad range of considerations and variables. Recent contributions to *Ecological Economics* and *Environmental Values*, to take just two examples, have vigorously debated the legitimacy of treating living things as exchangeable commodities or the validity of placing monetary values on parts of nature (Spash 2011). They have argued that the relationship of humankind with nature should not be reduced to narrow self-interest or cost-benefit calculations (Hourdequin 2010; Ojala and Lidskog 2011). There have also been discussions about the compatibility of ecological and economic rationality, the need to refine the definition of what counts as natural capital (and the extent to which this is an appropriate concept), or the value of ecological wealth (Dasgupta 2011).

While academic debate about the economic value of ecological wealth continues to unfold among economists and between economists and other social scientists, an increasing number of policy-makers, economic agents, and social and political actors have decided to include ecosystem resources and services in their decisions. These decisions have resulted in a multitude of trade-offs and economic transactions, including those documented in this book. Studying actually functioning 'markets' for environmental goods, resources, and services empirically is the best way to form a real understanding of their effectiveness in delivering environmental protection and a more equitable distribution of resources worldwide and between social groups at the local and national levels.

As the case studies collected in this book show, in practice, very different types of monetary transfers have emerged in response to the ecological crisis, including trading schemes for carbon credits, direct payments for compensating landholders for the adoption of more environmentally friendly practices, certification schemes, and contracts for potential future commercial use of biodiversity, among others. They also show how these monetary transfers are directly related to international commitments, particularly those of reducing emissions of carbon dioxide (and to a lesser extent other greenhouse gases) and of protecting and conserving the diversity of life, especially in tropical rainforest areas. Although carbon emission reductions can take two different forms (i.e. emission reduction and the production of additional carbon absorption capacity), emerging markets have mainly involved the latter under the guise of international carbon offset deals. The relationship between biodiversity conservation and markets has essentially involved the development of financial mechanisms to cover the costs of protecting nature and—to a lesser degree—address the social development needs of local communities. This approach has given rise to 'conservation concessions' based on the international willingness to pay for the conservation of valued ecosystems and aimed at compensating owners for the loss of alternative economic uses. Since 1990, there have also been agreements (e.g. bioprospecting contracts) for the use of knowledge from local groups living in developing countries in exchange for part of the revenues to be obtained by pharmaceutical or seed companies in case of successful patenting. Two additional market-oriented types of policy instrument have seen the light: negotiated agreements between downstream beneficiaries and upstream providers in watershed contexts and certified markets for biodiversity-friendly products. In forest areas, biodiversity, watershed, and carbon services are strongly linked, and countries such as Costa Rica have 'bundled' them together in a national PES programme.

The book contains examples of all these types of market-based policy instruments, as well as other policy tools for environmental governance. Seven contributions discuss Latin American cases, including Costa Rica (Chap. 12 by Le Coq et al.), Nicaragua (Chap. 18 by van Hecken et al.), Peru (Chap. 21 by Rojas and Berger), and Brazil (Chap. 16 by May and Vinha, Chap. 17 by Schmitt, Farley et al., Chap. 20 by Ribeiro et al., Chap. 19 by Andrade et al., and Chap. 2 by Börner and Vosti); three discuss African ones: Ghana (Chap. 22 by Insaidoo et al.), Ethiopia (Chap. 15 by Wiersum and Belay), and Madagascar (Chap. 11 by Bidaud et al.). There are five cases from Asia and the Pacific: India (Chap. 7 by Ananda), Java (Chap. 6 by Lukas), Japan (Chap. 10 by Yashiro et al.), Australia (Chap. 9 by Concu and Chap. 14 by Concu and May), and the Philippines (Chap. 8 by Toribio et al.). Finally, one case discusses France and its overseas territories (Chap. 13 by Maury et al.). These studies provide rich empirical data on the unique problems posed by the incorporation of ecosystems as natural capital (i.e. supplier of services) in economic decisions in both the developing and the developed world. They illustrate some of the dilemmas and conflicts involved in making the values of nature an integral part of collective choice and decision-making. We show that actors attempting to maximise the benefits derived from ecosystem goods and services adopt social constraints on production possibilities, whether these are self-imposed or imposed by others. The book thus illustrates the complexity and the cost of creating economic incentives for environmental improvement and poverty alleviation.

Although the market logic is simple (an economic agent deriving monetary benefits from the provision of ecosystem services will incorporate such services into her land-use decisions), calculations intended to bring ecosystem functions and ecological wealth into development decisions are marred with difficulties. As our contributors demonstrate, these difficulties are due to the fact that institutions and social values condition how monetary transfers and other policy tools work in practice. Markets and policies are embedded in structures of property rights, social relations, and cultural frameworks. Institutions, information flow, and cultural features thus play a critical role in conditioning the ways in which they operate. They also determine to a large extent how costs and benefits are allocated among different social groups. A central aim of the book, therefore, is to discuss the ways in which local institutions and cultural traits affect the performance of different combinations of policy instruments (particularly market-based ones) for enhancing the supply of ecosystem goods and services. As we argue below, this requires that we assess the role the state plays, or could play, in governing the provision of ecosystem services.

1.1 Rethinking Environmental Governance

The contemporary conception of environmental governance is closely related to the emerging scientific understanding of society and nature as forming complex and dynamic interrelations known as ‘social and ecological systems’ (Chapin et al. 2009), which in turn form the ‘human-earth system’ (Chapin et al. 2011). The concept of ecosystem service thus signals ‘fundamental changes in society’s approach to the environment’ (Nicholson et al. 2009: 1143), which require the study of (a) financial organisations and their role in governance at various scales (either as facilitators of new institutional arrangements or as negative forces); (b) the state as the central locus of regulation and enforcement at multiple levels; (c) the interplay between governance, scales, and institutions; and (d) new market-based instruments for managing the provision of ecosystem services.

The case studies presented in the book shed new light on the institutionalisation of mechanisms for collective decision-making and collective action with respect to natural resource management. Their comparative analysis highlights the central importance of the formal and informal ways in which the provisioning of ecosystem goods and services is organised and managed. Each chapter refers to aspects of what has come to be known as ‘environmental governance’. Like other scientific terms widely circulated within policy circles, the concept of 'environmental governance' is ambiguous and open to multiple interpretations. This term, however, is often used to evoke a decision-making process by which environmental sustainability and the common good get decided not only by governments but also by a wide range of local, national, and transnational actors operating both 'below' and 'above' the state. As Lemos and Agrawal (2006) note, the paradigm of environmental governance seeks to expand cooperation among non-state actors that may have been previously outside the policy process, such as corporate interests, social movements, and non-governmental organisations. The governance mechanisms reached by loose networks of institutions and actors, or, in Lemos and Agrawal's terminology 'multilevel, non-hierarchical, and information-rich coalitions', are thought to be more effective than state-centric control and regulation (see also Holling and Meffe 1996).

Panayotou (1993) was one of the first authors to argue systematically that states on their own are not the appropriate agents of environmental decision-making, and that traditional governmental policy-making should leave much more room to self-organisation. He argued that government policies, rather than correcting failures in markets for natural resources, tend to add distortions whether through taxes, subsidies, quotas, regulations, inefficient state enterprises, or public projects with low economic return and high environmental impacts (Panayotou 1993: 58–59). He added that 'the role of the state in the struggle for sustainable development is critical and fundamental but it is not one of direct management or command and control. The state's role is rather to establish new rules of the game and create an environment that fosters competition, efficiency and conservation' (Panayotou 1993: 144). He therefore called for policy reforms which would ensure that the state would remove the distortions that it had introduced in the first place. The role of the state, as he saw it, should be of creating market conditions for environmental resources and services, which, by not being brought within the present configuration of markets, were being undervalued and depleted. This vision depicting private or community arrangements as more efficient, in comparison to the state, has been increasingly challenged, particularly in the case of the management of common-pool resources. Empirical evidence suggests that natural resources are not best governed either by private owners, whose property rights facilitate efficient market regulation of environmental issues, or by the state, on behalf of the people. Rather, both governance structures can be either effective or ineffective, depending on the rules they rely upon and on how these are enforced (Ostrom and Basurto 2011). The public-private dichotomy is overly simplistic (Sikor 2007; Ostrom 2010).

The chapters composing this book corroborate the view that ecosystem service governance defies conventional dichotomies between state and market, public and private, or regulation and incentive. As our contributors show, new modes of environmental governance need to address the fact that while regulating and supporting ecosystem services are public goods (Farley and Costanza 2010), many provisioning and cultural ones are best understood as common-pool resources (Ostrom and Cox 2010; Brondizio et al. 2009). The insights gained in institutional economics during the past three decades suggest that neither hierarchies nor markets can be considered a priori better policy approaches to regulate the provision of such types of goods.

Our book contributes to this emerging understanding of environmental governance by highlighting the hybrid, multilevel, and cross-sectoral nature of decision-making and collective action that together redefine the social boundaries of markets. In this book, we show that environmental governance comprises a wide set of nested regulatory processes, from international governance mechanisms to the very local level, where ecosystems are appropriated by human societies. Although some environmental governance modes emerged during the neoliberal era as a non-state approach, controlling environmental degradation is no longer thought to be a choice between either political agreement resulting in government taxation and regulation or economic forces acting freely through market exchange. The governance of ecosystem service provision requires therefore that we draw attention to the different layers, scales, and dimensions nested through the generation and flow of such services.

1.2 PES and Other Emerging Policy Tools for Environmental Governance

The book is composed of four main parts, which highlight the institutional settings and the normative basis of ecosystem services on the one hand, and the complex and dynamic sociopolitical interactions between private and public stakeholders through which ecosystem services are supplied, on the other. The first part, 'Keywords and Concepts', offers a critical analysis of central tenets of the ES paradigm. The second part, 'Construction and Evolution of Governance Regimes', traces the conceptual development of the ES paradigm from a historical and institutional perspective. The third part, 'The social embedding of PES', includes a range of cases analysing governance schemes making use of payments for managing the provision of ES. The fourth part, 'The Special Case of Carbon Markets', contains four chapters dedicated to one particular type of ES, carbon sequestration, as well as the concluding chapter. Given the current 'carbon obsession' of the environmental policy agenda, it is necessary to assess critically the extent to which the emerging global governance regime for reducing emissions from forest destruction can actually save the threatened and very valuable—often due to reasons far beyond their carbon content—tropical forests.

1.2.1 Critical Analyses and General Overviews

Researchers studying the policy process have often remarked that analytical categories inevitably acquire normative connotations with the circulation of scientific concepts and ideas, and their appropriation by actors implicated in the formation of policy discourses (e.g. Gasper and Apthorpe 1996). A change of terminology often signals a change in the way problems are perceived and addressed and questions posed. Terms such as 'ES' and 'PES' (Chap. 4 by Pesche et al.), 'partnerships' (Chap. 3

by Kramarz), 'incentive' (Chap. 2 by Börner and Vosti), or 'bioprospecting' (Chap. 5 by Stromberg et al.) are all social constructions, which can be used to describe, analyse, conceptualise, or prescribe. Although they are intended to facilitate the making of decisions or the taking of actions, they can also be used as rhetorical tools. The four chapters in Part I present critical analyses and general overviews of each of these terms, which are central to the new environmental governance paradigm.

Börner and Vosti's contribution focuses on the many ways 'incentive' and 'disincentive' are being thought and deployed. Although written as a comprehensive survey of environmental management policies in the Amazon basin, this chapter provides a wide-ranging discussion of policy instruments available to all decision-makers aiming to avoid trade-offs between ecosystem conservation and human welfare through policy integration. The insights Börner and Vosti offer on motivation and behaviour are echoed in many of the book's case studies.

Kramarz deconstructs the notion of 'public-private partnership', which has become a central tenet of the World Bank. She shows how World Bank documents and activities frame co-management (whether collaboration between state agencies and communities, public-private partnerships between market actors and state agencies, or social-private partnerships between market actors and communities) as a necessary innovation to address the complexity of environmental problems and the democratic deficit in global governance. Partnerships are promoted as an efficient way of producing regulatory effects through individualised incentives and other market-based instruments, which, it is hoped, will catalyse the willing participation of a diverse range of actors. As they have become a normative imperative in global environmental governance, we need to ask why their emergence, which amounts to a change in procedural norms, does not correlate with the desired changes in substantive norms. This question is subsequently answered by a number of the book's empirically informed chapters.

Pesche and his co-authors take a historical approach to the gradual incorporation of ecosystem services into payment schemes and show the seminal role played by the Millennium Ecosystem Assessment in orchestrating the 'mutual justification' of ES and PES. They argue that the desire to raise public interest in biological diversity coupled with the imperative to secure funding to protect the natural environment has led to the parallel emergence of two new scientific fields, the science of ecosystem functions and the economics of conservation. While the term PES suggests the existence of well-defined and valued services and of market-based payments that accurately reflect the value of these services, in practice, these payments are really aimed at conservation activities. At the border between knowledge and intervention, the ES/PES paradigm becomes a multi-goal public policy instrument subject to power relations and social embeddedness. Several chapters discussing PES historically in various national contexts (e.g. Bidaud et al. Chap. 11, Le Coq et al. Chap. 12, or Lukas, Chap. 6) further support the contention that the PES concept and its power and generative capacity are best approached from a diachronic perspective.

Stromberg and his co-authors look at bioprospecting through the remits of the Convention on Biological Diversity (CBD), which intends to create strong conservation incentives for both biodiversity holders and external users. Their approach is both historical and comparative. Their historical reconstruction of the CBD from the 1993 Nagoya protocol to the present (i.e. October 2010), and their comparative examination of 67 cases on three different continents (selected out of a data base comprising 190 case studies) lead them to argue that transaction costs due to contractual uncertainty have deeply influenced the modes of governance of bioprospecting. Their conclusion on access and benefit sharing of genetic resources in bioprospecting projects linking southern providers and northern buyers has wider implications for PES more generally.

1.2.2 Deconstructing PES

The book builds on the insights of Muradian et al. (2010) regarding the nature of PES. As van Hecken and his colleagues (Chap. 18) explain, the 'Coasean' PES approach fails to account for the complex interactions between PES and the broader institutional context. It is therefore more appropriate to define PES as transfers of resources between social actors, which aim at creating incentives that align both individual and collective land-use decisions with broader conservation values and societal goals. It is worth noting that very similar debates about efficiency, equity, and governance structures exist regarding direct cash transfers to the poor, in particular with regard to the nature of incentives promoted by such payments. Discussions of the long-term effects of conditional cash transfers, the social costs associated with economic growth and market imperfections, and the role of the state in economy and society, especially when the need for more demanding institutional reforms is felt (Bastagli 2009; Barrientos et al. 2008), are all very relevant to the debate about PES.

As nature is increasingly being redefined in terms of the benefits humans derive from ecosystem functions, ES provisioning and ES valuation have become inseparable issues (Chap. 4 by Pesche et al. in this volume). Most of the book's chapters focus on the social relations through which ES are traded and used, rather than on those through which they are 'produced'. However, given the scientific uncertainty regarding the nature of ES goods and services, and their specific relationship to human welfare (Raudsepp-Hearne et al. 2010), it is important to mention the debates that are shaping the ways in which science co-evolves in society (Norgaard 2010: 1225). The differing conceptualisations of Mace et al. (2011) and Luck et al. (2009) on the roles played by biodiversity in ES processes and services, for example, are indicative of the difficulties scientists encounter when trying to determine the value of biodiversity. Biodiversity could be valued as a regulator of fundamental ES processes, a final ES itself, or a good. Although many policy advisers would underplay such valuation problems on the ground that, in practice, all what we need is an agreement on the need to maintain ES and an estimate of the cost of ES provision, it must be said that the choice of ecological framework to understand ecosystem functioning and ES provisioning and, consequently, the type of valuation and payment largely depends on how biodiversity is valued. As forcefully argued by Norgaard (2010), the ES perspective emerging from the Millennium Ecosystem Assessment is too narrowly framed within a stock-and-flow view of ecology, which fits the reductionist approaches favoured by dominant market thinking. Norgaard reminds us that ecological science relies on multiple patterns of reasoning, and that we need all of them to inform governance more fully.

Ecosystem goods and services are often defined in a compartmentalised way. Maury and collaborators (Chap. 13), for instance, differentiate ecosystem services as services provided by ecosystems to society from environmental services, which are produced by actors (see also Fisher et al. 2009). However, efforts to formalise transaction agreements lead to unsolved issues of classification and categorisation, such as those discussed by Stromberg and co-authors in the case of genetic resources (Chap. 5). Börner and Vosti (Chap. 2) similarly remark that if the ecological assessment's classification of ecosystem services into regulating, provisioning, and supporting services is useful to identify the amounts and pathways through which ES benefits flow to specific stakeholder groups, it is problematic from a management point of view, as it groups together ES with very different characteristics. They note, for instance, that managing flood versus climate regulating ES requires entirely different sets of knowledge and policy instruments. Chaytor (2002) similarly argues that, although environmental goods and services represent one of the fastest growing economic sectors, there is no clear-cut difference between 'good' and 'service', or between those that are 'ecosystem', rather than 'environmental' goods and services. As a result, definitions vary widely from country to country and from policy document to policy document.

Some of these issues of definition, categorisation, and relationship between biodiversity and traded ecosystem goods and services are taken up in two of the chapters on Brazil, those by May and Vinha (Chap. 16), and Schmitt and co-authors (Chap. 17). May and Vinha discuss the highly innovative National Plan for the Promotion of Socio-Environmental Chains, which aims to insert agro-extractivism within a 'solidarist economy' framework by guaranteeing minimum prices for the certified forest products of social and community enterprises involving low income groups. These communities depend on the stability of ecosystems that shelter components of Brazilian biodiversity for their livelihood. Schmitt and co-authors explain that if biodiversity is not an ecosystem service itself, it plays an essential role in sustaining *all* ecosystem services. Although all ES result from complex geo-biophysical interactions, as Börner and Vosti (Chap. 2) remind us, not all ES are equally 'systemic'. Furthermore, because ecosystems exhibit highly complex, dynamic, and nonlinear behaviour, including the presence of abrupt, irreversible thresholds, excessive conversion of forest to conventional farmland leads to the irreversible loss of essential services. Techniques that combine food production and biodiversity conservation such as agroecology should therefore be encouraged on a large scale. In the Atlantic forest where Schmitt and his colleagues (Chap. 17) carried out field research, as in many other rural parts of the world, the best way to prevent shortages of water, energy, food, or natural resources is by managing ecosystems for services, which requires large upfront public investments. They conclude that biological functioning cannot be protected through market mechanisms, as these fail to reward resource owners for the benefits of conservation. While some services are amenable to market institutions, others require public provision.

In a similar way, Börner and Vosti (Chap. 2) point to government involvement in establishing and articulating demand for ES. Concu (Chap. 9), in a very useful discussion of the actual—as opposed to rhetorical—differences between PES programmes and command-and-control approaches to environmental conservation, contrasts policies that encourage a change in output composition from those that seek to affect the revenue structure or production cost or volume. She also discusses the blurred distinction between PES and subsidies.

The chapter by Schmitt and co-authors (Chap. 17) can also be read as an effort to disambiguate the concept of PES. They stress that there are two general approaches to PES, one based on trying to force ecosystem services into the market model with the goal of increasing economic efficiency, and the other based on adapting economic instruments to the specific characteristics of ecosystem services in order to achieve a variety of goals, such as sustainability, justice, and efficiency, adding that only a minority of ES fit the market model. Schmitt and colleagues remark that it is because they do not take into account the fact that dealing with non-rival or non-excludable resources is inherently more difficult that authors such as Engel or Wunder view private sector PES as more effective and more efficient than public sector ones. They conclude with a proposal to redesign PES 'as a form of public sector venture capital, in which wealthy countries and national governments that benefit from the ecosystem services agroecology provides transfer resources to less wealthy countries and local governments otherwise unable to fully finance the necessary public sector investments'.

The chapter by Maury and co-authors (Chap. 13), which reframes subsidies paid by the French government to farmers as a kind of PES scheme, offers a fascinating complement to the chapters by Concu (Chap. 9) and by Schmitt and co-authors (Chap. 17). They too argue that the broader understanding of PES offered by Muradian et al. (2010) is needed to describe and analyse agri-environmental contracts. This position is echoed in other chapters, particularly those by van Hecken et al. (Chap. 18), Yashiro et al. (Chap. 10), Ananda (Chap. 7), and Wiersum and Belay (Chap. 15). The chapter by Maury and co-authors (Chap. 13) illustrates two points made by Börner and Vosti (Chap. 2) and echoed throughout the book. First, that we urgently need to develop 'methods to measure the benefits (though not always necessarily the monetary value) associated with particular ES or bundles of ES'. Second, that 'humans simultaneously adapt to and change ES provision through activities that alter natural, temporal, and spatial dynamics'. As Maury et al. show so well in the context of French agriculture, one way of adapting and changing is by arguing about what motivates us, humans, to act.

1.2.3 Rethinking the Role of the State in Governance Structures

As repeatedly argued by Elinor Ostrom, if natural resource systems are governed by complex local and national institutional arrangements, commons institutions evolve with the expansion of spatial and temporal scales. Therefore, our common challenge in the twenty-first century is to agree on a supportive legal structure at macro-levels that would facilitate the self-organising capacity of local groups and communities, who would be free to craft their own rules (e.g. Ostrom 2001). Common-pool resource thinkers have stressed the importance of self-organisation and the need to design a supranational level of governance consistent with the eight design principles they have identified. This has led them to focus on ensuring that the supranational level complements, rather than replaces, the essential national, regional, and local institutions. As a result, they have tended to neglect the role of the state in governance arrangements. Equally, and as argued by Eckersley (2004), there has been an unfortunate tendency among green scholars and environmentalists to characterise the sovereign state as ineffectual at best and ecologically destructive at worst. However, there is a need to rethink the state in light of the principles of ecological democracy, as a facilitator of transboundary democracy and a steward of the Earth System (see also Backstrand et al. 2010). Several of the book chapters acknowledge and discuss the new roles of governments in multi-centric governance structures. They envision the state as a core political institution capable of facilitating socially progressive environmental change and true sustainability and discuss initiatives and innovative paradigms of regulation that aim to tame the environmentally destructive potential of the state, while enhancing its emancipatory potential.

Re-engaging the state in structures of environmental governance advances the policy debate on how to combine incentive and control beyond the recognised shift in natural resource management from the polluter pays principle to the beneficiary pays approach. Many of the environmental governance structures discussed in the book make use of both regulation and incentive in hybrid systems or 'policy mixes'. Together, they illustrate the need for command and control to overcome the legal and institutional barriers that prevent the good functioning of incentives. States and intergovernmental agreements are needed to provide the necessary underpinnings for markets to work. Today, neither market actors nor non-governmental organisations (or public-private partnerships for that matter) have the political power to set up or regulate the evolving carbon market structures (Lederer 2010), for example. The four chapters in the last section of the book on carbon markets exemplify the coordination role of central governments, across both regions and sectors of the national economy. They show that governments are essential coordinating/integrating mechanisms, which help create functional interdependencies and strategic alliances.

As Börner and Vosti note, incentives often need to be part of a wider policy mix involving various measures, including actions to enable local economic development, given that 'trade-off relations between ES objectives and other development objectives are the rule rather than the exception'. May and Vinha stress another important role for governments beyond creating the conditions for partnerships and other private initiatives to be successful. If only the Brazilian government would implement green procurement policies, the market share of sustainably produced products would increase automatically. The three other chapters on Brazil (Schmitt et al. Chap. 17, Andrade et al. Chap. 19, and Ribeiro et al. Chap. 20) mention the enforcement of environmental laws through the supervision and monitoring of municipal, state, and federal agencies as a key issue. They also discuss the creation of conservation areas by either the federal state or sub-national levels of government, which fulfil the government's responsibility to ensure that the forests under its custody are protected or used sustainably. In regions where the gap between law and practice is wide and where regulatory policies are not implemented, enabling policies are not sufficient to protect ecosystems or biodiversity. In other words, these authors show that there is still a place for the classic role of governments, whose exclusive responsibility regarding land-use planning and law enforcement is crucial in the fight against biodiversity loss and environmental degradation.

Four additional aspects of the government's key role in environmental governance are underlined in various contributions to the book: its capacity to absorb and domesticate exogenous policies, its role in channelling investments, its responsibility in setting policy priorities, and, finally, its custodian obligations towards local knowledge, values, and institutions. Concu (Chap. 9), Concu and May (Chap. 14), Yashiro et al. (Chap. 10), Andrade et al. (Chap. 19), Ribeiro et al. (Chap. 20), and Rojas and Berger (Chap. 21), all mention the responsibilities of national governments as signatories to international treaties fostering the conservation of biodiversity and the protection of tropical rainforests. Stromberg et al. (Chap. 5) mention the key role of states in relation to the CBD, particularly in relation to sovereignty issues. Bidaud et al. (Chap. 11) discuss these international treaties in the context of dependency and postcolonial state building. Although in Madagascar environmental policy was initially imposed by donors, PES got gradually integrated within domestic agenda. Le Coq et al. (Chap. 12) similarly show that PES were pushed on Costa Rica by donors, but this did not prevent local appropriation over time. Moreover, payments to landowners would not have been possible without the state, which finances the scheme through a range of taxes. Maury et al. (Chap. 13) mention that if initially the highly centralised French state had to adjust to European policies favouring the neoliberal preference for greater use of self-regulatory markets and less public intervention, PES nevertheless became a mix policy tool that evolved not so much out of the pressure exercised by the Europe Union on France but, rather, from tensions between various ministries, which took different positions vis-à-vis European directives.

Schmitt et al. (Chap. 17), Toribio et al. (Chap. 8), Andrade et al. (Chap. 19), Ribeiro et al. (Chap. 20), Insaidoo et al. (Chap. 22), Ananda (Chap. 7), and Yashiro et al. (Chap. 10) show that decentralisation does not necessarily mean a lesser role for the national government, which retains the responsibility of assigning governance functions across scales. And where new levels of governance have been artificially inserted in compliance with donor demand or expectation, it often falls to the central government to readjust governance structures to improve efficiency and fairness. In addition, they show that implementing enabling management structures requires long-term coordination and the establishment and maintenance of legal regulating frameworks that require coordinated fund raising, as well as the ability to cover substantial upfront costs, all activities that are best undertaken by central governments. This is well illustrated by Ananda (Chap. 7), who discusses the problems of vertical control and horizontal coordination across different branches in watershed management in India. He concludes that national governments have an important role to play in determining how to achieve optimal delegation.

Several authors mention that resolving conflicts and deciding on trade-offs between development and conservation require the active involvement of the state. Andrade et al. (Chap. 19) mention the importance of conciliation and the creation of public arenas in which conflicts can be aired. Controversial laws need to be discussed in public hearings if they are to win politically where powerful actors remain unconvinced of their benefits. Le Coq et al. (Chap. 12) explain how Costa Rica's national PES programme gradually evolved, as the balance of power and resources between forestry stakeholders and environmentalists changed over time.

1.2.4 Land-Use Change and State Protection of Place-Based Knowledge

The ways in which the state can protect and promote place-based knowledge, rather than undermine it, are powerfully discussed by Concu and May (Chap. 14) and to a lesser extent by Wiersum and Belay (Chap. 15), Lukas (Chap. 6), May and Vinha (Chap. 16), Schmitt et al. (Chap. 17), and Maury et al. (Chap. 13), Concu and May's chapter focuses on indigenous protected areas (IPAs) in Australia. They analyse IPAs as resulting from international institutions and frameworks and their selective adoption by both the Australian federal government and indigenous people in pursuit of their own environmental, cultural, and economic interests. As a result, 'by incorporating and integrating non-indigenous institutional elements within indigenous land ownership, culture, and management systems', IPAs have come to occupy 'a unique intercultural space' in the Australian nation-state. May and Concu show how IPAs have been shaped by unequal relations of power between very different kinds of actors. These conservation spaces are defined according to non-indigenous concepts, principles, and practices, such as, for instance, the legal separation of land ownership and rights over marine resources. More significantly, IPAs are the products of convergence, as well as of tensions, between indigenous and non-indigenous values, interests, and knowledge. May and Concu argue that the state has a key role to play in ensuring that convergence overcomes tensions. This is demonstrated, for instance, in the 2008 Australian High Court ruling which extended indigenous rights over intertidal zones in the Northern territory, and, which, by doing so, acknowledged the validity of indigenous conceptualisations and meanings of space.

Wiersum and Belay's fine discussion of forest beekeeping in southwest Ethiopia illustrates the fit between traditional beekeeping and biodiversity conservation. Trees are actively protected from pests and fire, while beekeeping favours pollination, which, in turn, improves the regeneration of rare tree species. The *kobo* system, like the Aboriginal Australian estate formally recognised as IPA, has been incorporated within more formal forest governance arrangements. These may strengthen the traditional management system, or, instead, weaken it, depending on how successful the government is in curbing commercial interests and priorities. Wiersum and Belay (Chap. 15) describe four different types of tenure arrangements for hive hanging trees and show their flexible application to a wide and diverse range of local specificities.

What makes the *kobo* system so amenable to modern conditions is that it includes a transferable tree tenure system. Conflict resolution mechanisms to deal with disputes over honey colonies, honey trees, or forests where beekeeping is practised facilitate the adaptation of traditional beekeeping to modern conditions. With the gradual integration of beekeeping forests within coffee plantations, however, the *kobo* system weakens, even if government regulations and the official promotion of agroforestry protect it to some extent from commercial agriculture. The chapter shows very well how the tension between sustainable and unsustainable farming practices has resulted in formal forest governance arrangements gradually supplementing the traditional system, until they start competing with it, leading to the gradual erosion of *kobo* rights. Wiersum and Belay (Chap. 15) end their chapter by mentioning what the government could do to ensure that formal forest governance arrangements and the *kobo* system mutually reinforce each other.

Schmitt et al. (Chap. 17) describe similar attempts to create a regulatory regime to support the use of traditional techniques combining agriculture and nature conservation. Interestingly, in this case, traditional techniques had to be introduced (or reintroduced) to help local farmers develop agroforestry and agroecology food production systems. May and Vinha, who discuss the setting up of new commercial chains for non-timber forest products that generate sufficient revenues for producers without undermining forest conservation, explain how forest dwellers involved in these chains have come to be recognised by the government as culturally traditional rural dwellers. Yashiro et al. show the appropriateness of the traditional Japanese landscape management system *satoyama* for the design of a modern governance system in which the state could play an active role. Both Maury et al. (Chap. 13) and Lukas (Chap. 6) discuss conflicts between small farmers and government authorities over the best way of combining biodiversity conservation and agricultural production. What makes the French case discussed by Maury et al. so interesting is that the state could, through better ministerial coordination, reframe subsidies to grow food as payments for ecosystem services. This would facilitate the recovery by French farmers of their forebears' traditions, which, in turn, would help them think about the land they work in new ways. A growing number of young farmers already understand farming as place-based knowledge. Like in the Japanese case, the state would then play a key role in enabling the materialisation of a *satoyama* holistic vision of the rural landscape, organised according to the intersecting spatial and temporal spans of ecological processes.

1.3 The Challenges of Multilayered Governance

Together, the book's empirical chapters show that PES are best understood as influential governance tools actively promoted by both international agencies and national governments. They contribute to renew the discussion on how to reshape administrative boundaries and political regions in a way that allows for the provisioning of ES. We argue that the state has an important role to play in reconciling ecology, economy, and the social and cultural processes of local inhabitants, while resolving conflicts generated by overlapping jurisdictions and competing land management agencies. As Concu and May (Chap. 14) argue, environmental management that relies solely on political or administrative boundaries is unlikely to be effective for conservation landscape. Moreover, as shown by Schmitt et al. (Chap. 17) the emerging multilevel approach based on the vertical and horizontal integration of institutions and actors, and on local traditions and knowledge systems, requires that we think about ES in terms of public or common-pool goods.

One of the main insights emerging from our collection is that successful institutional innovations have treated the state as an important actor in the holistic management of social-ecological systems. Multilevel governance systems entail a complex architecture involving a multiplicity of actors and many interrelations between the 'local' and the 'global'. The resulting problems of regulation and enforcement at different levels have been even more challenging than in the past. This challenge requires that we move from thinking in terms of single, ideal managerial approaches (e.g. command-and-control, markets, or community-based management) to combining governance structures, scales, and tools. Management decisions regarding public goods (and most ecosystem services are public or common-pool goods) require that higher-level institutions and organisations be recognised as having other purposes and functions than just establishing the rules within which decision-making processes operate or simply defining the metarules for local resource users (Eckersley 2004). Nonetheless, without appropriate incentives or local engagement in rule making, there is abundant evidence that state policies might be ineffective. As McGinnis (2000) has argued, governance does not require a single centre of power, and governments should not claim an exclusive responsibility for resolving political issues. If politically the goal is to establish and sustain the capacities for self-governance, that is, the structured ways by which communities organise themselves to solve collective problems, achieve common aspirations, and resolve conflicts, then it may be time to move from thinking in terms of governing the commons to thinking in terms of greening the state.

The recent rise in the policy agenda of market-based mechanisms for environmental governance has shifted the emphasis from getting the right governmental regulation for conservation to getting the right price for ecosystem services. Our book, however, calls for moving away from this false dichotomy and to pay attention to getting the right set of rules and instruments, along multiple governance layers. Nested (polycentric) institutions have had a role to play in all the complex environmental governance systems discussed in this book, and central governments have been shown to be increasingly called upon to engage with other social actors to ensure the provision of ecosystem services.

Clearly, a number of issues are in need of further elaboration, and we end by mentioning two, which we develop in greater depth in the concluding chapter. First, ES governance, especially through PES, has proven far more difficult than anticipated. Second, there is yet insufficient conceptual and empirical clarity about what set of institutions are the most appropriate for the governance of ecosystem services.

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Part I
Keywords and Concepts

Chapter 2
Managing Tropical Forest Ecosystem Services: An Overview of Options

Jan Börner and Stephen A. Vosti

2.1 Introduction

Decision-makers have three types of choices to make regarding the management of natural resources and related ecosystem services (ES): (1) which ES to manage, (2) the quantitative and qualitative objectives associated with each ES (3) and the instruments for achieving these objectives. This chapter focuses on the final choice faced by decision-makers, that of selecting the proper instrument[1] from an array of options that includes standard environmental policies (e.g. land use regulations, taxes and subsidies) that are generally implemented by the State, as well as a broader set of alternatives (e.g. reallocation of property rights and joint management of common property resources) that can be implemented by stakeholders with or without involvement of the State. Choosing among alternative management instruments is difficult, first because information regarding their effectiveness and implementation costs is often missing or incomplete and, second, because managing ES often involves trade-offs with other policy objectives, such as economic growth, poverty alleviation or social equity (Cole and Grossman 2002; DeFries and Rosenzweig 2010; Lee and Barrett 2001).

[1] Throughout this chapter, we use the singular (instrument) with the understanding that sets of policy instruments (plural) may have to be deployed to achieve desired objectives; the effects and costs of these policy sets must be considered jointly.

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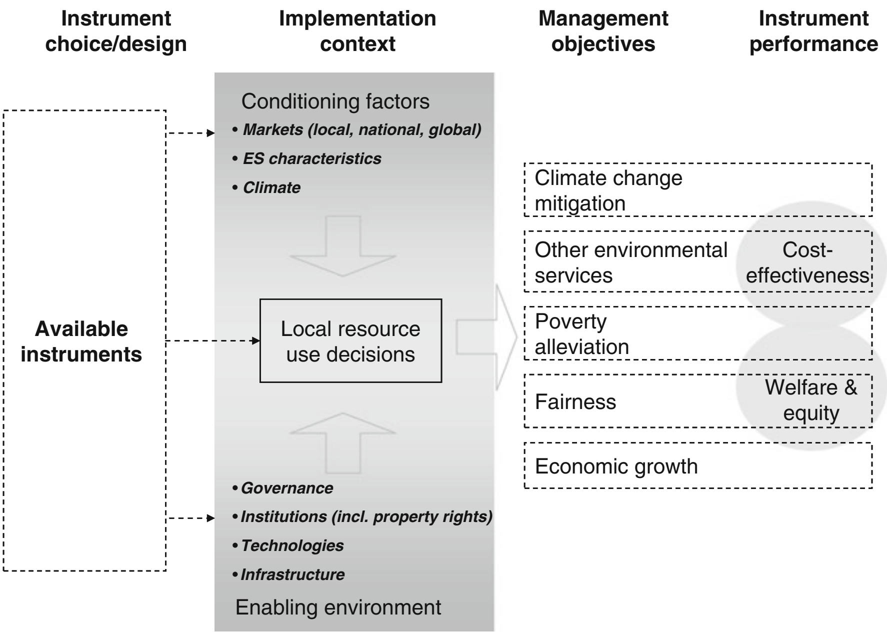

Fig. 2.1 A stylised impact pathway for ES management

The rationale for public policy attention (and perhaps action) in the context of ES with public good character is clear; environmental externalities, missing markets, information asymmetries, etc., suggest that without public sector interventions ES will be undervalued, overused and suffer from suboptimal levels of investment in many cases (Belli et al. 2001). Even ES that provide private benefits are sometimes underused or overused, for example, due to ‘conservation investment poverty’ (Vosti and Reardon 1997); these cases also merit public policy attention. However, since management is *never* costless, the existence of a market failure in the provision of ES is not sufficient to justify policy action. Decision-makers must understand the value or cost to ES beneficiaries of the effects of such a market failure and assess the effectiveness and costs of alternative options for addressing the problem. In the case of water pollution, for example, stakeholders may decide to invest in water treatment plants that substitute for natural water-purifying ecosystem services, especially if expanding natural purification systems displaces income and employment generating activities in upstream areas.

Therefore, taking any action at all requires that at least one instrument exists for which the benefits to society outweigh costs of implementing it over a defined time period; if this is not the case, then the socially optimal response is to take no action unless something changes this basic relationship. When several alternative instruments exist that pass this first fundamental test, attention is then focused on the relative cost-effectiveness of alternative management instruments. How a given policy instrument performs vis-à-vis alternatives depends crucially on the implementation context and design. To illustrate this, consider the conceptual framework in Fig. 2.1.

ES management fundamentally seeks to change natural resource use decisions in favour of a specific or collection of ES, for example, maintaining the climate-regulating function of tropical forests (i.e. reducing emissions from deforestation and degradation – REDD) to mitigate global climate change. Often environmental policies also have additional objectives, such as poverty alleviation and sustainable economic growth. Basically, three entry points exist to affect natural resource use decisions (Fig. 2.1). ES management can (1) change the rules of the game by affecting the conditioning factors of natural resource use, for example, market prices through certification of commodities produced with reduced impact on forest biomass (Veríssimo et al. 2005). Alternatively, decision-makers may (2) choose to influence natural resource use decisions directly, for example, through payments for ecosystem services (PES) (Ferraro and Kiss 2002), or (3) improve the enabling environment for sustainable local resource management, for example, through land tenure reform (Pacheco 2009).

Since local resource use decisions are influenced by conditioning (e.g. climate, natural resources and ES characteristics and markets) and enabling (e.g. institutions, infrastructure and technology) factors, the outcomes and performance (cost-effectiveness) of a given ES management approach also depend on these factors. For example, devolution of land rights to smallholders intended to improve community-based forest management may not be effective in reducing deforestation, where governance is weak and property rights are poorly enforced. Likewise, PES may exacerbate preexisting inequalities if land and pressure on forests is concentrated among only a few large landholders (Börner et al. 2010).

This chapter reviews the literature on ES management options in forested areas with two goals in mind.[2] Our primary objective is to provide an overview of instruments that have been used or proposed for managing tropical forest ecosystems and to assess their likely performance. Though we draw mostly on examples from the Amazon and Andes regions, many of the observations and conclusions are valid beyond this regional context. A second objective is to identify research needs. This chapter is organised as follows. The next section sets out a typology of management instruments. Section 2.3 identifies factors that influence the effectiveness of these instruments. Section 2.4 provides an assessment of the expected performance of specific ES management instruments. Section 2.5 concludes this chapter by providing implications for research, capacity strengthening and policy.

[2] We systematically screened over 600 peer-reviewed journal articles, research reports and institutional publications that dealt with the options for and the effects of environmental management. For each policy instrument category, key studies were analysed in more detail. Most publications deal with carbon, plant biodiversity and water-related ES; there were fewer studies of forest products, soil degradation and air pollution; few publications address specific and well-defined ES. We attribute this to the fact that the ES concept has only recently been widely adopted in the scientific literature, and that, with the exception of water, few ES-specific policy instruments are available. The Millennium Ecosystem Assessment (MEA 2005) provides one of the first, broad frameworks for defining and managing ES. A complete list of the reviewed literature can be obtained from the authors.

2.2 An Intuitive Typology of Management Instruments

Instruments to manage natural resource use and, hence, ES flows have been classified in many different ways. For example, Bayon (2001) distinguishes public-good-specific, incentive-changing and business options. Sterner (2003) divides the management toolbox into options to use markets, create markets, regulate use and engage the public. The MEA (2005) establishes categories of response options, such as legal, economic and social responses, among others.

Despite differences in the details associated with individual instruments examined in the literature, it is a common feature of all ES management instruments that they seek to influence human behaviour. Ideally, a decision-maker's goal is to 'adjust' human behaviour such that natural resources (and related ES) are used (or conserved) in socially optimal ways (Baumol and Oates 1988). In practice, decision-makers seldom know (or agree on) what this social optimum is and, even if they did, may not know how to achieve it. However, while we may not know what 'optimal' is, stakeholders in society do have strong preferences regarding natural resource management and ES, and we know that market forces (alone) will not deliver what most stakeholders prefer. Informed intervention thus requires an intuitive framework for choosing among multiple potential intervention options. If we classify management instruments according to how they attempt to influence human behaviour,[3] three basic (and admittedly not strictly separable) mechanisms can be distinguished:

1. Establishment of general conditions that *enable behaviour* driven by private incentives to contribute to achieving a given ES objective (Enabling)
2. Provision of (specific) *incentives that change behaviour* in ways that contribute to achieving a given ES objective (Incentives)
3. Provision of (specific) *disincentives that change behaviour* in ways that contribute to achieving a given ES objective (Disincentives)

2.2.1 Enabling Measures

Management options in the 'Enabling' category contribute to establishing conditions that lead to the management of ES in more socially desirable ways without changing underlying incentives to resource users. In a sense, these 'enabling' options allow for ES outcomes that would emerge *if* economic agents' behaviours were not constrained by an unfavourable conditioning or enabling environment. Common constraints in developing countries include the lack of basic public services, such as health and education, or the enforcement of property rights. In addition, the private sector also often fails to deliver agricultural technologies with large public good components or

[3] We emphasise the word 'attempt' because the intensity and duration with which a given instrument is used will, in part, determine its effect on human behaviour – for example, small price subsidies and short-term punishments may do little to change behaviour in the long term.

provide environmental education – in the former case, the private benefits of technological change that accrue to farmers are expensive to 'collect', while in the latter case, the benefits may not accrue at the individual level, so farmers will not be willing to pay for this service.[4] Societal demand for improved ES may in many such cases represent an argument in favour of measures that 'enable' farmers to increase natural resource use efficiency, and thereby ES provision levels, through the adoption and correct use of improved production technologies.

However, implementing enabling measures will seldom guarantee a more socially acceptable outcome; no single enabling measure is likely to 'remove' all of the constraints that preclude the desired human behaviour (e.g. credit provision may not be sufficient to overcome the effects of insecure land tenure). Or in a less-constrained situation, an individual may choose behavioural options that do not involve the targeted ES (e.g. credit provided for soil-enhancing investments may instead be used to pay for educational services for children). Research has, nonetheless, shown that in some situations enabling measures have contributed to more efficient and socially optimal ES use or to less damaging ES modifications (Kuyvenhoven 2004). Enabling policy is often viewed as complementary measure needed for effective implementation of some incentive- and disincentive-based interventions (Auty and Kiiski 2002; Börner et al. 2010). Lastly, because enabling measures, by definition and design, aim to increase options available to resource users, they can have negative spillover effects on natural resources and related ES, for example, providing rural credit in forested areas can increase deforestation unless forests are not protected effectively by additional measures (Angelsen and Kaimowitz 2001; Vosti et al. 2002).

Studies all over the world have shown that situations of poorly defined or incompletely enforced property rights (i.e. resource tenure insecurity) and nonexistence of property rights (i.e. open-access situations such as unguarded or unmanaged common pool resources) motivate natural resource 'mining' strategies, that is, the rapid exploitation of ES and ecosystem goods in the face of uncertain opportunities for future use (Hotte 2001; Schuck et al. 2002). A frequently cited example of an enabling ES management option to deal with this problem is the transfer of property rights to natural resources or related ES from (say) federal control to lower-level administrative units or even local resource users (Agrawal and Gupta 2005; Persha et al. 2011). The effectiveness of this approach naturally depends on whether the resulting new property right regime is accepted and enforceable at the local level.

Environmentally friendly technological alternatives to traditional technologies, if these alternatives can profitably compete with current practices, will likely be adopted by users without specific incentives that encourage their use or disincentives that discourage the use of environmentally more damaging technologies (Qaim et al. 2006). If, however, access to such technologies is limited by liquidity constraints, interventions such as rural credit schemes have been shown to increase adoption rates and improve ES provision levels (Anderson et al. 2002; Anderson and Thampapillai 1990).

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[4] For a list of reasons why the private sector will not provide the needed goods or services, see technical appendix in Belli et al. (2001).

Government and civil society engagement in environmental education and awareness building has shown to be a major contributor to reducing the costs of environmental management by affecting human behaviour in ways that narrow the gap between privately and socially optimal ES flows (Kollmuss and Agyeman 2002; Palmer et al. 1998).

In some cases, relatively small investments in establishing mutually beneficial partnerships can help solve environmental problems (Schwartzman and Zimmerman 2005). Research on partnerships, however, has also shown that implementing such enabling management instruments often requires long-term coordination and the establishment and maintenance of a legal regulating framework (Visseren-Hamakers and Glasbergen 2007), all of which can be expensive.

Finally, farm income (a commonly used indicator of human welfare) and some extreme ES flows are directly and negatively linked; for example, excessive rainfall can cause flooding that destroys crops or droughts can make agriculture infeasible. Insurance schemes have traditionally been used to mitigate the negative effects related to extreme weather events (Hazell et al. 1986), and the public sector has played important roles in establishing, monitoring and guaranteeing such schemes. In situations in which risk undermines the incentives to adopt ES-friendly technologies or farming practices, insurance schemes can contribute to stabilising or increasing incomes, and to ES conservation (Nail et al. 2007).

2.2.2 *Incentive-Based Management Instruments*

Whenever ES are underprovided, overused or underinvested in from a social perspective, one frequent cause is that the value of that ES is not evident to or captured by the individuals influencing their provision. In such situations, governments and local ES beneficiaries have often decided to provide direct incentives that encourage ES conservation or land use practices that provide additional ES (Portney and Stavins 2000). Subsidies represent one way of providing incentives, for example, by reducing the costs of fertilisers and fuel or providing cheap credit lines for particular agricultural activities (Huber et al. 1998; Lowe et al. 1999). However, subsidies supporting the production of goods that intensively use (e.g. water for agriculture) or compete with (e.g. forest clearing for agriculture) ES have often contributed to ES losses (Brouwer and Lowe 2000). Nonetheless, these policy instruments can be used and combined with other measures to change behaviour in ways that generate or protect ES (Oenema et al. 2006).

Payments[5] are generally perceived to be the most direct way to stimulate the provision of a given ES, and while few concrete examples are currently in place,

[5] The 'environmental services' addressed by most existing PES schemes are equivalent to ecosystem services with public good character, for example, carbon fixation and biodiversity-related benefits, or scenic beauty (Landell-Mills and Porras 2002).

PES has received much attention in the recent environmental management literature (Börner et al. 2007; Milne and Niesten 2009; Wunder et al. 2008). Costa Rica was one of the first countries to implement a national PES scheme to manage ES, such as biodiversity conservation, soil erosion control, water flow regulation and forest carbon retention (Pagiola 2008). However, the cost-effectiveness and equity effects of these pioneering projects have yet to be comprehensively assessed, especially for the case of large-scale interventions (Pattanayak et al. 2010; Wunder 2008).

Certification or ecolabelling is a widespread management instrument used to increase the market prices of products as an incentive in favour of ES-friendly production practices (Ferraro et al. 2005). The certificate or label is used to separate markets for conventional and more eco-friendly products and to allow consumers the option of paying (at a premium) for the improved management of ES. Some authors therefore refer to certification also as a market creation management instrument (Nunes and Riyanto 2001). Establishing and managing certification and ecolabelling schemes can be expensive, and an array of product, market and other conditions must be fulfilled for certification schemes to be successful in developing country contexts (Ebeling and Yasué 2009).

2.2.3 Disincentive-Based ES Management Instruments

Disincentives are the most commonly used policy instruments for environmental management, especially in Latin America (Huber et al. 1998; Seroa da Motta et al. 1996). Whenever the costs associated with ES use or modification are perceived by society to be excessive, disincentives can be used to reduce and regulate the agricultural or other activities that are causing ES losses. Measures such as regulations (e.g. forest retention laws), bans (e.g. trade bans on endangered species) and standards (e.g. gender and size restrictions on the harvesting of certain types of wildlife) are typical examples for disincentive-based management. Compliance is rarely voluntary, so fines and legal action (e.g. confiscation or even imprisonment) are often used to enforce compliance (Pearce and Turner 1990). Disincentive-based management has been largely criticised as being economically inefficient and as having negative effects on poverty (Dietz et al. 2003; Holling and Meffe 1996). Recent evidence, however, suggests that enforcement of forest conservation law has significantly reduced deforestation in parts of the Brazilian Amazon (Hargrave and Kis-Katos 2010). Yet, independent of impact assessments, disincentive-based ES management has remained popular among public policymakers, in part because regulations are relatively easy to establish (though eventually costly to enforce, especially in the context of developing countries, Robinson et al. 2010) but also because they can generate government revenue in the form of fines.

Environmental taxes, for example, on land, represent another disincentive-based policy option that has been shown to bring about both ES *and* welfare gains at least in the context of developed countries (Bosquet 2000; Johnstone and Alavalapati 1998). In developing countries, however, levying taxes to enhance ES flows can have adverse effects on the asset portfolios and income flows of the poor (Bruce and Ellis 1993).

User fees represent an option for managing ES use and modification at the local level, for example, in national parks and other forms of protected areas (Green and Donnelly 2003). User fees, for example, in the form of resource extraction permits for timber or non-timber forest products, also represent a form of regulating resource use and extraction (and hence some ES associated with these resources) (Simula et al. 2002; Sudirman and Nely 2005). Research has shown that large economic benefits can be derived from allowing controlled access to and use of protected areas, especially if they are successfully integrated in local and international markets for tourism and other rather 'nondestructive' uses (Amend et al. 2006). Whether protected areas achieve conservation objectives crucially depends on effective enforcement. Nonetheless, two global studies have recently confirmed that protected areas have worldwide significantly reduced the pressure on protected forests (Nelson and Chomitz 2011; Porter-Bolland et al. 2011). Surprisingly, community-managed forests and extractive reserves fared better than strictly protected areas in terms of conservation effectiveness, suggesting that protection need not be at odds with the sustainable management of forest resources.

2.3 Factors Affecting the Performance of ES Management Instruments

In this section, we discuss three sets of factors that influence the effectiveness of, and the co-benefits generated by, ES management instruments. We, first, focus on the biophysical characteristics of ES, such as spatial and temporal characteristics as well as complexity (Daily et al. 2009; Fisher et al. 2009; Kremen 2005). Second, we examine the socioeconomic conditions of natural resource users, which have recently become a key issue in the debate on the scaling up of incentive-based ecosystem service management under an international, REDD mechanism (Agrawal et al. 2011). Third, we explore the institutional environment for ES management that governs, at least in the short term, the pathways through which incentives, disincentives and enablement measures can be delivered on the ground and thus critically determines their cost-effectiveness (Howlett 2004; Ostrom 2008).

2.3.1 The Biophysical Characteristics of ES

The MA (2005) definition of ES included many types of benefits that humans can obtain from the environment. Understanding some of the basic properties of this mixture of ecosystem services and goods is therefore the first step to evaluating the potential costs and effectiveness of alternative management instruments.

2.3.1.1 Complexity and Interdependence

All ES result from complex geobiophysical interactions, but not all ES are equally ‘systemic’. Promoting carbon sequestration, for example, requires far less knowledge about ecosystem functioning than enhancing soil fertility or species diversity, both of which depend much more on the presence of a variety of ecological processes, and hence may be more vulnerable to ecosystem modifications. This biophysical ‘independence’ and (hence) predictability of management-instrument-specific effects has made it easier to develop quantitative measures of carbon stocks and flows in a variety of land use systems and to identify and test policy approaches to managing this important ES in the context of forest and in agroecosystems (Fisher et al. 2009; Pagiola et al. 2002). Ecosystem services related to biodiversity, on the other hand, tend to be more complex and interdependent, for example, the number of trees in a certain environment can be as important as the composition of tree species for habitat quality (Arroyo-Rodríguez and Mandujano 2006).

The MEA classifies ES based on their key functions in an ecosystem into regulating, provisioning and supporting (among others) services. This classification approach is particularly useful to identify the amounts and pathways through which ES benefits flow to specific stakeholder groups. Knowing these pathways facilitates the process of determining who could and should cover at least some of the costs of managing ES. From a management point of view, however, this categorisation of ES may be less convenient, because it groups ES with very different characteristics in the same categories. For example, managing flood versus climate-regulating ES requires hugely different sets of knowledge and policy instruments. Identifying cost-effective intervention mechanism thus often requires better knowledge about the implications of specific ES characteristics for management (Kroeger and Casey 2007).

2.3.1.2 Non-excludability

Both the challenge and the need for managing ES arise from the fact that the benefits derived from them are often ‘non-excludable’. Consider, for example, carbon sequestration in forest plantations: the climate-regulating functions of these plantations accrue to the society as a whole and not exclusively to the individual even if the individual owns the land on which the carbon was sequestered. Soil quality on private land, on the other hand, is an excludible ES that provides benefits exclusively to the owner. Conserving soil quality is thus often in the best interest of the owner.

If all benefits of a given ES can potentially be captured by the stakeholders that affect their provision, incentives and disincentives intended to increase their provision levels will usually not remove the underlying causes of ES losses. In the case of soil quality, the causes of degradation can often be traced back to the lack of access to soil quality conserving agricultural practices (Vosti and Reardon 1997).

2.3.1.3 Temporal and Spatial Dimensions and Interdependency of ES Provision

The provision of ecosystem services varies naturally over time and space (Kremen 2005). For example, rivers reach the ocean at specific points, and although the water they carry may influence ecosystems for many miles out to sea, eventually these influences disappear. Elevation patterns in mountainous regions, such as the Andes, introduce enormous spatial variations in the provision of ES, especially if they depend on climate conditions. But even in regions with less heterogeneous elevation patterns, such as the Amazon basin, bedrock characteristics and tidal inundation introduce additional spatial variability to ES provision and related benefits. For example, tidal movements in the Amazon allow for electricity generation in small-scale tidal power plants along some, but not all, rivers and temporally inundated areas (Charlier 2003).

Often there are also multiple temporal patterns to ES provision – diurnal, monthly, seasonal and interannual. For example, huge diurnal temperature variations in some mountainous regions affect the types of vegetation that will naturally occur and the agricultural crops that can be grown. Or, to take another example, river discharge in the Amazon has been shown to vary enormously depending on the ENSO cycle with implications for hydropower generation, fluvial transport and fishery production (Richey et al. 1989). These natural patterns affect the patterns of responses of ES to management interventions.

With regard to both space *and* time, there can be great uncertainty regarding ES flows. For example, we may not know where the end point of influence of a particular stream flow might be at a given point in time, because weather patterns in a given year can extend or reduce that stream's flow. Some of this uncertainty can be reduced or better understood with the proper investments in research/monitoring, but other aspects of this uncertainty will be difficult to discover; this is especially true for ES that have yet to be concretely defined or measured. Regardless, decision-makers wrestling with selecting management instruments and crafting them to be cost-effective in specific agroecological and socioeconomic circumstances should know that while 'on average' these management instruments may succeed in meeting stated ES objectives, there will be times and places (within their target temporal and geographic domains) when/where their final mode of intervention will overshoot and undershoot these same objectives. The costs to society of these over- or undershootings may be quite significant.[6]

Finally, as a result of these spatial and temporal dynamics, the social benefits of ES management in a given place and time may accrue to different stakeholder groups many years later, for example, slowing deforestation in the Amazon may help retain historical rainfall patterns at a continental scale (Werth and Avissar 2002).

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[6] For example, the value of surface water during the wet season can be much lower than the value of surface water during the dry season (Torres et al. 2012).

2.3.1.4 Implications of ES Characteristics for Choosing ES Management Instruments

Complexity, interdependence and uncertainty in temporal and spatial ES dynamics mean that management interventions can be ineffective or, worse, produce unexpected negative consequences. An important first step toward managing ES is to recognise that humans simultaneously adapt to and change ES provision through activities that alter natural temporal and spatial dynamics. These activities and related investments are undertaken to harness the private benefits of ecosystem services (e.g. diverting river flows to irrigate agricultural products) or to reduce the private damages associated with ecosystem disservices (e.g. building levees to reduce flooding), or for reasons that are not directly related to ES but still can affect ES flows. ES-modifying activities are thus not distributed randomly over the landscape; spatial patterns of investments and activities will be carried out where their private economic returns are positive, and they will be undertaken first in areas where these returns are highest (Chomitz and Thomas 2003; Pfaff 1999; Thünen 1826). Rapid and unexpected changes in ES caused by management interventions thus imply potentially large costs to those who are particularly dependent on (or who have adapted to) the ES targeted by policymakers for intervention. This often applies, for example, to downstream water users after the construction of dams.

It should also be noted that changes in natural phenomena can have similar welfare-reducing effects on individuals and communities that have developed livelihood strategies based on expected ES flows. For example, the 2005 drought in the Amazon exemplified the implications of water shortages to the local and regional economies, for example, disruptions in local and regional river-based transport, that have developed under conditions of relative water abundance (Zeng et al. 2008).

A first step toward ES management is therefore to understand how the characteristics of the targeted ES and its current use affect:

1. The types of benefits that the ES provide (e.g. income, air quality)
2. The ways in which these benefits are generated (e.g. income via agriculture)
3. The ways in which temporal and spatial patterns of benefits are generated
4. To whom these benefits directly and indirectly accrue

The second step is to ask how alternative ES management instruments affect these four items.

In practice, we may not have the means to measure actual ES, for example, accurately measuring watershed services can be exceedingly complex and expensive (Perrot-Maitre 2006). Hence, decision-makers are often forced to manage land uses (or other broader units that are relevant for ES provision) in the hopes of influencing specific ES flows (Wunder 2005). A common approach to managing these cases involves identifying (but not necessarily measuring) the target ES and the land uses or land cover types deemed most likely to generate it (e.g. river bank vegetation to reduce erosion and thus water turbidity levels). It is assumed that if the policies are successful in retaining or expanding the target land uses, this success will be proportionately replicated for the targeted ES. The literature has pointed out the problem associated with many of the assumptions underlying this approach, for example, the extent to which heterogeneity within land use categories can affect ES flows. For example, a forest use regulation and a cap-and-trade system may be equally effective in retaining a specific total amount of forested land; however, the geographic distribution of the forests retained by the two interventions will likely be different, with possible consequences for some ES (Debinski and Holt 2000).

One important implication is that decision-makers are generally forced to manage ‘bundles’ of ES by selecting management instruments that affect land use and land use change. Bundling may often allow decision-makers to credibly suggest that unknown or highly undervalued ES are included in these bundles (e.g. conserving primary forest carbon stocks through REDD will also conserve biodiversity). But, depending on how well individual components of a ‘bundle’ can be measured, this approach makes it harder (and more expensive) to identify the beneficiaries of ES management actions and to articulate demand for management actions (Wunder and Wertz-Kanounnikoff 2009).

2.3.2 Institutional and Socioeconomic Factors Affecting the Performance of ES Management Instruments

A series of institutional factors can affect the performance of ES management instruments. Often, ES benefits are not directly related to natural characteristics. Instead, various layers of property rights attached to natural resources through legal and/or customary norms and regulations usually govern local access to and use of ES, for example, living next to a river does not necessarily convey water use rights (Ostrom et al. 2005; Schlager and Ostrom 1992). In developing country contexts, the enforcement of legally defined property rights is often weak. In the Amazon and in the neighbouring Andes region, many natural resources are de facto open access resources with ill-defined, incomplete, nonexistent, conflicting or weakly enforced property rights (Ravnborg and Guerrero 1999; Seroa da Motta and Ferraz do Amaral 2000).

Lack of administrative capacities and operational infrastructure is often the reason for poor enforcement of property rights and at the same time limit the effectiveness and increase the costs of incentive and disincentive-based management options (Börner et al. 2011; Robinson et al. 2010). For example, PES schemes may be ineffective if the recipients (e.g. landowners) cannot exclude others from modifying ES originating from their land. Moreover, if property rights are poorly defined, regulators will find it harder to establish liability for illegal natural resource degradation. Some research, nonetheless, suggests that offering PES may actually encourage property right enforcement by rural communities and, overall, lead to positive environmental and welfare outcomes (Engel and Palmer 2008). Effective property right transfers or supporting local communities to build and maintain efficient local institutional arrangements that regulate resource use and access will nonetheless often be necessary to address situations where property rights are ill defined and/or weakly enforced (IFAD 2003; McGrath et al. 1993).

Even if effective institutions are in place to implement and monitor ES management instruments, socioeconomic conditions will play an important role in affecting the performance of ES management instruments. A straightforward rule that applies to incentives (e.g. payments) or disincentives (e.g. fines) for ES management is that the costs of noncompliance must outweigh the benefits, that is, compliance levels tend to be low if the expected value of fines are smaller than the expected benefits associated with noncompliance, because of low fine levels and/or low probability of enforcement (Becker 1968). Imperfect enforcement of payment contracts may have even worse implications for the cost-effectiveness of this particular management instrument, because higher (than opportunity cost) transfers will be needed to pursue recipients to comply with the conditions attached to payments (Börner et al. 2011).

Poverty, which is often associated with (or caused by) limited access to basic public services, credit and agricultural technologies, can represent a significant barrier to cost-effective ES management. If disincentive-based interventions restrict poor people's access to essential natural resources and their locally valued ES, management instruments may be ineffective, at best, or even deepen poverty. Poverty also typically coincides with poor institutional and organisational capacity, which represents a challenge for most enabling and incentive-based intervention options. It has, for example, been argued that the spatial coincidence of poverty and valuable ecosystem services, especially in Latin America, comes with an opportunity for achieving win-win outcomes through PES. Several studies, however, emphasise that high transaction costs (for both scheme implementers and for the poor) and the lack of formal land titles can limit the participation of poor ES service providers in such conditional incentive schemes (Pagiola et al. 2005; Pfaff et al. 2007; Rios and Pagiola 2010). Poverty alleviation objectives of PES, in addition, may be jeopardised if landless poor rural dwellers lose employment opportunities in set-aside (purely conservation-oriented) as opposed to asset-building (e.g. reforestation) schemes (Wunder 2008; Zilberman et al. 2008).

Farmers may also not be poor by standard poverty measures, but still be too poor to invest in sustainable land use practices, that is, they may suffer from conservation investment poverty (Vosti and Reardon 1997). In such cases, disincentive measures intended to induce more intensive land uses may fail to bring about the desired outcomes. However, even PES may fail if conservation investments are subject to severe liquidity constraints, for example, the purchase or rental of heavy land machinery. Yet, depending on conditioning factors, poverty may also inhibit environmental degradation. For example, subsidies to encourage investments in seemingly sustainable land uses can result in more, rather than less, deforestation and declines in forest ES (Börner et al. 2007; Vosti et al. 1997).

Finally, many management instruments in the enabling category, such as community-based resource management, require collective action, civil engagement and local organisational capacity, result in enhanced ES provision (Kellert et al. 2000). In recently settled areas, such as at the margins of tropical moist forest, most of these types of social capital are often in short supply (Fearnside 2001). Interventions that rely heavily on preexisting social capital may thus often represent promising options only in the long run and face large establishment and maintenance costs.

2.3.2.1 Implications of Institutional and Socioeconomic Factors for ES Management

Local institutional and socioeconomic circumstances mean than no blueprint approaches exist to managing ES, especially in the developing world. A couple of general lessons nonetheless emerge. First, when ES losses are the result of lacking formal or customary institutional structures and/or poverty, direct ES management through most incentive and disincentive-based instruments is unlikely to remove the root causes of ES loss and be ineffective and/or costly at best or result in undesired outcomes at worst. Enabling measures, such as provision of basic education and other public services as well as improving access to locally adapted technological innovations may instead often represent more effective initial investments toward improving ES provision.

Second, the size and timing of the net benefits associated with ES-modifying activities and investments and the stakeholder groups to whom these benefits accrue will influence ES management outcomes. For example, where such activities and investments are very profitable (as is often case for the extraction of precious timber resources), PES schemes will be expensive and perhaps beyond the fiscal means of policymakers or willingness of ES beneficiaries to pay. Under such circumstances, land use taxes (which reverse the financial flows among stakeholder groups, vis-à-vis PES) or land use regulations may be more cost-effective ES management instruments, even when enforcement/monitoring costs are included in the comparison.

Third, at least in the short term, trade-off relationships between ES objectives and other development objectives are the rule rather than the exception.[7] Hence, ES managers must recognise that individual ES management instruments may often not achieve both poverty alleviation and ES management objectives. Poverty alleviation generally requires a broader development approach involving various non-environmental policy investments and activities. The existence of trade-offs also often calls for negotiation-based solutions, in which stakeholders need equal opportunities to guarantee fair outcomes. If direct negotiation between ES users and modifiers is an option, for example, in small watersheds, PES schemes may often require little or no government involvement. Reducing deforestation at a larger scale will, however, always depend on the involvement of governments, because ES beneficiaries lack the means and legal mandate to monitor and enforce conservation contracts.

Fourth, because ES management is often land use based, rural landless people may be affected in unintended ways. Especially when ES management options, such as REDD, are applied. ES managers must therefore consider safeguards for this and other vulnerable groups.

[7] More fundamentally, economic efficiency requires identifying specific policy instruments to resolve specific policy problems; it will rarely be the case that an environmental policy instrument is the most efficient way to resolve (say) an economic development problem.

Finally, an old paradigm is that local problems require local solutions. As we have seen, this holds for some but certainly not all problems related to ES provision and use. Many central governments have tried to address this notion by delegating the management of some natural resources (and hence ES) to lower-level administrative units, such as states or districts, in both developed and developing countries. Decentralised management, however, poses new challenges to effective and efficient ES management, among them the risk that unprepared and underfinanced local governments lack the administrative and technical capacities to take and effectively implement policies related to ES flows (see Toni and Kaimowitz 2003 for the example of forest management in the Amazon).

But even if local governments and local civil society *are* prepared to cost-effectively handle local ES flow management challenges, such challenges are generally not exclusively local but rather 'spill over' spatially and temporally into the domains of other decision-makers. Some very important ES flows with large public good components do not coincide with, or are not contained within, administrative boundaries, so managing them requires cooperation across policymaking boundaries. Among the Amazon and Andean countries, this notion has given rise to the foundation of the Amazon Cooperation Treaty Organization (ACTO). However, multilateral environmental agreements and partnerships around the world are plagued with the same difficulties, for example, free riding (individuals, communities or even countries reaping the benefits of ES management without paying their share of management costs) and high transaction costs of intergovernmental negotiations (Chang and Rajan 2001).

2.4 ES Management Instruments and Expected Performance: An Overview

This section highlights selected factors that our review of the literature suggests are likely to affect cost-effectiveness and poverty alleviation objectives of ES management. Table 2.1 summarises key factors that (if present) influence the performance of selected ES management instruments.

Column 1 of Table 2.1 identifies the management option type, column 2 identifies the ES to be managed, column 3 addresses poverty alleviation, column 4 identifies factors that affect the cost-effectiveness of reducing poverty and the final column (5) examines factors influencing the cost-effectiveness of achieving ES management objectives.

Beginning with column 2, it is clear that most instruments can be used in theory to address either specific ES *or* a bundled set of ES. In practice, however, the great majority of ES management instruments has been used to influence human behaviour with respect to broad natural resource categories, such as forests or fishing grounds among many others (as suggested in Fig. 2.2), with expected direct effects on specific ES flows, most of which are not measured in detail. The notable exception

Table 2.1 ES management instruments and factors expected to influence their cost-effectiveness

ES management instrument	ES target	Conditions with potential to reduce (or not exacerbate) poverty	Factors reducing cost-effectiveness in achieving poverty alleviation if present	Factors reducing to cost-effectiveness in achieving ES objectives if present
PES (subsidies)	Specific Carbon Biodiversity Water Scenic beauty	If the poor can offer additional ES provision and If the poor have rights to exclusion	Poor can offer little additionality Poor have weak and insecure property rights No legal basis for PES from treasury	Spatial heterogeneity of opportunity costs High opportunity costs High transaction costs (monitoring and enforcement)
Taxes (user fees, etc.)	Specific and unspecific Timber NTFP Water Carbon Biodiversity Scenic beauty Climate regulation	If tax revenues are reinvested in (compensating) poverty alleviation measures or If the poor can directly capture the revenues or If the poor are excluded from the tax	If poor are allowed to capture user fees, they often lack capacity to attract users Few examples of environmental taxes	Weak enforcement capacity High transaction costs (monitoring and enforcement)
Certification	Specific and unspecific Water/air quality Carbon Biodiversity Soil quality Climate regulation	If the poor have market access and If the poor can capture price premiums and If the poor can meet quality standards	Limited market access Costs of certification process Costs of investments in to technology	High transaction costs (monitoring of difficult to measure ES) Weak enforcement capacity

Technological innovation	Specific and unspecific Various	If the poor have technology access (knowledge, extension, infrastructure) and If the poor can afford up-front investment costs	Limited technology access Investment poverty	High opportunity costs of conservation investments High development costs (R&D system)
Regulation (bans, standards, protected areas)	Specific and unspecific Various	If regulations do not affect essential ES consumption (or if so, compensating mechanisms are in place) or If regulations include special treatment of poor ES users	Low bargaining power to negotiate compensation and special treatment	Weak enforcement capacity High transaction costs (monitoring and enforcement) if under pressure
Local institutional arrangements (community-based resource management, partnerships)	Specific and unspecific Various	If the poor have equal bargaining power in the negotiation process or If institutions are set up in a way that promotes poverty alleviation	Low administrative and organisational capacity of local governments Low bargaining power Few cooperative experiences	High returns to free riding
Environmental education	Specific and unspecific Various	If the poor have access to education (infrastructure, costs) and If education addresses ES issues relevant for the poor	Low level of teacher training in rural schools High costs due to poor transport infrastructure	High returns to environmentally damaging behaviour (lack of rule of law)
Cap-and-trade schemes	Specific Air quality Climate regulation Various others	f the poor are sellers and If the poor have market access	Limited market access	Weak enforcement capacity High transaction costs (monitoring and enforcement)

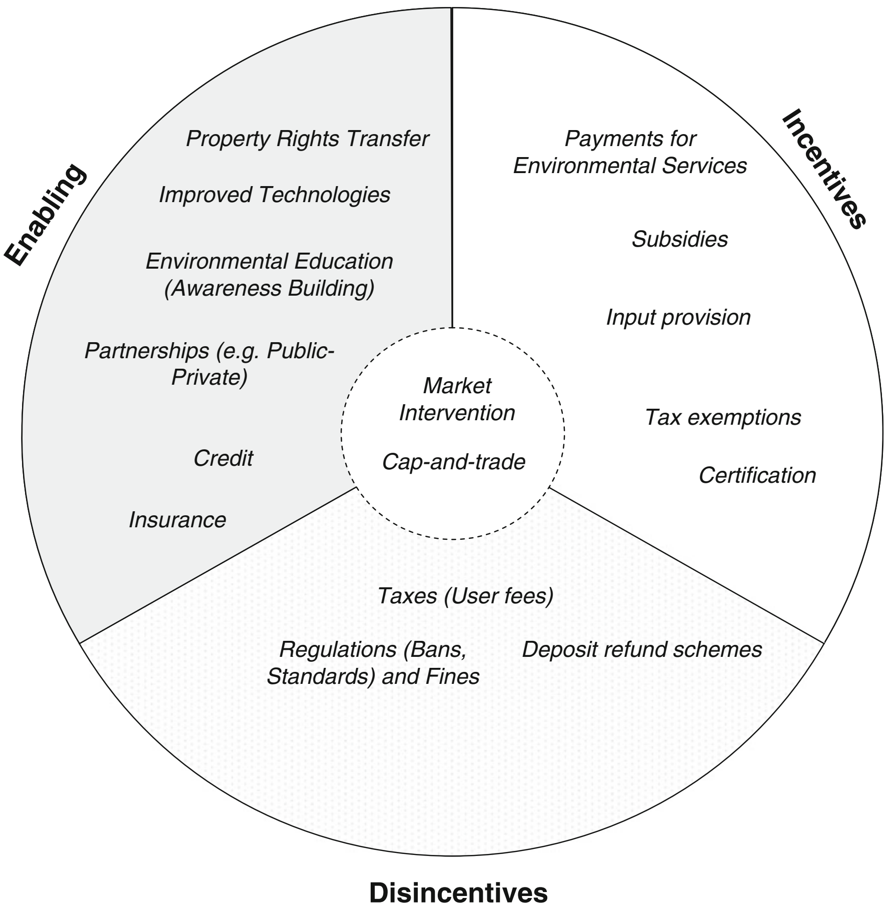

Fig. 2.2 ES management instruments by means of impact

are PES schemes that are often specifically designed to address one or two well-defined ES, such as carbon or watershed services, but which likely have spillover effects (of different magnitudes and perhaps in different directions, vis-à-vis the targeted ES) on other ES.

Almost all instruments can potentially be designed in ways that leave the poor unaffected, or perhaps even better off (column 3 in Table 2.1). However, designing and implementing measures to achieve poverty neutrality or to reduce poverty generally implies additional up-front costs (e.g. those associated with building participatory or institutional capacity) and operational costs, and decision-makers have not always been willing or able to incur these costs. Poverty effects, nonetheless, tend to vary across ES management instrument categories. Poverty effects of 'enabling' instruments, for example, those delivering technological innovations, tend to depend on whether the poor will be able to reap their benefits of ES management. When access to technological innovations is limited, the poor generally do not benefit and may even become poorer, for example, when productivity increases among nonpoor adopters result in lower product prices. Or, the poor may lack experience and (hence) skill and bargaining power, which can limit their ability to effectively participate in the design of ES management, such as community-based resource management or public-private partnership agreements.

In the case of disincentive-based instruments (e.g. taxes or fines), mechanisms need to be in place to compensate low-income groups (e.g. tax exemptions) for negative welfare consequences or instruments need to be designed in ways that leave the poor unaffected (e.g. allowing the resource-poor to continue to engage in subsistence activities in protected areas). Incentive-based instruments, such a PES, on the other hand, often require a minimum level of market access to work effectively. For example, conservation payments to farmers living in remote areas without access to food markets may not compensate for losses in subsistence production. Also, mechanisms need to be in place to make sure that price premiums actually trickle down to the poor instead of being captured by intermediaries, as has been the case for some forest certification schemes (Harris et al. 2001).

In many countries, ES modifying activities are already highly regulated, at least on paper. For example, over 40% of the Brazilian Amazon region is covered by various categories of protected areas and indigenous territories, whereas the remaining public and private land is subject to the national forest retention standard that requires 80% of landholdings to remain under primary forests. Many other countries, like Peru, have banned deforestation almost completely. In practice, however, deforestation continues to take place (illegally) wherever it is profitable to do so, that is, mainly alongside roads and highways (Laurance et al. 2002). Increasing the effectiveness of existing regulatory policies by enforcing them more rigorously is thus often seen as a low-hanging fruit for ES management. But, especially in remote parts of the Amazon, where field-based enforcement of disincentives can operationally become more costly than providing incentives, PES for avoiding deforestation may come to be a cost-effective complementary measure (Börner et al. 2010; Nepstad et al. 2007; Swallow et al. 2007).

Where property rights are secure, resource users are relatively homogeneous and communities are willing to cooperate; building capacities for more effective ES management is likely to help maintain essential ES and contribute to poverty alleviation. However, strong incentives for ES modifying activities will always represent a major challenge for ES management; wherever such incentives are high, ES management will also be costly. This is especially true if ES do not provide direct benefits to resource users or if beneficiaries do not have a voice to negotiate ES outcomes. Hard trade-offs therefore need to be faced by those that promote economic growth and infrastructure development, for example, at forest frontiers, unless the global community that benefits from forest-based ES is willing to compensate land users for foregoing economic opportunities.

2.5 Implications for Research, Capacity Strengthening and Policy

This chapter reviewed the theoretical and applied literatures on instruments that can be or have been used to manage ecosystem services (ES). The primary objective was to explore the biophysical and socioeconomic factors that affect the cost-effectiveness of alternative ES management instruments. A second objective was to assess the effects of ES management instruments on the welfare of the rural poor inhabiting areas targeted for ES management. We conclude here by indentifying key knowledge gaps with regard to three essential questions that decision-makers need to answer before making informed decisions on ES management.

First, what do we know about ES dynamics and their relationship with poverty? In terms of measuring poverty (especially measured in terms of income), we are on solid theoretical and empirical grounds – we can measure poverty and poverty dynamics – so any major gaps in knowledge are primarily attributable to insufficient resources having been dedicated to identify the poor and measuring the depths and nature of their poverty. As regards ES dynamics, the scientific base is much weaker – with relatively few exceptions (e.g. biomass carbon measurement), science has yet to provide decision-makers with practical measures of ES that capture their most important spatial and temporal dynamics. Complexity and uncertainty, however, suggest that such measures may often not exist. In times of climate change, research on how ES management can optimally deal with persistent and/or changing uncertainty is therefore all the more necessary.

Second, if stakeholders are unhappy with current levels of ES provision, why and what are the nature and degrees of their displeasure? Answering these questions requires knowledge of the private and social benefits associated with ES, the costs associated with changes in these flows and how these benefits/costs vary across stakeholder groups. Progress on this front has recently been made, for example, through initiatives like The Economics of Ecosystem and Biodiversity (TEEB),[8] but large gaps in knowledge remain particularly in developing countries. The most important of these gaps is the development of methods to measure the benefits (though not always necessarily the monetary value) associated with particular ES or bundles of ES, and how these benefits change as ES are modified. To address concrete decision-makers' needs, these methods must incorporate the site-specific patterns of ES flows and generate ranges of expected benefit flows that capture the inherent uncertainty associated with important ES and their values to society.

Third, if policymakers decide to take action, what should be done and by whom? Answers to this question must build on the site-specific responses to the previous two questions plus an understanding of the determinants of the behaviour that modifies the ES in question as well as its responsiveness to management intervention. Research on land use and land cover change has made significant progress in understanding household level decision-making, but many of the most pressing

[8] www.teebweb.org

ES management problems will have to be resolved at the community or at more spatially and socioeconomically aggregate levels. To extract lessons learned from current and past environmental policy initiatives, more rigorous impact assessments are needed that lead us to understand how policy design and the local conditioning environment affect the performance of individual and combinations of ES management instruments. Often such evaluations will have to rely on methods that allow establishing credible counterfactuals for policy interventions, such as matching analyses (Ferraro and Pattanayak 2006).

Finally, ES management choices are not merely decisions of independent social welfare optimising principals but the result of political bargaining processes. Work on new ways of establishing and managing dialogue related to ES management among stakeholder groups is progressing; such dialogue is generally seen as a necessary condition for achieving successful and sustainable outcomes. However, large gaps remain in identifying the most efficient methods of establishing and managing multi-stakeholder interactions and in developing mechanisms to generate and deliver needed science-based information into such discussion settings.

As the debate around reducing emissions from deforestation and degradation (REDD) evolves, many proponents begin to realise that only incomplete answers exist to the three questions posed above. The resulting uncertainty about how REDD could and should be implemented has led to mounting opposition against the concept that threatens its successful implementation (Agrawal et al. 2011). Forward-looking, scenario-based policy research that openly deals with the uncertainties attached to costs as well as environmental and social impacts of ES management could provide crucial input for a constructive, outcome-oriented policy debate.

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Chapter 3
Partnerships in Global Governance: The Growth of a Procedural Norm Without Substance?

Teresa Kramarz

Public-private partnerships (PPPs) have become a favourite governance arrangement among most international organizations (IOs) working on global environmental issues. The World Bank has not only featured partnerships with business and non-governmental organizations (NGOs) as a best practice in global environmental initiatives but also incorporated them into its mission statement: "Our mission is to fight poverty with passion and professionalism for lasting results and to help people help themselves and their environment by providing resources, sharing knowledge, building capacity and *forging partnerships in the public and private sectors*" (World Bank 1999). The Bank has even referred to them as the only way to do business. "For the World Bank or WWF, or any other institution intent on securing our future on a liveable planet, going it alone is not an option," states President James Wolfensohn in the 1999 Annual Report of one of the Bank's first high-profile partnerships in biodiversity conservation (World Bank/WWF Alliance 1999).

These PPPs (called partnerships from now on) are institutionalized, co-governance arrangements between the Bank and private actors, including business firms, NGOs, foundations, trade organizations, academic institutions or other actors independent of state governments. Jointly, they design, deliver and finance conservation activities in developing countries. They have been lauded as textbook examples of cooperation that can produce innovative solutions to complex global problems, enhance the participation and decision-making of stakeholders and generate needed financial resources from dedicated partners to address the provision of global public goods.

Partnerships are framed as precisely the kind of solutions required to address global environmental problems and are sometimes recommended as the solution to problems that are yet to be fully identified. To the extent that they are taken for

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granted as an optimal governance tool, I refer to partnerships as an established norm in global governance. Some have written about them as a trend with no alternative (Richter 2004).[1] But how does the partnership norm matter?

The environment has been one of the primary focal areas for global partnerships. Most IOs have developed partnership programmes within their environmental portfolios (Andonova 2010). Inge Kaul has noted in a study of 100 global partnerships that the environment and health are the main issue areas that generate partnerships (Kaul 2006). This focus on global environmental partnerships parallels the phenomenon at the World Bank. The environment is the most important sector accounting for the largest number of partnerships and dollar volume. The Private Sector and Infrastructure group follow second in importance (World Bank 2002).

Within the environmental sector, biodiversity is an arena where traditional policies and authority to govern have been heavily contested, while financial means to address the problem has been very limited (Kramarz 2008). Hence, the partnership approach potentially opens up governance space by bringing in new actors, interests and ideas; devolving decision-making authority; and mobilizing new sources of funding. The case of biodiversity conservation is particularly suited for analysis because, like climate change, the appropriateness and legitimacy of global governance responses are divided along north-south, global-local, public-private lines (Backstrand 2006). Yet, unlike climate change, conservation has received much less theoretical attention in the environmental governance literature. This is a significant deficit in our research if we note three essential qualities of the biodiversity loss crisis: Biodiversity constitutes "the very foundation of human existence" (Toly 2004, p. 49 as quoted in); that foundation is disappearing and that loss is irreversible (Toly 2004).[2]

Much of the analysis on partnerships has focused largely on questions of policy effectiveness, including performance (Brinkerhoff 2002), optimal design and management features (Beisheim forthcoming), problem-solving capacity (Biermann et al. 2007) and extent to which they can fill implementation gaps (Reinicke and Deng 2000; Benner et al. 2003; Biermann et al. 2007).[3] Some authors have argued that partnerships have limited power to address the three mainly functional deficits in global governance: regulation, participation and implementation (Biermann et al. 2007). Others have deemed them potentially legitimate and effective solutions to global governance (Börzel and Risse 2005).

Within this growing literature, IOs are recognized for their special role as entrepreneurs, orchestrators or nodes of said partnerships (Andonova 2010; Abbott et al. 2010). However, the literature focuses on the policy effectiveness of the resulting partnerships, with insufficient attention given to their substantive significance. The question of what policies IO partnerships promote or what governance norms

[1] Richter is critical of this view, but highlights the degree to which PPPs are taken for granted as optimal arrangements.

[2] "Some 60% of the ecosystem services examined in the Millennium Ecosystem Assessment—including fisheries and fresh water—are being degraded or used in ways that cannot be sustained" (Island Press 2007, p. 3).

[3] There are exceptions to this functional driven trend. For example, see Börzel and Risse (2005).

they enact by how they choose to finance certain projects and not others has not been systematically addressed.

I take the view previously argued by Bernstein (2001) and Conca (2006) that literature on cooperation in international relations has largely ignored the importance of "principled content" and the direction in which those principles are trending. There has been recognition that partnerships are potentially emerging authorities in global governance, but the question of how democratically partners exercise that authority has been under-explored. Finally, the financial impact of IO partnerships vis-a-vis traditional multilateral spending has received marginal attention.

Despite this lacuna, it is the substance of partnership governance that has awakened a number of fears. Many practitioners and academics have warned of a new era of private governance, where firms will shift from being rule takers to rule makers and nature will become commoditized for profit. Or an unelected cadre of NGOs will gain global governance authority.

I argue that partnerships in some issue areas and institutional settings have become entrenched as procedural norms, but their substantive significance requires deeper investigation. This chapter proceeds in three parts. First, it provides a brief introduction to the theoretical lens of norms. Second, it narrows the definition of partnerships. Third, it traces the growth of the procedural norm to partner at the Bank and distils from this history three substantive goals the Bank envisioned achieving through partnerships: policy innovation, democracy and additional financing. Fourth, using biodiversity conservation as an issue well suited for substantive analysis, it suggests parameters for an evaluation of achievements in this area.

3.1 Theoretical Approach

Constructivists have provided valuable insights on the significance of norms in IR. Norms are collectively held rules of appropriate behaviour. They depend on a shared moral judgement of what conduct *ought* to take place in a given context and as such have power to prescribe behaviour (Finnemore and Sikkink 1998).

Norms are different from mere ideas, in that they are held intersubjectively. "Unlike ideas which may be held privately, norms are shared and social; they are not just subjective but intersubjective" (Finnemore 1996, p. 23). For example, actors hold different beliefs of appropriate responses to biodiversity loss. When partners hold these beliefs intersubjectively, they create the standards of what will be considered good governance.

Norms may start as principled commitments by entrepreneurs. If these are effectively diffused and a critical mass of norm followers adheres to them, a tipping point is reached and a norm emerges. The degree to which the norm can be "cascaded" through institutions, such as IOs, can account for its growth. This is the model of a norm's life cycle (Finnemore and Sikkink 1998).

Therefore, IOs are important conveyor belts of ideas. As Park (2009) and others have discussed, they can diffuse norms across a wide range of issue areas from human rights to alternative conceptions of security, legitimize norms, teach states norms or be a final arbiter of contested norms by institutionalizing one among many divergent visions (Claude 1966; Finnemore 1993; Barnett and Finnemore 2004; Park 2009). The World Bank is not only the largest financier of biodiversity initiatives but also one of the largest lenders in development. Its financial clout gives it "a special position in international development policy, [and] it has in many cases the power of definition and interpretation" (Jakobeit 1999, p. 5). Uncovering how the Bank's partnerships define who is the appropriate authority to govern and who are the appropriate beneficiaries of conservation can help us understand the growth of substantive governance standards.

It is important to consider the dual dimensions of procedure and substance in norm evolution. Both do not necessarily accompany or complement each other. Just as legal systems have bodies of procedural and substantive rules, in global governance, both dimensions of a norm can coexist and reinforce each other, or decouple in potentially dysfunctional ways. A new procedural norm may have the desired substantive consequences, but there is no a priori reason to assume this. Social constructivism proposes the analytical distinction of constitutive versus regulative norms. The former create new actors or structures, such as partnerships, while the latter regulate behaviour within them (Finnemore and Sikkink 1998; Adler 2002).[4]

In this chapter, I suggest that the procedural aspect of partnerships, or partnering understood as a constitutive norm, stands on solid footing. The substantive component of this form of governance, or partnering understood as enabling regulative ideas, needs to be further examined.

3.2 Defining Partnerships

One of the clearest demonstrations that partnerships are entrenched procedural norms is the difficulty that arises in trying to define them. Institutions consistently claim to be working "in partnership." The partnership concept has become so fashionable in IO circles, such an imperative tag on speeches, that defining what actors mean when they say they are working in a partnership can be a daunting task.[5] As Nelson argued, "There has been a tendency, within the United Nations system and elsewhere, to use the concept of partnership very loosely to refer to almost any kind of relationship" (Nelson 2002).[6] Even one-time meetings refer to themselves as public-private

[4] The dynamic interaction between these two types of norms in environmental governance requires greater theoretical analysis, but is beyond the boundaries of this chapter.

[5] Stephen Linder (1999) offers a helpful review of the multiplicity of meanings of partnerships in contemporary discussions. His taxonomy reveals usages that vary from privatization disguised as partnering to actual power sharing structures.

[6] For example, the UN's definition of a UN-business partnership shows the vagueness with which the concept is used: "a mutually beneficial agreement between one or more UN bodies and one or more corporate partners to work towards common objectives based on the comparative advantage of each, with a clear understanding of respective responsibilities and the expectation of due credit for every contribution" (Tesner and Kell 2000, p. 72).

partnerships (Broadwater and Kaul 2005). However, it is not all rhetoric since new institutions are created to run partnerships, financial resources are committed, and actual projects are implemented on the ground by these governing arrangements.

The first step in assessing the substantive significance of partnerships is to narrow them down conceptually. In this chapter, I refer to partnerships that meet five conditions: They are global, co-governed, public-private, service-providing initiatives that finance biodiversity conservation. They are global if they implement activities across a number of countries. They are co-governed when they include "the establishment of a new organization or entity with separate governance and management structures" (World Bank 2004, p. 111). Restricting the definition of partnerships to only those cases where partners share decision-making authority is what distinguishes partnerships from other types of collaborations (Richter 2004; Nelson 2002). In contrast to partnerships, traditional environmental programmes can include the participation of many actors, but these do not have authority to identify the nature of the problem or prescribe related responses. Another condition is that partnerships are composed of both public and private actors. The Bank and other multilateral or bilateral organizations represent the public sphere, while private actors can include firms, non-governmental organizations, foundations and academia. Partnerships between the Bank and other multilateral organizations date back many decades. However, the novel development of public and private actors sharing governance functions arises in the 1990s and poses a different set of questions regarding democracy, innovation and financing that should be further explored.

Partnerships are often differentiated according to their function (Börzel and Risse 2005; Andonova et al. 2009; Beisheim forthcoming). I suggest analyzing partnerships that implement activities on the ground. The discourse of these partnerships is important and should be studied, but because practising that discourse requires deciding how to allocate limited funds, the implementation aspect is a defining moment for demonstrating normative priorities. Examining service providing partnerships in biodiversity conservation offers an excellent opportunity to analyze what global environmental governance means in practice. Finally, these partnerships should be primarily focused on addressing biodiversity loss.

The conditions detailed above correspond roughly with the Bank's own criteria of global partnerships, which state that these initiatives must generate benefits "intended to cut across more than one region of the world and in which the partners: reach explicit agreements on objectives; agree to establish a new (formal or informal) organization; generate new products or services; [and] contribute dedicated resources to the program" (World Bank 2004, p. 2).

3.3 Growth of an Approach: History of the World Bank and the Partnership Model

In the next two sections, I trace the evolution of the procedural norm then offer parameters for analyzing its substantive significance.

In the mid-1990s, the Bank initiated a growing number of global partnerships. By 2009, it had launched 206 global partnerships with resources of $2.8 billion dollars (World Bank 2005). What follows is a narrative of how partnerships became a prominent business line for the World Bank.

The meaning of what it is to be a partner and what defines a partnership has undergone significant transformation. Originally the concept focused on the reciprocal responsibility for development outcomes between donor and recipients. By the mid-1990s, "partnership" referred to inclusion, participation and empowerment of civil society and business actors. It became defined, albeit rather vaguely, as sharing risks and opportunities. It also became an invitation to and an opening for private actors to join the governance process, catalyze new financial resources and innovate policies for the provision of global public goods.[7]

In 1967, World Bank President George Woods (1963–1968) called for the formation of a commission of experts, at the end of a discouraging decade for development results, and concerns about decreasing flows of financial aid to the developing world. The Pearson Commission on International Development, which became known simply as the Pearson Commission after its chair, was tasked with reviewing the previous 20 years of development experience and providing recommendations for the strategic directions of World Bank development assistance (World Bank 2003). The resulting report, entitled "Partners in Development," was presented to incoming President Robert McNamara in 1969. The reference to "partners" signalled a new relationship between donors and recipients. It referred to partnerships as a bargain between developing and donor countries, where the former would commit to poverty reduction and good governance and the latter to consistent flows of development aid (Pearson 1969).

The concept of partnering as shared responsibility between donors and recipients as equals was not surprisingly received with widespread criticism coming from an institution identified with top-down prescriptions and aid conditionality. Critics argued that the World Bank was now using partnerships as vehicles for deflecting responsibility for aid effectiveness (Helleiner 2000). On the one hand, the Bank dictated prescriptions for development loans, but on the other, it could invoke the partnership concept as justification for shared responsibility in the outcomes of those prescriptions. For instance, structural adjustment policies, which encourage neoliberal prescriptions as a condition to Bank loans, are one of the most widely

[7] A public good is non-rivalrous and non-excludable in consumption. As Inge Kaul et al. (1999) have stated, there are very few pure global public goods. Impure public goods such as club goods and common pool resources are the more prevalent cases in the global commons. In either case, the market alone does not provide for their sustainable consumption, and states have to coordinate to ensure it is not over consumed and depleted. This is the case of many environmental issues including biodiversity. As Kaul has argued, global public goods have the added concern of providing equally for all beneficiaries. To qualify as a true global public good, provision must be quasi universal: across generations, state borders, socio-economic groups and gender. This definition is a demanding one for state coordination and suggests an important role for international organizations that can facilitate consultations, negotiations, monitoring and follow-up operations in countries (Kaul et al. 1999).

cited examples of the considerable influence the Bank wielded in selling not just its loans but its particular ideas on development policy (Momani and Kramarz forthcoming). When these policies turn out to be wrong, it would seem convenient for the World Bank to speak of partners in development and focus on the recipient side of the equation questioning their commitment to good governance.

During the 1980s and into the 1990s, Bank aid focused strongly on conditionality. Countries struggling to meet debt obligations after over-borrowing during the 1970s petrodollar boom looked to the Bank and IMF for loans to service their debts. Developing countries accepted conditionality as the price of doing business with this development agency. However, the structural adjustment programmes prescribed as part of the loan conditions soon begun to draw heavy criticism from civil society for the social cost they exacted. The Bank came under attack from NGOs for large infrastructure projects that displaced people and caused environmental havoc. The most controversial examples were the Narmada Dam in India and the Polonoroeste development project in Brazil (Keck and Sikkink 1998).

It was in the midst of this backlash from civil society, the rise of good governance debates, diminishing official development assistance, increasing private capital flows to borrowing countries and poor project performance results publicized by the internal Wapenhans Report, that James Wolfensohn became president of the World Bank in 1995. The structural environment and the personal vision of the organization's new leader became mutually reinforcing forces, paving the way for a re-emergence of the Pearson Commission's discourse on partnership, ownership and participation (King and Mc Grath 2004).

With President Wolfensohn, partnerships became high-profile initiatives. He repeatedly invoked the approach as the new way of doing business and made it a centrepiece of his plans for institutional renewal. The Bank transformed its view of civil society and the private sector. While these actors were traditionally seen as rule takers, the Bank begun to refer to them as joint rule makers. The degree to which this view was internalized in meaningful ways needs to be further analyzed. The historical narrative that follows suggests a nuanced interpretation. There have been some important departures as well as continuities with previous conceptions of partnerships.

In his first annual meetings as president, James Wolfensohn's remarks to the Board of Governors were firmly grounded on the idea of partnerships. In a departure from the previous Pearson Commission conception of partners as donors and recipients, Wolfensohn emphasized the role of civil society and the private sector as partners. Throughout his 10 years in office, Wolfensohn's references to partnerships highlighted varying messages, most often focusing on democratization of development governance, innovation and raising the required financial resources to protect global public goods.

When speaking to developing countries, he often asserted that the Bank was becoming a better "listener" of its partners, showing respect and sharing with them authority to plan their development (Wolfensohn 2001). He repeatedly put emphasis on cooperation (Wolfensohn 2000a). To civil society, he offered acknowledgements that the World Bank had sat on a mountain with the IMF for 50 years and it was now changing and looking to join forces with civil society in meeting the challenges of development. (Wolfensohn 2000b). Internally, before the Board of Governors, he said: "To be a good partner, we must be ready to listen to criticism and respond to constructive comments" (Wolfensohn 1995, p. 12). During the 2002 Annual Meetings of the Bank and IMF, the President spoke to the Board of Governors of the three key dimensions of partnerships: inclusion, participation and empowerment.

In the Monterrey Conference on Financing for Development, President Wolfensohn referred back to the message of the Pearson Commission. He said to other financial institutions that partnerships would mean a commitment in which developing countries would pursue good governance and developed countries respond with the needed resources (Wolfensohn and Kircher 2005). Wolfensohn referred to this agreement between donors and recipient countries as partnerships in several speeches, memos and media interviews. This conception of partnerships as a bargain between developing and developed countries bears striking resemblance to the ideas of the 1969 Pearson Commission. This shows important instances of continuity with traditional conceptions of partnerships as a contract between equals, ignoring the asymmetrical negotiating position of recipient countries. Wolfensohn took great pains to point out that the developing world's commitment to better governance, including judicial reform, better financial systems and fight against corruption, was not being imposed from the top-down (Wolfensohn and Kircher 2005). This time, the Third World had freely chosen its development priorities; this is what made the donor-recipient relationship a partnership and renewed the legitimacy of each actor's commitments. It remains a matter of interpretation if Wolfensohn believed that donors and recipients stood on equal footing to negotiate development assistance as partners, or by speaking in these terms, he attempted to cajole other donors to steer away from the traditional hierarchical relationships with recipient governments of developing countries.

When speaking to Bank staff, Wolfensohn signalled the financial impact of private sector partners on development. He reasoned that while private sector finance for development was half the size of overseas development assistance (ODA) in the 1980s, it had grown to six times the size of ODA by 2000 in terms of dollar volume. "The involvement of the private sector is, of course, essential, and so as a source of partnership in terms of our own activities, the volume has risen dramatically" (Wolfensohn 2001).

When President Wolfowitz arrived in 2005, his endorsement of the partnership model paled in comparison to that of the previous administration. In a climate change conference, he wondered during a speech if public-private partnerships, as a model, could help promote climate-friendly technologies (Wolfowitz 2007). His anti-corruption agenda dominated his tenure, and he drew heavy criticism within the Bank for appointing a team of outsiders to conduct evaluations of corruption surrounding Bank projects. Taking what was deemed an outsider and hierarchical approach to internal management decisions, it is unsurprising that Wolfowitz would not rely on the language of partnerships, inclusion or participation in decision-making. President Wolfowitz's few references to partners concerned developing countries or multilateral development organizations, a move back and away from the Wolfensohn conception of civil society and business actors as key partners in the provision of global public goods.

When Robert Zoellick became president of the Bank in 2007, a renewed vision of partnerships emerged. In a widely discussed speech in development circles, entitled "Democratizing Development Economics," President Zoellick referred to partnering in terms of inclusion. He stated that partners are vital sources of resources and knowledge. No longer would aid prescriptions flow from the Bank in a unidirectional way. Zoellick's message in the eve of the 2010 Annual Meetings was on democratizing development and recognition of partners' roles in that enterprise (Zoellick 2010a). However, the image of the Bank that emerges in Zoellick's speeches is more restrained than that apparent in Wolfensohn's statements. The Bank tends to be described more as a connector of actors, rather than a coordinator.

The foreign aid regime has grown in size and complexity in the last two decades. Many development institutions have emerged since President Wolfensohn left office. Today, municipalities in developed countries provide direct technical assistance to counterparts in the developing world, emerging economies have become major bilateral donors, private actors in developing countries extend commercial loans to governments everywhere, and manufacturing companies in developed countries invest directly in development projects for emission trading credits. Within this heavily populated constellation of development options, President Zoellick has been at pains to redefine the role of the World Bank. In what he termed a new multilateralism, Zoellick referred to the Bank as a connecting tissue over a large sprawl of institutions and individuals who have power to influence global governance. He likens the new multilateralism to the sprawl of the Internet and contrasts it with old multilateralism where a club of developed nation states made the key economic decisions that would impact the life of most of the world's populations. During a speech at the Woodrow Wilson Center for International Scholars, Zoellick said, "Woodrow Wilson wished for a League of Nations. We need a League of Networks" (Zoellick 2010b).

The wider conception of partners as public and private actors is again prevalent in Bank literature, and partners are identified as valuable sources of new ideas. In a recent speech, Zoellick affirmed: "Donors need to be more flexible and innovative in creating partnerships with new players. I'm delighted that we have so many foundations and civil society groups joining this effort. I found them to be extremely useful in prodding, thinking, and trying to come up with new ideas" (Zoellick 2008).

3.4 The Substantive Promise of Partnerships

To assess the substantive significance of partnerships, I propose starting with an inductive framing of the issue; this entails taking the Bank's justifications for pursuing partnerships as the point of departure.[8] As is evident from the previous section,

[8] It is important to note at this point that the growth of global partnerships within the Bank has not been uncontested. The Committee on Development Effectiveness, one of the five standing committees of the Board of Executive Directors, stated that some directors believe the Bank is involved in too many global partnerships and "may be spreading its resources too thinly and losing the focus on its main mission" (World Bank 2004, p. 248).

the Bank has advanced three general rationales for promoting partnerships. First, partnerships are institutional arrangements that facilitate policy *innovation*. Second, partnerships *democratize* decision-making. Third, partnerships catalyze the needed *financial resources* for the provision of global public goods. Does the mantra of innovation result in the adoption of new biodiversity policies that enact new governance norms, and is the Bank used to socialize and legitimate the normative order advanced by their non-state partners? Do partnerships enable democratic processes in global environmental governance? And do they bring together financial actors to address global problems in ways that could not be done by multilateral governance as represented by the limited gains of traditional Bank projects in biodiversity conservation?[9]

These questions have direct local and global implications. Locally, this research sheds light on the values and approaches that partnerships privilege and financing that is made available to promote a given vision of conservation. This brings us closer to understanding who benefits, who decides and what nature is protected. In many cases, the stakes are particularly high for local communities whose autonomy, control over local resources and livelihoods are affected by the quest to protect the global commons (Molnar et al. 2007). Globally, this research can test the relevance of partnerships as substantive alternatives to the conservation work that is already being done under multilateral governance. Furthermore, this research can contribute to a growing literature on the influence of transnational actors in shaping Bank conservation policies (Domask 2003; Park 2009).

3.5 Policy Space: Do Partnerships Promote New Ideas?

3.5.1 States, Markets and Communities

The dominant policy approach in global biodiversity governance has involved regulation to ensure humans have limited impact on the wild (Wilshusen et al. 2002). The belief that underlies this approach is that nature is best preserved when the state builds fences to keep wildlife in and people out. Hence, the most often-used policy tool is creating protected areas. This is dubbed "fortress conservation." Most Bank partnerships finance activities that include creating and strengthening protected areas, developing economic activities in buffer zones around parks to keep communities from encroaching on park, developing scientific research for coding and

[9] It may be reasonable to expect that framing the biodiversity problem and attempting to address it in new ways through transparent, accountable and participatory processes with additional financing may be inputs for potentially effective partnerships. However, this research is not directly focused on partnership effectiveness but on the substance of policies, norms and processes that partnerships promote and the financial sources and volume they add to multilateral interventions for biodiversity protection.

organizing nature to facilitate its control and administration (i.e. geographic information systems, hotspots or ecosystem maps, taxonomies, and stocktaking and monitoring) (World Bank 2008).

Centralization and a focus on compliance are defining characteristics of this type of conservation approach. The state is seen as the appropriate authority in conservation, and this is evident in the financing dedicated to institutional strengthening for state control over the territory and its resources. One way to recognize policy innovation in conservation is assessing how often partners implement projects that go beyond this traditional "command and control" approach. This practice has historical roots in the United States and has created significant controversy on how it has been applied in the developing world where most of the existing biodiversity is concentrated.

It has been argued that this policy choice is based on the American experience, John Muir's legacy and the model of Yellowstone National Park—the first of the United States' successful parks system (Wilshusen et al. 2002; Lewis 2003). Muir's principle, on which Yellowstone was created, was of a wilderness that is set aside and separated from people. This became national policy in the 1964 Wilderness Act, which defines wilderness as a place "where man himself is a visitor who does not remain" (United States Congress 1964).

The Yellowstone model has proved impractical for different reasons in the developing world. Critics have argued that poverty in rural communities pushes people to exploit the natural resources that surround them, and neither fences nor park guards will keep people from trying to secure their livelihoods. Community conservation advocates argue that the people who have lived among biodiversity for generations have often been nature's best stewards by reserving areas of forest for conservation, rotating crops for better maintenance of soil fertility, etc. (Ostrom 1994). Building fences around parks not only disrupts rural livelihoods but also denies local communities recognition of the traditional work they have provided to the global commons by maintaining their ecosystems (Molnar et al. 2007). Dowie also argues that command and control policies have created millions of conservation refugees (Dowie 2006).

Given the deeply problematic outcomes of traditional conservation policies, partnerships with the private sector hold the promise of a much-needed revision. There are at least two broad challengers to traditional conservation; each relies on different sources of authority, maintains its own discourses and is enacted through specific types of project interventions (Kramarz 2008).

The first is guided by the belief that the best way to preserve nature is to commoditize it; attaching a price to its services will ultimately save it from further deterioration. The market is recognized as the appropriate authority to regulate actor behaviour. Some of the references to this norm in the discourse are decentralization, internalization of externalities, self-financing and efficiency gains. In general, project activities can include nature tourism, payment for ecosystem services and privatization of protected areas. It is the presumption that business actors will favour this approach to environmental governance that has mobilized much of the controversy surrounding public-private partnerships (Richter 2004; McAfee 1999).

The second alternative is community-based stewardship. It is a critical approach to traditional conservation by states and markets and is driven by the norm that conservation ought to be reconciled with local human needs. The emphasis is on rights and livelihoods. Projects focus on recognizing cultural claims on resources, enabling the participation of communities in political decisions on the environment and creating property rights that secure a community's interest in maintaining their resources (Western et al. 1994). "Community" is the locus of authority on conservation decisions, although there has been a critique on what community entails. Agrawal and Gibson have argued that the term "community as a small spatial unit, as a homogeneous social structure, and as shared norms" are the most conventional references in the literature (Agrawal and Gibson 1999, p. 629). They advocate instead a focus away from these assumptions about "community" and towards institutions that enable local management of resources.

There is no theoretical reason to assume that public and private partners are simply involved in a functional task of coordinated action towards a shared purpose, rather than as political actors, with distinct ideas, and power to promote them. Their definition of the biodiversity problem, ideas of who is the appropriate authority to govern and what are the appropriate responses need not be either similar or compatible.

For example, rather than homogenizing the NGO sector, one study identified 21 different conceptions of biodiversity conservation, each demonstrating conflicting views on what is the target of environmental protection (human or only biological species), where conservation should be focused (locally or globally) and how decisions should be made (e.g. according to the most efficient use of resources, international recognition of sites worth conserving or extent to which they can ensure benefits to people) (Redford et al. 2003).

If there are such fundamental cleavages within the NGO community alone, then partnerships between more disparate actors (from civic, state and market spheres) suppose an even greater level of normative divergence.[10] In other words, there is room to expect divergence in views among partners and variance in the types of projects financed by different partnerships. This may be a strength of the partnership approach; the plurality of interests may result in a larger repertoire of policy scripts that respond to broader global constituencies. In this vein, partnerships may be a perfect arena for innovation. Yet they may also be used to re-entrench existing conservation approaches under the guise of a fashionable new procedural recipe. This is the "old wine in new bottles" argument on partnerships (Melber 2002).

[10] There is a deep divide in the domestic, public administration literature between authors who question whether public and private spheres should ever be combined in mixed governance arrangements, or if their respective values, goals and motivations ought to keep them always functioning separately (Box 1999).

3.6 Inclusion: Do Partnerships Democratize Decision-Making?

Since the early 1990s, multi-stakeholder governance has been growing as a practical phenomenon and been reflected in the IR literature. There has been extensive debate among scholars and policymakers on the significance of a weakened state and of groups of undemocratically elected private actors gaining rule-making authority that potentially further weaken the state system. Some have wondered, if governance were to become a "franchise," who would hold business and civil society organizations accountable and how could this be done without a hierarchy of authority? (Levy and Andonova 2003). The announcement of 200 type 2 partnerships after the Johannesburg Summit confirmed many critics' fears and advocates' hopes that the world was moving away from the traditional, state-centred model described by realist conceptions of IR and towards a new paradigm of transnational relations (Elsig and Amaric 2008).

"Are partnerships the new multilateralism*?* Yes—partnerships create a web of relationships and programs, bringing in new players and innovative proposals," affirms a PowerPoint presentation by the Bank's office for Global Partnerships and Trust Fund Policy (World Bank 2010). Partnerships have been deemed necessary legitimating arrangements. An internal evaluation on the prospects for partnerships states: "Various Bank documents since then [2000] have emphasized that the provision of global public goods *requires* partnerships [emphasis in original document] to increase the legitimacy of traditional international organizations and to engage the perspectives and expertise of other stakeholders" (World Bank 2002, p. 2). The Bank also states that partnerships are driven by common responses to global problems in ways that are "transcending old power relationships" (World Bank 2010). The Bank's emphasis has often been specific on aspects of democracy that can be achieved through partnerships. "If we embrace a comprehensive approach, working in partnership with governments, and if we achieve this participation, this equity, and this inclusion, then we will have democratized development" (Wolfensohn 2000a).

Public actors are the traditional suppliers and private ones the consumers of the decisions made by those governing. Partnerships break down that divide by putting both public and private actors as joint decision-makers formulating rules for the world (Barnett and Finnemore 2004). Hence, partnerships that set up public and private actors as co-governors of global public goods are seen as institutional innovations to achieve more inclusiveness, representation and participation. This rationale highlights the power-sharing component of partnerships. Advocates of privatization have argued that it empowers consumers because it devolves power downwards from governments to individuals. Partnerships, instead, redirect power horizontally and give government, civil society and private sector an arrangement for sharing responsibility, knowledge and risk. In cases of regulation and compliance with environmental standards, partnering takes the adversarial edge away from the government-business relationship (Linder 1999).

However, the presumption that inviting for profit and non-profit actors to global policy making leads to greater democracy has been challenged. Börzel and Risse have taken up this debate and concluded that further empirical evidence is required to assess if and when partnerships may serve as vehicles to greater participation and accountability in global governance (Börzel and Risse 2005).

Conceptions of what constitutes democratic decision-making evolve over time, as norms and values change. Studies of democratic decision-making in natural resource management have used a wide array of democratic measures, which limits conceptual agreement. Defining what constitutes democratic decision-making is sensitive to interpretation by actors in particular contexts. A United States conception based on Jeffersonian ideals is said to be dominant at the Bank, but this represents a limited worldview on democratic values (Kenney 2000). How do developing country institutions define democratic governance? Those inputs need to be better incorporated in the Bank's conceptualization of democratic decision-making.

With these limitations in mind, I refer to three key general components of democracy that the Bank advocates and are adapted from studies in an extensive study of collaborative management in natural resources by William Leach (2006). These are inclusiveness, representativeness and empowerment. Inclusiveness is an important initial measure of democracy because it tells us who is allowed to participate in a given process. Representativeness is also a key dimension of democracy because it describes whose interests are represented at the governance table. This requires evaluating how partnerships are governed. Empowerment measures the extent to which partners are able to bring their preferences to bear on the partnership. To assess the substantive effect of partnerships on democracy, these three fundamental dimensions need to be considered.

Empirical assessments of partnerships offer a word of caution against too much optimism on the democratic potential of these governance mechanisms. Smith et al. argue that priority areas for conservation are often designated by international NGOs and academics without regard for who will implement activities there; consequently, biodiversity effectiveness suffers and democracy in conservation governance is eluded (Smith et al. 2009). Ribot shows how internationally led decentralization initiatives, where locals are supposed to lead, often result in international institutions, making all the decisions and the locals being assigned the implementation work on the ground (Ribot 2004). Large-N studies confirm these findings regarding the limited empowerment of new actors in global governance through the partnership mechanism (Biermann et al. 2007). As all these authors suggest, the promise of democracy needs thoughtful reevaluation.

3.7 Financing: Do Partnerships Catalyze Financing for Biodiversity?

The need for resources beyond traditional ODA became widely referenced after the announcement of the Millennium Development Goals (MDGs) and the report of the High-Level Panel on Financing for Development, headed by Ernesto Zedillo, which announced shortfalls in the resources required to meet the MDGs (Zedillo 2001).

Partnerships can be fundraisers. The financial arrangements that emerge from partnerships are frequently identified as innovation. For example, partnerships like the International Financial Facility for Immunization have been used to front load official development assistance (ODA) by selling bonds against future grant commitments from traditional multilateral or bilateral donors (International Financial Facility for Immunization 2011).

Partnerships with the business sector have leveraged private financial resources to deliver public goods. The public side offers risk guarantees in exchange for private-side investment. One estimate states, "Bank Group-supported PPP mechanisms registered leveraging ratios of 6.7 through Bank Group partial risk guarantees and insurance schemes, 7.5 through partial credit guarantees, and 5.6 on debt buy-downs using private donations" (Girishankar 2009, p. 9). These types of partnerships have become well established in the Bank's financial, extractive and infrastructure sectors, with the latter including provision of water, roads and health systems. However, the same has not been the case with biodiversity conservation partnerships.

Partnerships are conceived as "an instrument of choice" because of their potential to catalyze financial resources from a number of actors (World Bank 2010). Wolfensohn articulated this succinctly in several speeches by referring to the ratio of private sector financing to ODA. Since the 1990s when he assumed the presidency of the Bank, ODA has become dwarfed by private investment. Private giving has also risen. Foundations, NGOs and other civil society organizations from OECD countries financed $40 billion worth of development initiatives in 2006, which was 50% of official ODA to lower- and middle-income countries that year (Girishankar 2009). Hence, joining forces with firms, NGOs and foundations supposes greater impact and harmonization of initiatives under a common financial umbrella.

Critics have argued that the opposite is true. Partnerships can fragment or duplicate efforts, and raise transaction costs (Girishankar 2009). This has been empirically supported by the Bank's evaluations of its global partnerships (World Bank 2004). Partnerships can then create a capricious hierarchy of geographical and thematic issues that does not reflect need but the outcome of atomized efforts. Also, partnerships are not inexpensive arrangements to manage. In 2001, the Bank devoted $30 million of its administrative budget to global programmes and partnerships (World Bank 2004).

In analyzing the financial component of partnerships, the question that needs to be examined is not only whether these raise money for conservation beyond traditional public funds but also if funding creates incentives for the selection of certain partners and types of partnerships; what local versus global resources are mobilized; which actors in the private sector contribute and why; and what conservation approaches receive financing and which do not.

3.8 Conclusion

The objective of this chapter was to examine the growing phenomenon of public-private partnerships as an instrument of choice in global environmental governance. The argument it advanced is that the norm of partnering needs to be evaluated on procedural and substantive terms. I suggest that partnerships understood as constitutive or procedural norms have become well established in global environmental governance. Yet the substantive significance of the norm, that is, its regulative effect, is still underdeveloped in theory and practice.

The meaning of what it is to be a partner and what is the rationale for forming partnerships has evolved since the Bank's first references to this concept in 1969. Since James Wolfensohn's presidency, the World Bank has advocated partnerships based on a set of good governance values. The organization states that partnerships are ideal mechanisms to realize innovation, democracy and financial benefits in the provision of global public goods. Hence, these co-governance arrangements are not ends in themselves, but means to achieve higher order governance values.

Yet, there is evidence that calls into question the extent to which Bank partnerships are achieving these values. In biodiversity, it is not clear that partnerships actually create policy space for new actors to promote alternative conservation strategies. Likewise, the proposition that partnerships can improve inclusion, representation and empowerment of new actors in global environmental governance has also been challenged by large-N and focused case studies. Finally, partnerships may not leverage the financial resources in biodiversity conservation that the Bank envisioned, and may instead raise transaction costs.

If form follows function, then the procedural norm of partnerships needs to be more carefully aligned to produce the substantive goals for which they were set up: incubating policy innovation, empowering new actors to govern and promoting more efficient financing of global conservation.

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Chapter 4
Ecosystem Services and Payments for Environmental Services: Two Sides of the Same Coin?

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Abbreviations

BBOP	The Business and Biodiversity Offsets Programme
CBD	Convention on Biological Diversity
CCT	Centro Cientifico Tropical
CGIAR	Consultative Group on International Agricultural Research
FAO	Food and Agriculture Organization
GBA	Global Biodiversity Assessment

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GCTE	Global Change and Terrestrial Ecosystems
GEF	Global Environment Facility
ICSU	International Council for Science
IGBP	International Geosphere-Biosphere Programme
IIED	International Institute for Environment and Development
IMoSEB	International Mechanism for Scientific Expertise on Biodiversity
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISEE	International Society for Ecological Economics
MA	Millennium Ecosystem Assessment
MBI	Market-Based Instruments
PESP	Payments for Environmental Services Programme
SBSTTA	Subsidiary Body on Scientific Technical and Technological Advice
TEEB	The Economics of Ecosystems and Biodiversity
UNCCD	United Nations Convention to Combat Desertification
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNEP-FI	United Nations Environment Programme Finance Initiative
UNFCCC	United Nations Framework Convention on Climate Change
WBCSD	World Business Council for Sustainable Development
WRI	World Resources Institute
WWF	World Wildlife Fund

4.1 Introduction

The topic of ecosystem services, ecological services, environmental services (ES) and payments for environmental services (PES) has recently become the main reference for international environmental policies (broadly including forest policy, agro-environmental measures and conservation policies). Brought to media attention by the Millennium Ecosystem Assessment (MA) in 2005, these notions have spread rapidly in both political and scientific arenas. But there has been very little analysis retracing the social construction and political scope of these concepts in the scientific and policy fields. It is as if thinking in terms of ecosystem services and promoting payments for environmental services were taken for granted. This chapter seeks to fill this gap, offering a historical and institutional analysis that explores the relationship between the ES and PES concepts.

Recent studies on the history of ES and PES show that, conceptually, PES has emerged within a global process of commodification as a policy instrument intimately linked to the notion of ES (Gómez-Baggethun et al. 2010). PES framework is understood as a consequence of the MA process in mainstreaming ES into conservation and environmental policy (Redford and Adams 2009). We put forward the hypothesis that two relatively independent and contemporary processes, at least during the 1990 decade, led to the emergence of the ES concept on one hand and the PES concept on the other. Whereas the concept of ES is closely linked to a desire to attract official attention to the threats to ecosystems posed by human pressure, the concept of PES seems rather to have stemmed from a concern to ensure funding for conservation in tropical countries over the long term (Landell-Mills and Porras 2002; Wunder 2005). In the past few years, the two concepts have gradually converged, apparently due to a shared desire to translate them into operational form through public policy instruments.

Taking a multidisciplinary approach combining political science, sociology, economics and law,[1] we aim to substantiate this hypothesis using the notion of an *epistemic configuration*, derived from that of an *epistemic community* (Haas 1992). For Haas, a transnational epistemic community is a network of professionals of recognised expertise and knowledge in a particular field, wielding authority by virtue of the "relevant" knowledge it can call upon for the makers of policy in that field. This notion has at times been criticised for suggesting a relatively close-knit, homogeneous group. In our case study, the emergence and spread of the ES and PES concepts have resulted from the work of brokers, mediators, and entrepreneurs who have linked together very different types of groups and networks with a view to influencing political processes (Roberts and King 1991; Nay and Smith 2002). They are mainly scientists working on environmental issues, conservation NGOs and experts who favour the introduction of new systems for funding conservation, corporations, intergovernmental organisations such as the World Bank and FAO and also some governments.

Given the wide diversity of groups and organisations involved, we prefer to use the term *epistemic configuration* to highlight the composite nature of these groupings and the fact that they fluctuate over time. We began by testing the concept of an epistemic configuration in an ongoing work on the experts and scientists involved in the Millennium Assessment (MA) (Pesche 2011). It is similar to the concepts of "distributed research, assessment, and decision support system" and "polycentric network of semi-autonomous research nodes" that have been used elsewhere to describe forms of interaction between scientists and decision-makers involved in global environmental evaluations (Cash 2000; Cash and Clark 2001). As we shall see, global assessments are key moments when new concepts emerge and gain international acceptance at a rapid pace. This is clearly the case for the MA with the notions of ES. These close interactions between science and policymaking in environmental matters can be analysed as a process of co-construction of knowledge whose impact can only be grasped if it is thoroughly understood (Jasanoff 2004).

Epistemic configuration and co-construction allow us to better understand the work of building the new concept of ES, by combining scientific and expert knowledge with political and social beliefs, leading to the creation of new perspectives for governing nature. In this chapter, PES is analysed as a new instrument promoted

[1] This article draws on a number of ongoing studies conducted by the Serena programme, which receives funding from the Agence Nationale de la Recherche under the SYSTERRA programme (ANR-08-STRA-13) http://www.serena-anr.org/

by a more restricted epistemic community that is focused on tropical forest conservation policies. In both situations, scientists are not the only key players. As Jenny C. Stephens et al. show in their study, the "Carbon capture and storage community" includes scientific and technical experts, as well as representatives from business, government, academia and non-governmental organisations (Stephens et al. 2011).

In the first section below, we study the emergence of the concept of ecosystem services and the development of PES. We show how the two terms are produced by different epistemic configurations. In the second section, we show how the two terms have converged at the international level, during the MA process but mainly afterwards. In the third section, we try to identify new trends and ongoing processes concerning ES and PES.

4.2 ES and PES: Distinct Origins

4.2.1 Ecosystem Services: Science and Policy Before MA

Historically, the emergence of the ES concept is an integral part of the history of biology and ecology in the United States, particularly through the trail-blazing work of Marsh and Leopold (Mooney and Ehrlich 1997). But not until the 1970s did the term ES appear explicitly in the literature, after the idea of an "environmental service" was used in a 1970 report written in preparation for the first world summit on environmental issues, held in Stockholm in 1972 (Report of the Study of Critical Environmental Problems SCEP 1970). For those authors, the notion of an environmental service referred directly to the ecological functions provided by ecosystems. It was not then a question of conserving those services or the ecosystems concerned but of reducing the negative impacts of human activity. This first appearance of the ES concept, it must be stressed, took place in a setting where scientific knowledge was linked with a desire to formulate recommendations for official action.

In science circles, biologists and natural scientists were the first to use the notion of ecosystem services (Ehrlich and Mooney 1983). In continuity with the work begun in the 1970s, it served to put forward the idea that the services rendered by nature are poorly known and insufficiently taken into account. These scientists, who were active in environmentalist science circles, were keen to advocate nature conservancy. Gretchen Daily's 1997 book marked a turning point in the recognition of this concept in academic circles (Daily 1997). For them, the notion of ecosystem services was closely linked to more general research on biodiversity issues.

On the economics side, the notion of ES is rooted in the first studies that sought to bring environmental issues into economic analysis in the 1970s. In a context deeply marked by the publication of the Meadows Report (*The Limits to Growth*) and the first oil crisis in 1973, the emerging ecological economics movement produced a series of modelling exercises aimed at alerting public opinion. Another source lay in the energy analyses first introduced by Odum; these were to structure the heterodox current in environmental economics, which argued against the reductionism of monetary valuation. A third approach (so-called London School) made its name in the early 1990s, first in the lead-up to the Rio Conference and then with the development of biodiversity economics (Gómez-Baggethun et al. 2010). In this way, ecological economics developed in a somewhat hybrid context, on the one hand drawing on systemics, energy studies, complexity and a long-term approach and on the other applying one-dimensional, atemporal monetary valuation (Meral 2010).

An example of this ecological economics approach was the 1997 paper by Robert Costanza and his colleagues proposing monetary valuation of ecosystems worldwide (Costanza et al. 1997). This chapter had a considerable impact, both through media coverage of its findings and through the controversies and discussions it provoked within the scientific community.

By the late 1990s, the notion of ES had achieved widespread recognition in several currents of thought. Put forward by noted biologists and ecologists and in ecological economics papers, it remained a specialists' concept, largely unknown outside the circles concerned with environmental questions. This first convergence between scientists from different disciplines and currents around ES can be seen as the start of the construction of the early segments of an epistemic configuration, connecting together biologists and economists. From 1991 onwards, a number of scientists determined to influence decisionmakers took upon themselves to build a 'global' ecology. For instance, The Ecology Society of America, with its Sustainable Biosphere Initiative (SBI), aimed to set off "a significant increase in interdisciplinary interactions that link ecologists with the broad scientific community, with mass media and educational organizations, and with policy-makers and resource-managers in all sectors of society (...) There were during this period a number of efforts made to develop programmes closely related to the SBI's research agenda, both with intergovernmental (e.g. UNESCO) and non-governmental (e.g. International Council of Scientific Unions ICSU) International bodies. (Lubchenco et al. 1991).

Despite its heterogeneous nature, this configuration held some ideas in common: that the notion of ES as such should be recognised and that it was important to analyse the relations between the degradation of ES and human well-being. Its members shared the conviction that highlighting the idea that ecosystems render services was a strong argument for changing decision-makers' thinking with regard to the growing environmental degradation which by then was receiving increasing attention from the media.

4.2.2 Payments for ES and the Issue of Financing Nature Reserves

The notion of payments for environmental services (PES) arose from changes in perceptions of the efficacy of conservation policies in developing countries with high levels of biodiversity.[2] There were more operationally active and committed

[2] There are numerous publications on the PES subject. See for instance Engel et al. (2008), Muradian et al. (2010), and Farley and Costanza (2010).

stakeholders involved in the emergence of the PES concept than it was the case with ES. The concept emerged, in part, through several interconnected developments, as follows.

Firstly, during the 1990s, policymakers had favoured policies of conservation through development. The ICDPs (Integrated Conservation and Development Projects) were based on the development of income-generating activities that enabled farmers to earn more from pro-environment activities than from over-exploiting their environment. From the late 1990s, several authors questioned this type of intervention and suggested direct payments (Ferraro and Kiss 2002; Simpson 2004). At a symposium of the Society for Conservation Biology, several case studies of direct payments were presented.[3] ICDPs were described as "conservation by distraction," a term that Franz Tattenbach, director of FUNDECOR (Costa Rica), suggested to Ferraro and Kiss (note the parallel with the initials CBD for Convention on Biological Diversity) (Ferraro and Kiss 2002). Ferraro and Kiss (2002, pp.17–18) sum up this current of thought neatly: "After decades of global efforts to conserve biodiversity through indirect approaches, there is a growing recognition that such initiatives rarely work (…). The conservation community must reconsider its attempts to provide biodiversity through indirect means. If we want to get what we pay for, we must start tying our investments directly to our goals."

Secondly, the 2002 book by N. Landell-Mills and T. Porras was the first to make a systematic connection between environmental services and biodiversity markets (Landell-Mills and Porras 2002). The book identified all market-based instruments according to four environmental services (carbon, biodiversity, watersheds, recreation), thus introducing in the economic literature the classification into four services adopted in the 1996 Costa Rican law (see Sect. 4.2.3 below).

Thirdly, the problem of financing protected areas also helped to promote PES. From the early 2000s, many conservation actors became aware of the lack of long-term funding for the global network of protected areas. The realisation that many of them existed only on paper and the desire to promote an ecosystem approach within the network were reflected in a search for innovative funding mechanisms and the development of networks aiming to spread "good practice" in matters of funding for protected areas: Ecosystem Market Place, Conservation Finance Alliance, etc. PES were seen as a tool that could capture "benefits beyond boundaries."[4]

These three factors speeded the emergence of the PES theme, which has since been institutionalized through a number of publications (Pagiola et al. 2002; Pagiola and Platais 2004). In 2005, Sven Wunder of CIFOR helped to give PES their canonical definition, adopted or commented on in many subsequent publications (Wunder 2005). The dynamics involved in the emergence of the PES concept look like the construction of an epistemic community, more homogeneous than the configuration

[3] Direct Payments as an Alternative Approach to Conservation Investment, meeting held in London in 2002 during the 16th Annual Meetings of the Society for Conservation Biology: http://www2.gsu.edu/~wwwcec/special/special.htm

[4] Fifth IUCN congress, Durban, 2003.

of actors involved in the emergence of the ES concept. Most of the promoters of PES belong to the fairly restricted circle of those working on conservation problems in intertropical forest areas.

4.2.3 Interactions Between ES and PES

The emergence of the ES and PES themes was a separate process, which led to the adoption of two different reference frames. The ES concept was used in the preparation of the MA to construct an analytical framework comprising four kinds of service: supply services, regulation services, cultural services and support services. The PES concept had more operational aims, and its promoters identified four services (carbon, biodiversity, watersheds and recreational services) that could be specifically remunerated. Bibliometric analysis shows that the two flows of ideas are separate. The work of Daily, Mooney and Ehrlich is not cited in reference publications on PES (Mayrand and Paquin 2004; Wunder 2005), and the problem of financing conservation is not mentioned in the book edited by Gretchen Daily (1997). The authors and journals are rarely the same. In terms of publications, Robert Costanza is the only name that appears in both streams.[5]

But the two dynamics are not entirely independent. There are several bridges between the two, and interactions appear particularly around the setting up of the payments for environmental services programme (PESP) in Costa Rica. In the early 1990s, the Costa Rican Tropical Science Centre (CCT), one of the oldest "scientific" NGOs,[6] conducted economic valuations of the country's national parks in collaboration with scientists, including Robert Costanza, and several top Costa Rican civil servants. Although the term "environmental service" was not used directly, the idea of setting a value on the environment in order to better take it into account in policy was clearly present. US-trained environmental economist Jaime Echeverría was involved in the process: in 1995, with WWF support, he published the first study of the economic value of a nature reserve. He was one of the advisors to environment minister René Castro for setting up the PESP in 1996; he then advised the governments of Panama and El Salvador on the design of similar systems.[7] The conceptualisation of the PES programme in Costa Rica was the result of closed contact between René Castro and international experts, involved in the promotion of economic instruments for environmental management and sustainable development like Theodore Panayotou from Harvard University (Panayotou 1994).

[5] Bibliometric analysis (WOS) on the terms "ecosystem services" and "payments for environmental services."

[6] The CCT was founded in 1962 by several US and Costa Rican scientists to study biodiversity and natural resource management. It was the CCT that initiated the creation of Monteverde, the oldest private nature reserve in Costa Rica.

[7] Interview with Jaime Echeverría, July 2009, Serena Programme.

At the beginning of the process, the World Bank was mainly a follower in PES implementation. In this period, the World Bank was strongly criticised by environmental NGOs for its weak integration of environmental concerns in lending programmes. After Rio (1992) and CBD implementation (1994), the Global Environment Facility (GEF) was created and located within the World Bank (Young 2002). It was politically important for the World Bank to demonstrate its capacity to invest money in new environmental programmes and activities. This was accomplished in a context of diversification of funding sources. Several World Bank's experts analysed the Costa Rican experience with the aim of creating a model for funding new environmental programmes.

Other scientists and research bodies facilitated the process of bridge building between ES and PES on the basis of the Costa Rican experience, such as, for instance, Bruce Aylward, who, after undergraduate studies in Human Biology at Stanford, had earned a Ph.D. in Environmental Economics. Working for the International Institute for Environment and Development (IIED) from 1990 to 1996, he conducted a number of surveys on valuing local ecosystems in Costa Rica and did more work in Costa Rica for the World Bank in the early 2000s. This work helped to make the Costa Rican experience more widely known in other Central American countries.[8]

In the late 1990s, his institution, the IIED, was very active in promoting market instruments for forest conservation and used the Costa Rican experience among others to illustrate its arguments (Landell-Mills and Porras 2002). Gretchen Daily, known for her key role in establishing the ES concept, maintained a range of professional relationships in Costa Rica, regularly citing that country's experiments in taxation and then PES to support her work of advocating recognition of the services provided by ecosystems.[9]

In 1994, the recently formed International Society for Ecological Economics (ISEE) held its 3rd world congress in Costa Rica: while the first two congresses had gathered some 400 participants, the San José congress brought in some 1,300, marking an acceleration in the growth of this international scholarly organisation (Røpke 2005).

Until the end of the 1990s, PES had made relatively little progress except in a few countries, mainly in Latin America. Some scientists promoting ES and working in tropical and forest areas already knew about the first experiments in PES instruments, and some were involved, even if indirectly. However, these links were still tenuous, and it was not until the early 2000s, in the wake of the Millennium Ecosystem Assessment (MA), that the pace accelerated in the promotion of ES and market instruments like PES and in the interactions between them.

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[8] These elements are from an in-depth biographical study of the experts and scientists involved in the Millennium Assessment (see below).http://oregonstate.edu/gradwater/sites/default/files/bio/aylward_0.pdf

[9] http://woods.stanford.edu/docs/news/gdaily-strategy.pdf

4.3 The Millennium Assessment: Towards International Recognition of the ES and PES Concepts

Since the 1992 Rio summit, global environmental changes have become a growing preoccupation, and global scientific assessments have become more and more frequent. These assessments, like the MA, can be regarded as a formal effort to assemble selected bodies of knowledge and make such syntheses publicly available in a form useful for decision-making (Mitchell et al. 2006). Mitchell et al. think that global environmental assessments should be seen as social processes rather than simply in terms of the documents they produce. For these authors, the influence of these assessment exercises lies more in their characteristics and the extent of the process, both during the assessment and after publication of the reports: "We came to see assessment as a social process, in which scientists, policymakers and others stakeholders are (or are not) gathering data, conducting analyses, explaining, debating, learning, and interacting with each other around the issue on which the assessment focuses. The process by which information is generated and delivered affects the potential of that information process to influence outcomes (…). We therefore shifted our focus from evaluating the influence of assessment reports to the influence of assessment processes. We began looking at assessment reports as simply one visible indicator of a larger social process that seemed to be the real source of any assessment's influence."

From this standpoint, analysis of the ES epistemic configuration, which was strengthened and expanded through the MA, gives a better grasp of the mechanisms of this process of international promotion of the ES concept (which was increasingly coupled with the idea of PES).

4.3.1 Genesis and Characteristics of the MA

While the MA formally took place from 2001 to 2005, preparations were already under way in late 1998. Another exercise in global biodiversity assessment had been carried out a few years earlier. The Global Biodiversity Assessment (GBA, 1993–1995), initiated by UNEP and supported by the Global Environment Facility (GEF), was intended as an independent scientific exercise to assess the state of the art on the many questions connected with the complex issue of biodiversity (Heywood 1995). The report made no recommendation for decision-makers. It simply identified the weak state of knowledge on biological diversity without addressing the public authorities; this limited its political impact (Watson 2005). Moreover, its legitimacy was limited by the fact that scientists and experts from southern countries were scarcely involved (Biermann 2001). The promoters of the MA had to set up a working framework in which scientists and NGO representatives would be closely associated to ensure that the analysis framework and the knowledge produced were jointly constructed from the outset. A preliminary exploratory committee, set up on the initiative of the World Resources Institute, World Bank and UNDP, operated from 1998 to 1999. Its composition was a good reflection of the nature of the process, with fairly balanced representation between well-known and respected scientists (some with political or institutional responsibilities), representatives of international organisations (WB, FAO, UNEP and UNDP) and more specifically environment-related international conventions (CBD, UNCCD, UNFCCC).

From 2000 onward, the MA process was managed by a Board that was a good illustration of the concern to mobilise different stakeholder categories around the issue of assessing ecosystem degradation; it had the same participant profiles as the exploratory committee, with the addition of representatives of governments, NGOs and the private sector.[10] This multi-actor aspect of the MA process gave it legitimacy and was undeniably a factor in disseminating the knowledge produced, via the various networks involved.

4.3.2 Connecting Numerous Subnetworks Involving Both Scientists and Decision-Makers

Altogether, the MA mobilised more than 1,360 "experts" around the world, but if we analyse the overlapping responsibilities and varied degrees of involvement, we find a group of about 30 people. This core group of the MA process can be regarded as the nucleus of an epistemic configuration comprised of actors with a range of profiles. Together they form a kind of tentative coalition, oriented by the implementation of the MA and anchored in a number of countries. This group was to play an active part in spreading the ideas of the MA after 2005.

Within the ES epistemic configuration, one can identify key people acting as mediators, members both of the MA's Board and its scientific bodies (assessment panel and editorial team). Among these brokers, we find pioneers of the ES approach (Harold A. Mooney, Angela Cropper), others who were closer to the PES epistemic community (P. S. Dasgupta) and scientists specialising in facilitation for processes of this kind (Robert Watson, Walter Reid). These people, at the interface between the international scientific and political arenas, connected four different subnetworks within the MA, which subsequently facilitated the rapid dissemination of the concept in various professional circles and social milieus.

One of these subnetworks comprised representatives of ecological economics (EE), the Beijer Institute and the biology department at Stanford. It reflected the scientific roots of the ES concept in a coming together of ecologists, biologists and environmental economists. Harold A. Mooney and his collaborators at Stanford had long been working on ES. In economics, as already mentioned, it was the ecological economics current that did much to develop the ES concept, especially through key

[10] The government representatives were also mobilised to revise the provisional versions of the reports and chapters of the MA. These facts are drawn from ongoing research (Serena programme: http://www.serena-anr.org/)

authors such as Robert Costanza and Rudolf de Groot (Gómez-Baggethun et al. 2010). This first subgroup highlights the origins of the notion of ES, its connections with PES and the institutional and symbolic resources that were to enable the ideas stemming from the MA to spread.

A second subnetwork was structured around two international scholarly institutions, the International Council for Science (ICSU) and Diversitas.[11] Both had run international research programmes during the 1990s, particularly the IGBP (International Geosphere-Biosphere Programme) launched in 1990 and the subprogramme GCTE (Global Change and Terrestrial Ecosystems) (Kwa 2005). What all those involved in these programmes were trying to do was to link ecosystem analysis with the problem of climate change, which was much in vogue internationally. These learned societies financed, ran and/or gave their seal of approval to research programmes, but they also wanted their results to have an impact on policy decisions. For example, in the 1990s, Diversitas dropped its quantitative approach of cataloguing species in the wild and turned to systemic analysis of ecosystems, which was more appropriate for dialogue with policymakers. It was this systemic approach that was developed in the MA and placed the notion of ES on a foundation of broader research making the connection with climate change. The research programmes launched by ICSU and Diversitas or given their seal of approval were both channels for recruiting scientific experts to the MA and networks capable of ensuring rapid dissemination of the concept in international scientific circles.

The scientific arenas of the CBD and especially the SBSTTA (Subsidiary Body on Scientific, Technical and Technological Advice) formed a third subgroup which was to ensure that the ES concept was disseminated in international conventions (see Sect. 4.2.3 below).

The fourth subnetwork comprised scientific experts from the World Bank, CGIAR and development circles more broadly. Some of them, like Stefano Pagiola, were also advocates of PES approaches. The presence of this fourth subnetwork reflects the broader trend of development circles and conservation circles drawing closer together (Marhane 2010; Young 2002).

These four subnetworks were not watertight. Some scientists were involved in several institutions or moved from one subgroup to another; in this way, the ideas promoted by the MA spread all the faster in the different networks. Their interconnection and the positive responses to the MA helped to extend and strengthen the epistemic configuration of people who argued that to improve human well-being, the services rendered by ecosystems have to be taken into account. The connections between these four subnetworks were forged by experienced people holding multiple positions, and who were to become the pillars of the dissemination work (Watson, Mooney, Cropper and Hamed Zakhri). Through its diverse branches and their links within the MA, this epistemic configuration thus helped to bring together the ES and PES concepts. By examining its work, we can better understand the way these two concepts have spread, and are still spreading, in a number of spheres.

[11] Diversitas is in a sense "the biodiversity branch" of ICSU.

4.3.3 Close Connections with International Conventions

The co-production of the ES concept was partly a result of the conjunction between scientific processes and the intergovernmental processes of international conventions on environmental issues. The emergence and rapid spread of the ES concept were part of a broader process of building an intergovernmental institutional framework to address environment issues. Since the early 1990s, this institutional architecture had been richly diverse but fragmented (Biermann et al. 2009).

A first trace of a concept close to ES in international law can be found as early as 1992 in the statement on forest principles adopted in Rio de Janeiro that year.[12] Principles 2 and 6 of the document refer to the "services" supplied by forests. Principle 6 recognises that "a comprehensive assessment of the economic and non-economic values of forest goods and services and of the environmental costs and benefits"[13] should be used when taking decisions about forests.

The gradual adoption of the ecosystem approach within the CBD can be regarded as a process that created favourable conditions for the emergence of the ES concept and its rapid adoption in the convention. The ecosystem approach was mentioned for the first time in 1995,[14] then in greater detail at the 5th CBD Conference of the Parties. The ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and equitable, sustainable use. Thus, the application of the ecosystem approach will help to achieve a balance of the convention's three goals: conservation, sustainable use and the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources.[15] The 5th principle of this approach states that "Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach." The ecosystem approach is intended to be a management approach combining the broad aims of the CBD: it establishes a favourable framework for introducing the ES concept into international arenas and then gradually into national contexts.

Representatives of UN agencies were closely associated with managing the MA process: they were well represented on the MA Board and in the executive committee

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[12] The exact name of this declaration is the Non-legally Binding Authoritative Statement of Principles for a Global Consensus on the Management, Conservation and Sustainable Development of all Types of Forests.

[13] Principle 6c of the forest declaration.

[14] The CBD's second (Jakarta, November 1995) adopted the idea of an ecosystem approach as the main framework for action under the convention but made no mention of ES.

[15] UNEP/CBD/COP 5/23, *Report of the Fifth Meeting of the Conference of the Parties to the Convention on Biological Diversity*, Nairobi. The descriptions and principles of the ecosystem approach, prepared by the SBSTTA, were adopted at the fifth COP meeting in Nairobi by decision V/6. They were then detailed from the standpoint of implementation by COP 7 in Kuala Lumpur in 2004 (decision VII/11).

that managed the process more closely, meeting several times a year.[16] Within the ES epistemic configuration, some of the scientists and experts involved in the MA process had hybrid profiles combining considerable scientific repute with an international career in the CBD, either as executive secretary (Angela Cropper, Calestous Juma, Hamdallah Zedan) or as chair of the SBSTTA (Peter J. Schei, A. H Zakri, Cristián Samper, Alfred Oteng-Yeboah, Christian Prip).[17]

In the view of the MA's promoters, the involvement of international conventions (mainly CBD and Ramsar) and intergovernmental organisations should, by extension, help to raise governments' awareness and influence policy. A peer review system involving experts and government representatives, aiming to get beyond the fragmentation of the international arenas on environmental issues, was a first step in socialising the analytical framework that incorporated the notion of ES. After 2005 and the publication of the global assessment reports, UNEP, the CBD and other UN agencies were important channels for the MA promoters' work of spreading the concept.

Although all the conventions associated with the MA process are connected with biodiversity conservation, they take different approaches. Many environmental law experts focus on the CBD's economic approach promoting market-based instruments (Kiss and Beurier 2010), while the conventions on conservation of migratory species and wetlands take the view that the environment should be protected for its intrinsic value. Highlighting the connection between the ES concept and these four conventions, as the MA did from the outset, made it easier for this scientific concept to find its way into various kinds of public policy.

4.3.4 ES and PES Draw Closer During the MA Process

During the MA, the ES and PES epistemic configurations drew closer. The private sector and NGOs were more sensitive to the operational side of PES. From the outset, the private sector actively supported the initiative via the Avina Foundation which became one of the MA's first financial partners. In the early 2000s, a few major multinationals were expressing increasing interest in market-based instruments for protecting biodiversity. Even before the end of the MA, some were particularly active in setting up PES mechanisms and market-based instruments more broadly (Lafarge and Rio Tinto, e.g., were leaders in the field of ecological compensation mechanisms).

[16] The members of the MA Board's Executive Committee are representatives of the CBD, CCD, Ramsar, UNEP and GEF and presidents or chairs of other MA functional bodies (UNEP. 2000. *Cooperation with the Global Biodiversity information facility (GBIF) and the Millennium Ecosystem Assessment*, UNEP/CBD/COP/5/INF/19, avril 2000, Nairobi). The more general organisation of the process and its relations with UN agencies are described in UNEP 2002. *Status of Implementation of the Millennium Ecosystem Assessment*, UNEP/GC.22/INF/27, Nairobi.

[17] Angela Cropper and A. H. Zakri were to play a front-line role in the MA process and its follow-up.

Jeffrey McNeely, chief scientist at IUCN, had been an active advocate of PES approaches since the 1990s (McNeely 1988) and played a decisive part in the MA, as a member of the exploratory committee and by taking part in drawing up the overarching synthesis report.

The increasing alignment between the promoters of PES and of ES was particularly noticeable during the drawing up of one of the MA synthesis reports, *Ecosystem and Human Well-Being: Opportunities and Challenges of Business and Industry*. Although they claim to approach the issue in terms of ES, the 16 authors seem mainly interested in the economic opportunities that a biodiversity market might offer. They do not attempt to spell out the conceptual links between ES and PES but focus mainly on the economic opportunities offered by a PES approach and particularly market-based instruments (MBI).

One of the contributors to the report was Stefano Pagiola, member of the World Bank's environment department and leading promoter of PES. He has published numerous papers on the PES theme in general and the Costa Rican experience in particular. His presence on the editorial board for the "ES and industry" report was a factor in aligning ES and PES instruments more closely in the MA. At the interface between the science world and the international arena, Stefano Pagiola has acted as a broker for the idea of PES.[18] In the late 1990s, he was working with Michael Jenkins, then senior forestry adviser to the World Bank. In 1998, he founded the NGO Forest Trends which was to give rise to a number of "incubator" organisations that favoured MBI, such as the Katoomba group (2000), BBOP (2005), Ecomarketplace, SpeciesBanking.com, ForestCarbonPortal.com and the Chesapeake Fund. Some of these incubators work to produce scientific data on PES (Katoomba). Pagiola and Wunder are involved. The findings are fed to the satellite organisations working to set up market-based instruments that particularly interest some companies and public authorities.

The careers of some of the MA report's authors do not only reflect the connections between the conservation economics world, the international political arena and the private sector (WBCSD, Rio Tinto, Unilever, etc.) but also facilitate the circulation ideas on PES instruments in different circles. They also reflect growing interactions between the ES and PES approaches within the MA process. These connections intensified during the MA because PES-type instruments could achieve more media coverage and reach other stakeholders who might be interested in the approach. The MA acted as a kind of echo chamber for both concepts and began the process of "mutual justification" between ES and PES that began in 2005.

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[18] S. Pagiola also participated upstream in designing the MA's analytical framework. Another link between Costa Rica's ESPP experience and the MA was in the person of José Maria Figueres, MA board member and former President of Costa Rica (1994–1998).

4.4 Politicisation of the ES and PES Concepts After the MA

The work of the ES epistemic configuration did not end with the publication of the MA reports in 2005. The "MA configuration" remained active and gradually changed to address the dissemination of the ideas developed during the MA. Interactions with the promoters of market-based instruments for ecosystem management also multiplied.

4.4.1 MA Follow-Up Process

A good deal of communication work was done around the publication of the MA reports. This also helped to promote the ES concept. In 2007, a consortium of partners was formed to follow up on the MA, with a secretariat run by UNEP[19] in collaboration with UNDP. A strategy and a road map were drawn up, its first aim being to continue the work of knowledge construction that the MA had begun and to develop tools for "mainstreaming ecosystem services into development and economic decision making" in order to integrate the MA's ES approach into decision-making at all levels.[20] A working group was set up to continue sub-global assessments, with a secretariat based at United Nations University/Institute for Advanced Studies (UNU/IAS) which works on issues related to the UNDP/UNEP Poverty and Environment Initiative. A multidisciplinary group of experts was formed to identify gaps in knowledge and draw up a research agenda (ICSU-UNESCO-UNU 2008). An ecosystem assessment manual for decision-makers was finalised in 2010 (Ash et al. 2010). Some of the scientists involved in the MA process continue to work with the ES concept with a view to identifying the boundaries of ecosystem research (Carpenter et al. 2006, 2009).

In the eyes of its promoters, the MA follow-up process was somewhat disrupted by a French consultation initiative launched in 2005 to establish an International Mechanism for Scientific Expertise on Biodiversity (IMoSEB), which concluded its work in 2007. But the two processes came together around the creation of an Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) under UNEP leadership. It is envisaged that this platform "will complement, among others, the scientific subsidiary bodies of the biodiversity- and ecosystem-related conventions and relevant intergovernmental bodies with the needed scientifically credible information on emerging issues in the science of biodiversity and ecosystem services" (UNEP 2008b). Being an intergovernmental body, it is seen by some scientists as less independent than the MA exercise. The way the IPBES is to

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[19] UNEP/DEPI (Department of Environmental Policy Implementation).

[20] The detailed MA follow-up activities programme was presented in 2008 at the 9th CDB Conference of the Parties (UNEP 2008a. *The Millennium Ecosystem Assessment Follow-up: A Global Strategy for Turning Knowledge into Action* UNEP/CBD/COP/9/INF/26, Nairobi).

function is still under discussion but, rather like the IPCC, it will be an important arena for the dissemination of the ES concept both internationally and in-country.

The IPBES does not mark the end of the MA follow-up process, and complementarities are sought, with the idea that the IPBES should concentrate on producing knowledge at the global and regional levels, and on scientific assessments, while the MA follow-up could focus on capacity building and policy implementation, particularly in developing countries (SwedBio 2010).

In terms of international law, it is mainly through documents adopted by the Conferences of the Parties that the environmental or ecosystem services concepts have emerged since 2005. Among international agreements, only the International Tropical Timber Agreement of 2006[21] recognises the importance of "environmental services" in its preamble and of "ecological services" in Article 1. Apart from that agreement, the first legal applications of the concept are to be found in decisions by the Conferences of the Parties. Logically enough, there are examples of its use in decisions by the four conventions that were involved in the MA as partners. For example, one of the goals of the CBD's action plan to 2020 is to "ensure the continued provision of ecosystem services."[22] Similarly, a resolution of the 10th Conference of the Parties to the Ramsar convention recommends (in Point 39) studying the concept of payments for ecosystem services.[23] The Conference of the Parties to the desertification convention, in its resolutions, also invites the participants to intensify actions aimed at maintaining ecosystem services.[24]

This gradual incorporation of the ES concept in international arenas and national policies has also been facilitated by the sometimes controversial tendency to put ES together with valuation of ES and the introduction of market mechanisms.

4.4.2 ES as Economic Justification for Environmental Policies

The two most significant contributions made by the MA are, on the one hand, the analytical framework (now firmly embedded in the literature and public policy) that incorporates the concept of ES; and, on the other hand, the wake-up call to the international community to draw its attention on the cost of doing nothing, as well as on the monetary value of ecosystems. The main event illustrating this trend is the emergence of The Economics of Ecosystems and Biodiversity (TEEB) initiative. Building on the conclusions of the MA, Pavan Sukhdev has proposed to continue to raise the

[21] This agreement, signed in Geneva in 2006, follows on from the agreements on trade in tropical timber. The timber industry had been promoting regulation of the tropical timber trade since 1983. One of the effects of this agreement was the creation of a permanent organisation, the International Tropical Timber Organisation.

[22] Revision and Updating of the Strategic Plan: Possible Outline and Elements of the New Strategic Plan, UNEP/CBD/SP/PREP/2, November 2009.

[23] Resolution X.24 of the 10th COP held in 2008 in Changwon, South Korea.

[24] Decision 4 of COP.8 held in Madrid in 2007.

alarm about the global loss of ES by valuing them in monetary terms. The first step (TEEB 2009) was to assess the state of knowledge on the monetary value of ecosystems, show how some policies run counter to sustainable management of ES and list the economic instruments best able to help conserve them. This list includes PES.

The use of ES in farming policy has also been a major trend in recent years. The FAO's (2007) annual report suggests a synthesis between ES and PES and recommends a distinction between *ecosystem services* and *environmental services*. The latter, derived from the theory of externalities, cover non-traded services and so exclude supply services. More interestingly, incorporating ES in the analyses leads to a completely different view of the positive externalities that agriculture provides. It is no longer the ecosystems that provide the services but economic actors (farmers) who through certain practices produce positive externalities that should be internalised by paying for them. In the same way, nature reserve managers use the rhetoric of ES to argue that they produce externalities that should be paid for in order to conserve biodiversity or watersheds. These examples illustrate how arguments based on the idea of PES have spread widely and rapidly after the MA' s publication. Although, as we have shown, PES were first developed as a way to fund conservation, particularly tropical forest conservation, the international audience captured by MA has made it possible to use the concept as a policy instrument more widely, regardless of a country's level of economic development.

The wide recognition of the ES concept means that today, there is an international audience for such instruments as compensation (mitigation/conservation banking, BBOP, etc.). While these instruments have long existed in the United States, internationalisation of the ES issue, thanks to the MA, means that today, there are numerous networks promoting these economic instruments in many countries (Madsen et al. 2010).

Compensation apart, the ES theme is also spreading in economic circles through initiatives supported by various institutions or networks. Examples are UNEP-FI, WRI, WBCSD and The Natural Value Initiative. The aim is to encourage industry to better identify their dependence on ES and the potential damage to their activities from the loss of ES. In 2008, the WRI, WBCSD and Meridian Institute through their Corporate Ecosystem Services Review initiative proposed a methodological guide for the private sector which also aims to help firms identify and control their dependence on ES. Similarly, in 2009, the Nature Value Initiative developed an Ecosystem Services Benchmark to encourage firms in the farming and food sector to identify their dependence on ES and the economic opportunities for incorporating these ES in their development strategies (Grigg et al. 2009).

4.5 Conclusion

The evolution of the science-policy interface on the issue of ES seems to follow the same trend as with climate change. Often presented as the biodiversity equivalent of the IPPC initiatives and the Stern report, the MA and later the TEEB have helped to steer policy on conservation and sustainable management of biodiversity towards various means of economic regulation. Just as climate change issues are today presented from the standpoint of tonnes CO_2, the international community in charge of the biodiversity agenda wants a means of measurement, a simple indicator that can be easily incorporated into public policy, practical provisions, corporate strategies and international institutions (CBD 2003; Godard 2005).[25] The ES concept seems to offer that possibility.

At first largely independent of practical experiments with PES-type instruments, the notion of ES is now firmly anchored within decision-making circles, where interest in market-based ecosystem management instruments is growing steadily.

These two notions developed in composite epistemic configurations that fluctuated over time and have been the locus of controversies that had previously been contained. Although there seems to be little at present to counterbalance the controversial tendency to regulate ecosystem services mainly through market-based mechanisms, it may be useful to monitor over the coming years how scientists and society accept it or reject it. The recent creation of IPBES and the debate on the relationship between scientific knowledge and other forms of knowledge on ecosystems (NGOs, local communities, etc.) constitute a major challenge. The current enthusiasm for setting up market-based instruments around biodiversity can only intensify that challenge.

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[25] See also the growing concern on indicators and measuring biodiversity (for instance UK parliament, http://www.parliament.uk/documents/post/postpn312.pdf, the private sector and NGOs: http://www.businessandbiodiversity.org/what_is_measuring.html; see also the third chapter of the TEEB on the use of indicators).

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Chapter 5
Property Rights and Government Involvement in Market-Like Biodiversity Conservation: An Empirical Analysis of Bioprospecting

Per M. Stromberg*, Unai Pascual, and Claudia Ituarte-Lima

5.1 Introduction

Genetic resources (GR) and the traditional knowledge (TK) about their use, for example, for traditional medicinal purposes, hold multiple values for society. They also form valuable inputs into basic research and development activities in the life science industry. However, the sustained conservation of nature that hosts GR, and the access to them, requires governance structures which involve clear property rights. It is in this context that the Convention on Biological Diversity (CBD) addresses bioprospecting projects in developing countries by aiming to provide conservation incentives under favourable conditions to biodiversity holders while facilitating GR access to external users.

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Much of the economic analyses on bioprospecting tend to focus on GR valuation and on theoretical assessments of the effect of the current international patent legislation, which is often criticised for giving firms only short-term incentives to invest in biodiversity (e.g. Goeschl and Swanson 2000; Rausser and Small 2000). Other research (e.g. Mulholland and Wilman 2003) has explored theoretical aspects of the functionality of different benefit sharing modalities in bioprospecting projects. Other analyses have addressed the relationship between bioprospecting outcomes and the jurisdictional governance setting (e.g. OECD 2003; SCBD 2008; UNU-IAS 2008). However, as highlighted by Polski (2005), there is a lack of empirical evidence about the performance of bioprospecting contracts, especially on the governance factors that influence bioprospecting schemes as framed by the CBD. One such understudied governance aspect is the nature of the contractual hazard involved in bioprospecting projects (SCBD 2008).

In this chapter, contractual hazard refers to the conditions that make a contract be interrupted or finished before the respective rights and responsibilities of the project are fulfilled. This notion has a non-evaluative connotation; that is, it is not associated with a normative evaluation of whether bioprospecting contracts as such are positive or negative in terms of conserving biodiversity, nor if they are an effective means for promoting fair and equitable allocation of rights and responsibilities between stakeholders. In fact, bioprospecting contracts, as market-based legal mechanisms, vary substantively because it is up to the parties to decide the content of each individual contract.[1] Among the many normative interpretations of what is a successful bioprospecting project (see e.g. Shiva 1997; ten Kate and Laird 1999), here we define a successful bioprospecting project more simply as a project that proceeds without cancellations or interruptions.

In this chapter, we cast new light on the link between different institutional designs of bioprospecting projects and the project outcomes. We analyse the main institutional roles of governments in terms of clarifying and enforcing property rights of GR. This type of analysis is carried out using standard concepts from institutional economics (e.g. Oxley 1999; Williamson 1985, 1999, 2005). The main idea or hypothesis that we hold here is that transaction costs associated with public policies to regulate bioprospecting might cause contractual hazard in such projects, which may bear negative effects on their outcomes.

We specify a theoretical framework based on the idea that there is likely to be trade-offs between having clear and enforceable property rights for biodiversity holders and the level of transaction costs associated with setting those property rights. We also pose that governments might under certain circumstances ease contractual hazards. In order to understand the link between government intervention and concrete outcomes of bioprospecting contracts, it is necessary to understand the role of government intervention within the overall context of the contractual project.

[1] Interruption of a contract and its subsequent renegotiation or even its premature termination may not be a failure but a success in terms of agreeing a more equitable allocation of rights and duties over the use of GR and associated TK and/or in a better adequate way of conserving biodiversity.

In this chapter, we explore empirically the role that the two mentioned government functions have on the overall contractual context of the projects to shed light on whether and the extent that governments can aspire to have a significant role in affecting bioprospecting outcomes. Here, we refer to the contractual context as the institutional conditions under which the parties negotiate the content of bioprospecting projects and implement them.

5.2 The Nagoya Protocol's Influence on Sovereignty and Property Rights

The entering into force in 1993 of the CBD was a critical event for rights claims over GR because it spread the debate of whether sovereignty implies property rights over GR including access and benefit sharing rights and obligations over these resources and associated knowledge.[2] In 1992, the CBD was opened for signature, and it has been ratified by 193 countries to date.[3] The CBD recognises the sovereign rights of states over GR and mentions that national governments have the authority to determine the access to or exclusion from GR through national legislation (CBD 1992 Article 15.1). The CBD, in its Article 15, entitled "Access to Genetic Resources," states: "1. Recognizing the *sovereign rights of States* over their natural resources, the authority to determine access to genetic resources rests with the national governments and is subject to national legislation" (italics added).

It is important to note what the CBD explicitly expresses in terms of sovereignty and property rights. The relationship between sovereignty over GR and property (which is not explicitly mentioned in the CBD) is often politically and academically contested (see e.g. Coombe 1998; UNEP 2005; Elvin-Lewis 2007; Caneiro-da-Cunha 2008). Sovereignty does not necessarily equate to property. Johnston (interview 21 January 2009) considers that the relationship between sovereignty and property implies a political exercise.[4] It is up to the countries to shape their own interpretation concerning sovereignty rights to GR under Article 15 of the CBD.

Countries have opted for three main approaches: first, some countries have signed and ratified the CBD but have not related the term sovereignty to property. A second approach has been chosen by several of the so-called developing countries which have actively engaged in its interpretation and implementation. These countries emphasise the states' sovereignty over GR as being recognised under the CBD, with

[2] Property can be broadly understood as the social organisation of rights and entitlements over resources, both physical and intellectual, and may include the right to access biocultural resources or to exclude others from accessing these resources.

[3] For a list of the countries that are party to the CBD, see http://www.cbd.int/convention/parties/list.shtml.

[4] Sam Johnston, Senior Research Fellow, United Nations University-Institute of Advanced Studies TK initiative.

national legislation about access and benefit sharing, and property rights over GR. A third approach is followed especially by industrialised countries, which does not consciously refer to the CBD but use other international treaties to make the connection between GR and property. For example, the USA makes the connection between GR and property without referring to the interpretation of Article 15 of the CBD but relating GR to property under the intellectual property rights system. The intellectual property rights law has expanded in many ways, including into fields such as software, and biotechnological products and processes. In this context, the CBD has had a strong impact on the sociolegal dynamics associated with biocultural rights in national and international law.[5] Hence, the Convention on Biological Diversity, as an international legal instrument with a binding character, has changed the landscape of property rights claims over biocultural resources. In particular, the CBD has influenced the way in which bioculturally-rich countries reassert and interpret the legal principle of state sovereignty over plant forms.

In 2010, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization (Nagoya Protocol) was agreed in the 10th Conference of the Parties of the CBD in order to advance in the implementation the CBD's third objective.[6] The Protocol is still ambiguous in parts, such as regarding products derived from genetic resources (Bille et al. 2010). However, in contrast to the text of the CBD, the Nagoya Protocol refers explicitly to intellectual property rights in relation to benefit sharing, prior informed consent and mutually agreed terms. In its Article 6, it mentions the need to "establish clear rules and procedures for requiring and establishing mutually agreed terms. Such terms shall be set out in writing and may include, inter alia: … (ii) Terms on benefit-sharing, including in relation to *intellectual property rights*" (emphasis added) (Article 6.3(g)). Article 6 also states that "access to genetic resources for their utilization shall be subject to the prior informed consent of the Party providing such resources that is the country of origin of such resources or a Party that has acquired the genetic resources in accordance with the Convention, unless otherwise determined by that Party" (Article 6.1). In the Annex to the Nagoya Protocol, "Joint ownership of relevant intellectual property rights" is mentioned as a potential monetary and non-monetary benefit derived from access and benefit sharing agreements.

Based on the above-mentioned Article 15 of the CDB, one interpretation is that states have the right to vest the property rights over GR located in their territory and allocate these rights on the government or alternatively on individual or collective owners of land where the GR are located. Consequently, the CBD has strengthened GR providers' claims on benefit sharing (e.g. ten Kate and Laird 1999; Tobin 2002).

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[5] A database on access and benefit sharing measures undertaken by Parties of the Convention can be found at: http://www.cbd.int/abs/measures/

[6] The Nagoya Protocol is available at http://www.cbd.int/abs/text/. By October 2011, 65 countries had signed the Nagoya Protocol (http://www.cbd.int/abs/nagoya-protocol/signatories/). The CBD's third objective is "the fair and equitable sharing of the benefits arising out of the utilization of genetic resources" (Article 1).

Under the CBD and the Nagoya Protocol in particular, if the interests of the government and local communities on whose lands GR resides are in tension, the final decision-maker would be the national government and would need to base its decision on the respective national legislation (see Article 15.1, CBD).

Provisions under the CBD (e.g. Article 8(j)) and the Nagoya Protocol recognising the rights of indigenous and local communities to GR and TK include such a limitation by including phrases such as "in accordance with domestic legislation." For example, Article 5 of the Nagoya Protocol entitled "Fair and Equitable Benefit-sharing" mentions: "Each Party shall take legislative, administrative or policy measures, as appropriate, with the aim of ensuring that benefits arising from the utilisation of genetic resources that are held by indigenous and local communities, *in accordance with domestic legislation* regarding the established rights of these indigenous and local communities over these genetic resources, are shared in a fair and equitable way with the communities concerned, based on mutually agreed terms" (emphasis added) (Article 5.2). The recent Nagoya Protocol though may offer certain means of articulating the interests of governments and local communities specifically through notions such as the recognition of biocultural community protocols and customary norms including the use and exchange of GR and associated knowledge within and among indigenous and local communities (see Articles 12.1; 12.3(a); 12.4 and 18.5).[7]

5.3 The Role of Governments in Bioprospecting

In this section, we focus on two key roles played by governments in bioprospecting: firstly, to set the scene for bioprospecting by shaping the national regulatory framework for such projects, and secondly, to enforce that framework by participating in the implementation of bioprospecting projects.

5.3.1 Setting the Market Scene

The CBD aims to provide long-term conservation incentives (e.g. MA 2005; Bille et al. 2010). However, here we question whether the CBD may cause unintended effects in the short term in terms of potentially placing obstacles in bioprospecting projects. Specifically, CBD may cause uncertainty within the projects that may reduce their scope for providing long-term conservation incentives.

[7] At the national level, certain countries such as India and Peru have been very active in developing laws and policies on ABS and local communities (Ituarte-Lima and Subramanian 2011). These countries would be already on their way of implementing certain related obligations derived from such a Protocol. Nonetheless, they would still need to develop and articulate different national provisions in order to fully implement its content and specify many areas that are not detailed in the Nagoya Protocol but which refer to the development of national legislation (see e.g. Articles 15 and 16).

Uncertainty tends to cause transaction costs (Williamson 1985; Bromley 1991) where such costs are broadly understood as the costs of running the economic system (*sensu* Arrow *sensu* 1969: 48) and create concomitant contractual hazard, that is, negatively influencing factors that increase the risk of deviating contract outcomes from the contractual goals (Oxley 1999). Transaction costs in terms of contractual hazard and their links to modes of governance have been explored in the literature, not least regarding business alliances at the domestic and international levels (Oxley and Sampson 2004; Oxley and Silverman 2006) and their effect on project outcomes (Poppo and Zenger 2002; Wang and Chen 2006). Generally, under high uncertainty, leading to transaction costs, coordinated instead of autonomous adaptive capacity to uncertainty is usually held to perform better (e.g. Oxley 1999; Williamson 1999; Oxley and Silverman 2006).

Transaction costs in bioprospecting projects are directly linked to government policies in order to regulate bioprospecting projects and may cause contractual hazard, which often bears a negative effect on their outcomes. A potential consequence is a trade-off between having clearly enforced property rights by governments regarding biodiversity holders such as rural communities, and transaction costs potentially leading to contractual hazard and increasing the risk of failure of bioprospecting contracts. This may be due to contract hazard being a function of the attributes of the providers or the demanders of GR as well as their capacity to adapt their alliance within bioprospecting projects.

Well-defined property rights are generally held as a precondition for reducing uncertainty in investment decisions (Pindyck 1988; Caballero 1991; Dixit and Pindyck 1994; Bell and Campa 1997). This argument has been put forward also for bioprospecting, leading to the idea of the need for clear regulatory frameworks (Bhatti 2003; Larson-Guerra et al. 2004) to facilitate negotiation of new projects (Tobin 2002). Prior to the CBD, access to GR was often gained without consent of GR holders, leading to situations known as *biopiracy*. Demanders used to identify and locate GR that appeared valuable for their aims. Bioprospecting projects were conducted largely without formal contracts, but instead demanders of GR would sometimes pay a small amount of money up-front to the provider of GR, as a compensation only for the labour time local people who helped to locate the GR being sought. However, under the CBD, countries have the right to vest the property rights over GR located in their territory and grant these rights to the state or alternatively on individual or collective owners of the land where the GR can be found (CBD, Article 15). As a result, the CBD has strengthened GR providers' claims on benefit sharing (e.g. ten Kate and Laird 1999; Tobin 2002).[8]

Changes in local institutions often affect contractual hazard because they can potentially open up for disputes of interest in the quest for private appropriation of

[8] Provisions under the CBD (e.g. Article 8(j)) and the Nagoya Protocol recognising the rights of indigenous and local communities to GR and TK include such a limitation by including phrases such as "in accordance with domestic legislation."

benefits (e.g. Libecap 1989; Ostrom 2007). This is typical, for example, with the entrance of external stakeholders in order to extract locally available natural resources. One example associated with bioprospecting refers to the situation where biotechnology has expanded the use of GR in pharmaceutical research and has increased demand for GR from the South (Parry 2004). Consequently and logically, the CBD is striving to solve the resulting North-South disputes in such situations (Dutfield 1999; Suneetha and Pisupati 2009). But in doing so too, the CBD might have increased transaction costs in bioprospecting (Swanson et al. 2002), for instance, by increasing the need to identify and specify ownership to GR and associated TK. Such transaction costs can be especially high especially in situations where ownership of GR is contested among cross-border communities and whose investment in biodiversity conservation is often distributed across generations (Laird 2002; Parry 2004; Dedeurwaerdere 2005).

The number of stakeholders and the heterogeneity in bioprospecting contractual arrangements have increased significantly, following CBD ratification among countries. Since the notion of "rights" encompasses different definitions for different bioprospecting stakeholders (Parry 2004; Hayden 2008), differences in beliefs and motivations among project participants have also increased (ten Kate and Laird 1999), regarding legal concepts, as well as differences in how the agents involved in projects organise their social and economic activities (Brush 1999). This can result in a higher degree of uncertainty about whether there is, or what is the definition of, a "just sharing of benefits" from GR and TK (Laird 2002). Hence, bioprospecting legislation becomes more complex and harder to use as a means to assist the governance of the different interdependent interests that need to be addressed. The latter ranges from social development and biodiversity conservation to a predictable investment context (Larson-Guerra et al. 2004). For example, even the Costa Rican bioprospecting legislation, which has received much praise in the past, has been criticised for not sufficiently addressing indigenous communities' claims over ownership of GR and TK and hence appropriate compensation levels (Carrizosa 2004).

Additionally, the effectiveness of property rights over GR hinges on the cost of enforcing them. Increasing the level of detail in national laws inspired by the CBD also increases the bureaucracy in source country governments, which tends to further increase transaction costs in bioprospecting. In addition, binding laws with a lack of clear authority can create further obstacles, especially in settings where there is a lack of clearly defined authority to issue the necessary permits for bioprospecting (Laird 2002).

5.3.2 Active Government Participation in Project Implementation

Another way for governments to influence the outcome of bioprospecting projects is by directly engaging in their implementation. The role of transaction costs in contractual hazard and modes of governance, such as in business alliances at the

Table 5.1 Typology of governance attributes of bioprospecting contracts

Governance attribute	Private-private contract	Public-public contract	Public-private contract
Incentive intensity of management	High	Low	Medium
Adaptive capacity to uncertainty	Autonomous	Coordinated	Autonomous/ coordinated

domestic and international levels (Oxley and Sampson 2004; Oxley and Silverman 2006), and the effect that transaction costs have on project outcomes (Poppo and Zenger 2002; Wang and Chen 2006) can be explained focusing on ideas from new institutional economics.

Decentralised organisations tend to provide high-performance incentives, also known as "incentive intensity" (Williamson 1985).[9] They also tend to have high capacity for autonomous adaptation to uncertainty. However, when transaction costs are high, due to contractual uncertainty, coordinated as opposed to autonomous adaptive capacity to uncertainty tends to perform better (e.g. Oxley 1999; Oxley and Silverman 2006; Williamson 1999). It follows that contractual hazard in bioprospecting could be reduced by an adequate organisational set-up (ten Kate and Laird 1999). Table 5.1 characterises bioprospecting projects as conforming to either two private participants ("private-private"), two governments ("public-public") and a mixture ("public-private"), with their respective expected characteristics.

Governments are an example of strong vertical integration with high capacity for coordinated adaptation to uncertainty. They may be well placed to handle complex project coordination tasks that are themselves a result of multifunctional resources. Governments can also build and transfer knowledge collectively about how to manage complex projects, such as in the context of the CBD. The question thus arises as to whether government participation in the implementation phase of bioprospecting projects, by, for instance, providing capacity for coordinated adaptation to the inherent uncertainty of such projects, can help to reduce transaction costs and contractual hazard in the context of bioprospecting. Similarly, one could ask whether weakening the role of the private sector as a bioprospecting partner reduces the capacity of projects for autonomous adaptation to uncertainty. Answering these questions helps to shed light on the potential role of public-private alliances to reduce the level of transaction costs that are common to most bioprospecting endeavours. In the next sections, we provide an empirical analysis to shed light regarding this issue.

[9] The concept of incentive intensity can be exemplified by contrasting the market mechanism with governments (Williamson 1999). The profit goal of a private company is likely to provide a more direct link between performance and reward, that is, high incentive intensity. As a comparison, this link is in general lower for government activities, partly as a consequence of the public good nature of many of the goods and services it provides, which among others makes monitoring more difficult.

5.4 Methods

5.4.1 Data

A database of 190 bioprospecting case studies was constructed from a systematic review of the literature that for the most part contained information from individual project case studies described by social scientists who revised individual projects in which they were not themselves directly involved. In a few cases, the reports were written by the bioprospector themselves (e.g. ICBG), and these were quality checked with interviews with independent experts from academia, ex situ collections for GR and the private industry.

A detailed analysis was conducted on a subset of 67 cases which held sufficient information for the purpose of the analysis. The dataset included bioprospecting projects that were initiated between the years 1990 and 2003. The geographical spread is Africa (11 cases), Asia (16 cases), Latin America (28 cases) and Small Island Developing Nations (12 cases). These projects were associated with the transaction of principally plant GR but also microorganisms and in one case, animal GR. Most of these cases were also associated with TK and in some cases involved the explicit participation of traditional communities in the bioprospecting projects.

Since there is no centralised accurate dataset of bioprospecting cases, it was not possible to determine the actual number of all bioprospecting cases in the world. While the results cannot be directly extrapolated directly, the cases in the sample used here are fairly representative of typical North-South bioprospecting contracts. It could be argued, though, that there might be some bias as data for relatively successful cases might be overrepresented. However, the fact that the database includes also a large part of more or less failed contracts partly responds to this concern. Nonetheless, the overall results should be taken with due caution as they represent a first attempt at understanding contractual hazard based on available data rather than on all existing bioprospecting cases.

5.4.2 Identification of Relevant Variables

Following the discussion in Sect. 5.2, it is held that contractual hazard constitutes the link between the market setting and project outcomes. We expect that higher transaction costs in the contracts cause contract hazard, which in turn increases the likelihood of negative project outcomes.

Bioprospecting projects are re-evaluated along the contracting process as typically any investor faces repeated situations where they need to choose whether to continue the contracting process or to wait in order to acquire additional information. A variable is specified that denotes the outcome of individual bioprospecting projects. Projects that have either been cancelled or experienced substantial interruption are distinguished from those that have proceeded uninterrupted.[10] Table 5.2 describes the variables and adds additional information and a principal component analysis is conducted (Sect. 5.1).

In order to look into the potential effect of the legal framework for bioprospecting causing transaction costs which in turn may increase contractual hazard, hence potentially undermining contract outcomes, a set of three categorical variables which are interpreted together is introduced: "*CBD RATIF*" denotes the strongest form of formal legal certainty, that is, projects initiated in a country that has ratified CBD.[11] "*CBD NONE*" denotes projects subject to low formal legal certainty because they were initiated before CBD entered into force globally in 1993. The comparison variable "*CBD WORLD*" denotes whether the bioprospecting project was initiated before CBD came into force globally (which implies it must have been initiated before source country ratification of CBD) or after the CBD came into force globally (which can imply either before or after source country ratification of CBD). *CBD RATIF* could be expected to have a negative effect on bioprospecting project outcomes by incrementing transaction costs and contractual hazard. *CBD NONE* is expected to have a positive effect.[12,13]

The complementary question regarding the potential effect of government participation in bioprospecting projects is addressed in a tentative way by analysing how the participation of different kinds of project participants affects the outcome of the projects. A supply side and a group of three demand side variables as well as an interaction variable are introduced to represent the level of government participation in the project. "*PROVIDER GOV*" denotes whether the source country government participates as an active partner in the bioprospecting contract.[14] We primarily

[10] "Interruption" refers to any delay anticipated to obstruct any progress towards completion of the project in the foreseeable future. Project initiation refers to when project funding was approved for the demander, or, when not available, the first year we have a record of that the project was negotiated or implemented. The lower limit, 1990, allows for pre-CBD cases to be included as a control group. We assume that the years outside the 1990–2003 period are not as well reviewed and hence less representative than within the period. This is because we do have records of projects existing beyond this period, but we have not been able to obtain review reports about their outcome. The upper limit, 2003, allows for reasonable time for the project to have been scrutinised in available case studies. A sensitivity analysis shows that more recent projects did not experience fewer interruptions.

[11] Because CBD is binding once it is ratified, and one of the components of CBD is to legislate about bioprospecting, we expect that on average, there is a positive and reasonably strong correlation between CBD ratification and implementation of bioprospecting policies.

[12] The *CBD WORLD* variable is time dependent and may be correlated with the maturity of direct and indirect source country stakeholders. That is, *CBD WORLD* may be related to the maturity of international watchdogs (because such non-governmental organisations and other actors can be argued to have been affected by the CBD coming into force at the international level). *CBD RATIF* helps to control for the time dependency. Namely, while *CBD WORLD* represents 1 year, 1993, *CBD RATIF* relates to different years in different countries, from 1994 to 2003.

[13] Regarding the interpretation of the CBD variables: the fact that bioprospectors did not acknowledge prior informed consent and access and benefit sharing issues prior to CBD signals that it is the regulatory pressure that drives CBD compliance and not the demander's project rationale *per se*.

[14] That is, beyond providing the necessary permits and similar bureaucratic tasks.

Table 5.2 Description of variables

Variable	Explanation	Mean	Min	Max
OUTCOME	1 = Project without cancellation or substantial interruption, 0 = cancelled or substantially interrupted project	52%	0	1
Market setting for bioprospecting				
CBD NONE	Dummy.1 = Project started before that CBD entered into force globally, 0 = lse	10%	0	1
CBD RATIF	Dummy.1 = Project started after CBD ratification in provider country, 0 = else	81%	0	1
CBD WORLD	Dummy.1 = Project started after that CBD entered into force globally, but before CBD ratification in provider country, 0 = else	9%	0	1
Type of government involvement in project				
PROVIDER GOV	Dummy. 1 = The provider country government participates in the project (with or without other, non-governmental, participants), 0 = not (only non-governmental provider/s participates)	66%	0	1
ICBG	Dummy. 1 = The demander is ICBG, 0 = else	28%	0	1
NCI	Dummy. 1 = The demander is NCI, 0 = else	13%	0	1
DEMANDER PRIVATE	Dummy. 1 = The demander a private firm, university or botanical garden, 0 = else	58%	0	1
NCI ICBG-GOV	Dummy. 1 = Project is an alliance of NCI or ICBG + the provider government, 0 = not (any of the three alternative combinations of market and non-market participants, that is, a government provider + private demander; non-governmental provider + private demander; or non-governmental provider + NCI/ICBG demander)	37%	0	1
Other variables: industry type				
DEMANDER RnD	Dummy. 1 = The demander is a pharmaceutical R&D firm (without commercialisation to end market), 0 = else (demander organisation is either *DEMANDER NON-PHARM or DEMANDER ENDMARKET*)	52%	0	1
DEMANDER NON-PHARM	Dummy. 1 = The demander organisation is not in the pharmaceutical sector, 0 = else (demander organisation from the pharmaceutical sector)	13%	0	1

(continued)

Table 5.2 (continued)

Variable	Explanation	Mean	Min	Max
DEMANDER ENDMARKET	1 = The demander is a pharmaceutical firm, with commercialisation to end market (with or without conducting R&D), 0 = else (demander organisation is either *DEMANDER RND* or *DEMANDER NON-PHARM*)	34%	0	1
DEMANDER DOMESTIC	Dummy. 1 = The demander is from the provider country, 0 = else	10%	0	1
RENEWAL	Dummy. 1 = Project is a renewal of a previous bioprospecting project, 0 = else	26%	0	1
GDP CAP	Continuous. Gross domestic product per capita in the provider country, the year the project started (USD 2008 value)[a]	4,265	223	688,337
POP GROWTH	Continuous. Rural population growth (annual percentage) in the provider country, in 1998[b]	0.19	−2.06	3.0
PHILIPP	Dummy. 1 = The project is implemented in the Philippines, 0 = else[c]	15%	0	1

[a]World Bank. http://ddp-ext.worldbank.org/ext/DDPQQ/report.do?method=showReport. Accessed 20 May 2008
[b]World Development Indicators, World Bank
[c]The variable is included in the CatPCA analysis only

expect that active government participation makes it more likely that the CBD provisions are implemented, adding a layer of transaction costs to the project. But source government participation may to some extent also provide further capacity for coordinated adaptation to uncertainty. A positive coefficient associated with this variable may suggest that the positive influence of such capacity is stronger than the negative bureaucratic influence on project outcomes.

A group of three variables representing different levels of government participation is also accounted for, reflecting different levels of vertical integration. The strongest government participation case is represented by the variable "*NCI*" which is associated with projects by the US National Cancer Institute, a governmental organisation. Another variable, "*ICBG*", denotes the International Cooperative Biodiversity Group and represents a consortium of governmental, industry participants and often academic participants. The comparison variable "*DEMANDER PRIVATE*" denotes a non-governmental demander such as those from the pharmaceutical sector.[15]

Since the capacity for adaptation to uncertainty represented in the entire project alliance is expected to be relevant key aspect affecting project outcomes, the variable "*NCI ICBG-GOV*" denotes that the provider government participates and that the government is present on the demander side (either by ICBG or NCI). We expect a negative effect because both capacities for coordinated and autonomous adaptation may be needed to govern GR.

Further, it is also necessary to analyse the determinants of the various project outcomes both at the contract level and at the level of the provider country. Firstly, the intended use of GR by the demander may affect project uncertainty, and to control for this, two categories of pharmaceutical companies are taken into account.[16]

We expect that the pharmaceutical sector in general has attributes associated with high uncertainty, transaction costs and therefore high likelihood of contract interruptions. The reason is the high uncertainty associated with developing new drugs, gaining patent approval and regulatory approval for marketing and subsequently successfully markets the drug. The variable "*DEMANDER END*" represents pharmaceutical organisations that commercialise products at the end of the innovation chain (although they may additionally enrol in research and development, R&D, activities). Another type of demander not engaged in commercialisation, but only in research and development activities, is denoted by "*DEMANDER RND*." Lastly, the third variable in this group, "*DEMANDER NON_PHARM*", denotes a minor number of bioprospecting cases in which the demander is from other than the pharmaceutical sector.

The variable "*DEMANDER DOMESTIC*" denotes whether the organisation on the demander side of the project is located in the provider country, with an expected

[15] "Private" is used in the meaning that there is not explicit participation in the project on behalf of the demander governments. The category includes both for profit organisations such as Pfizer, but also universities and botanical gardens. Notably, in the sample provider, country governments tend to participate more often in such private endeavours as compared to in NCI or ICBG projects.

[16] Both are GR demanders that are dedicated to pharmaceutical products. "Industry" is used to denote the orientation of the demander, that is, applied research and/or product development aimed for commercialisation, as opposed to basic research.

positive association with uninterrupted outcomes (due to e.g. an informational advantage concerning the cultural setting, as well as national legal and institutional frameworks).[17]

It is also important to control for whether the bioprospecting projects constitute an extension to prior bioprospecting projects. Project renewals are expected to affect project outcomes positively by giving more room for sequential decision-making and hence reduced problems of measurement and behavioural uncertainty (e.g. Balakrishnan and Koza 1993; Williamson 1985). This is taken into account by the variable "*RENEWAL*".

Other factors at the more macro level which might influence the outcome of bioprospecting projects can be controlled for to some extent. For example, GDP per capita in the provider country ("*GDP CAP*") is included to control for the possibility that governments in poorer countries have fewer resources to set aside for implementing and enforcing regulation of bioprospecting (Gupta 2004; Siebenhuner and Suplie 2005). Likewise, information about rural population growth ("*POP GROWTH*") is included, since rural population growth might put pressure on local institutions and property rights regimes, thereby affecting project outcomes in a negative way.[18]

5.4.3 Description of the Data

Figure 5.1 depicts the main group of variables related to market setting attributes associated with the property rights setting of bioprospecting projects. The figure relates property rights regime (the three CBD variables) to project outcome and is consistent with the expectation as developed in Sect. 5.2, that is, that the market setting for bioprospecting, measured by the status of CBD, is associated with the outcome of bioprospecting projects. As it can be seen, the highest share of unsuccessful project is in countries that have ratified the CBD.

Table 5.3 describes the data regarding the type of active government participation in the project. The table orders the variables with respect to project participation by provider country governments. It can be seen that slightly more than half of the projects in the sample proceeded without cancellations/interruptions (the mean value of *OUTCOME* is 53%). Interestingly too, it can be seen that provider country governments participate more frequently in countries that have ratified the CBD.

[17] US demanders, by originating in a country that has not ratified CBD, could be expected to apply CBD guidelines only seldom and hence face lower transaction cost. However, this is counterintuitive to the fact that the US data is biased by NCI and ICBG cases (headquartered in the USA), both of which often adopt fairly detailed ABS regulations.

[18] Note that due to data constraints, this variable represents the year 1999, for all projects. Although this is not fully representative since some projects were active during other years, the majority of the projects were active close to this year. Furthermore, it is perceived that rural population growth is relatively stable across short periods of years, such as in the dataset. Hence, we hold that it is reasonable to use this specification of the variable.

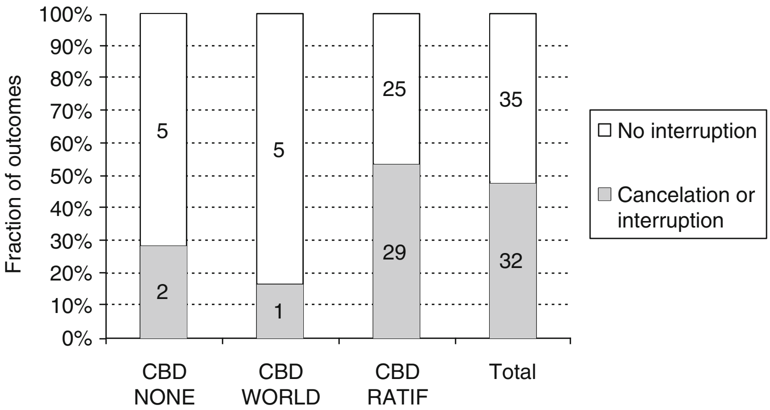

Fig. 5.1 Project outcome and status of CBD for the cases used in the categorical principal component analysis (the numbers in the bars indicate number of cases per outcome, $n=67$)

Table 5.3 Variables ordered by whether the provider country government participated or not (mean, $n=67$)[a]

Variable	Government participation	No government participation	Average
OUTCOME	53%	50%	52%
CBD RATIF	84%	73%	81%
CBD WORLD	11%	5%	9%
CBD NONE	4%	23%	10%
NCI	13%	14%	13%
ICBG	40%	5%	28%
DEMANDER PRIVATE	47%	82	58%
DEMANDER ENDMARKET	33%	36%	34%
DEMANDER RND	60%	41%	54%
DEMANDER OTHER	7%	23%	12%
DOMESTIC DEMANDER	9%	14%	10%
RENEWAL	27%	23%	26%
POP GROWTH	0.09	0.43	0.20
GDP CAP	5,193	2,414	4,280

[a]The table reads as follows (e.g. for the variable *OUTCOME*): the mean of *OUTCOME* in projects with participation of the source country government is 53% for the sample (i.e. in 47% of projects with government participant, the outcome was negative). When the source country government does not participate, the outcomes were somewhat lower on average, that is, only 50% of projects had a positive outcome

However, participation by provider country governments in the contracts does not appear to be associated with project outcomes. Additionally, the private sector is the most common bioprospector in our sample with pharmaceutical RnD organisations being the most common demander, followed by pharmaceutical organisations that also engage in manufacturing and sales (*DEMANDER ENDMARKET*). The most notable difference in the level of government participation from the source country is among RnD pharmaceutical organisations and non-pharmaceutical organisations. A minority of projects, 26%, are renewals.

Table 5.3 also shows that source country governments participate more frequently in richer developing countries (the mean GDP per capita is higher in projects in which the source country government participates, with USD 5,193 as compared to USD 2,414). This may indicate that countries with more solid government institutions (as typically associated with higher GDP per capita) have a higher ability to implement international legal obligation and country level legislation in general.

5.4.4 Analysis of Contractual Hazard in Bioprospecting

In order to understand the link between government intervention and specific outcomes of bioprospecting contracts, it is necessary to understand the role of government intervention beyond the contract level. Therefore, we empirically explore the role that the two mentioned government functions have on the overall contractual context of the projects. Although the data does not allow controlling for all potential factors that might affect project outcomes, the included variables can together be related to a substantial source of influence in contractual hazard.

Based on a principal component analysis (PCA), we identify dimensions (or groups of variables) which account for underlying relationships in the data beyond the effect of isolated individual variables alone. Specifically, we use a categorical PCA (henceforth CatPCA) to provide insight by (1) identifying which groups of variables associated with the role of the government in setting the market scene, or actively implement bioprospecting project as an active participant, have influence over the project contractual context and the degree of that influence; (2) pointing out pre-established expected relationships or, in an explorative way, gain insight into the role of variables not envisioned to have an influence on project outcomes; (3) looking at how such dimensions rank in importance between each other; and (4) looking at how individual variables rank in importance within each dimension.

A particularly useful feature of CatPCA that adds to standard PCA is a rescaling procedure. In standard PCA, only continuous or categorical variables can be analysed separately, not together. The CatPCA rescaling procedure transforms continuous variables to categorical variables ordered in seven levels. While this means that information is lost as compared to the original continuous variable, it does allow including considerably more information as compared to a transformation to a dichotomous variable as typically used in standard PCA.

5.5 Results and Discussion

Table 5.4 shows the results of the CatPCA. All three dimensions included have an eigenvalue above one: 3.67, 2.83 and 1.95, respectively. The overall explanatory power of the variables is reasonable, at 49.7%, with 21.6%, 16.6% and 11.4% of the

Table 5.4 Summary of categorical principal component analysis ($n = 67$, variables ordered along dimensions and along their factors loadings with highest loadings to the left)

Principal component dimension	Variance explained	Cumulative (%)	Variables with moderate to high loadings (above 0.3)
1. Type of government involvement	21.59	21.59	*NCI ICBG-GOV, DEMANDER PRIVATE, ICBG, PROVIDER GOV, CBD WORLD, DEMANDER NON-PHARM, DEMANDER END, DEMANDER DOMESTIC*
2. Market setting	16.64	38.23	*CBD RATIF, DEMANDER RND, CBD NONE, POP GROWTH, DEMANDER END, NCI, PHILLIP, PROVIDER GOV, DEMANDER NON-PHARM, OUTCOME*
3. GR use	11.44	49.67	*NCI, DEMANDER DOMESTIC, DEMANDER END, DEMANDER RND, CBD NONE, CBD RATIF*

variance explained, in the first, second and third dimension, respectively. We follow Kline (1994) and classify loadings higher than 0.30 as "moderate to high." According to this criterion, all variables except for *RENEWAL* and *GDP_CAP* have reasonably high explanatory power in at least one of the three so-called underlying, or latent, dimensions.

The first dimension is largely explained by variables relating to the governments' active implementation as a project counterpart in the bioprospecting cases (Table 5.2). The highest component loadings are represented by projects with governments at both the supplier and demander side *(NCI ICBG-GOV*), followed by project with private demanders (*DEMANDER PRIVATE*) and ICBG projects. Fourth are projects in which the provider country government participates (*PROVIDER GOV*). A key focus is in interpreting the interaction variable, since the contract hazard is a function of the overall capacity for adaptation in the alliance of providers and demanders, not only of the attributes of the providers or the demanders as analysed separately. Hence, when analysing the supplier and demander side together instead of separately, the variable "*NCI ICBG-GOV*" shows that strong government participation (i.e. governments participate as both supplier and demander) has a strong influence over the bioprospecting contract context as compared to other projects where there is no governmental participation at all. This may suggest that capacity for coordinated adaptation is important in order to address the high level of uncertainty about, for example, commitment to contractual terms in bioprospecting.

This information provides tentative support for the role that different kinds of government participation plays in explaining the bioprospecting contractual context. This dimension being the first in terms of component loadings, it means that among the variables included, government participation of one kind or the other is what most influences the bioprospecting contract context.

The second dimension relates somewhat to the market setting of bioprospecting contracts.[19] The variable representing the specification and protection of property rights for GR has the highest component loading (*CBD RATIF*). *CBD NONE* also has a significant loading in the second dimension and also represents the market setting. Rural population growth (*POP GROWTH*) with the fourth strongest loading might be assumed to proxy the broader institutional context of the project. Taken together, the results of these three variables can be interpreted as that the second strongest influence to the bioprospecting contractual context among all the variables assessed is the government's role to specify the market context.

Lastly, the third dimension can be said to represent the purpose (commercial/non-commercial) of the demanded GR, with the two variables denoting a demander from the pharmaceutical industry (*DEMANDER END* and *DEMANDER RnD*) having the third and fourth highest component loadings within this dimension according to the categorical PCA.

The activity of the demander (*DEMANDER RND, DEMANDER NON-PHARM*) is less clear to interpret, since they are distributed across two different dimensions, and does not have significant loading in any of the two. Therefore, it is not possible to interpret the different effects of having pharmaceutical end market firms, pharmaceutical R&D organisations or non-pharmaceutical organisations playing a role in bioprospecting contracts. One possible interpretation is that uncertainty related to institutional factors (e.g. market setting and government participation) has a greater role in project outcomes as compared to technical uncertainty of downstream research and commercialisation activities. Interestingly, the fact that projects might be renewed (*RENEWAL*) does not seem to influence the contractual context, possibly due to the strong influence of the government's role both as active implementation participant and by setting the market scene.

The results of the CatPCA analysis are fairly consistent with the conceptual framework regarding the role that active government participation in project implementation plays in the bioprospecting contractual context. The results indicate that governments might not only influence the project by setting the property rights scene (through ratification of CBD) but more importantly by actively implementing such projects as a project partner.

5.6 Conclusions

Against the background of the recent Nagoya Protocol (October 2010) on access and benefit sharing of genetic resources, in this chapter, we have attempted to cast new light on how the CBD might be, in an unintended way, affecting bioprospecting

[19] Note that many of the variables are represented in both dimensions. However, their component loadings are in several cases very different (see Annex A2), hence suggesting their different roles in each dimension.

projects in the short term. The focus has been on assessing the government's role in setting the market scene for genetic resources by specifying the property rights and implementing the bioprospecting policy framework. Based on a systematic review of bioprospecting case studies, we suggest that the CBD, which has led to more clearly defined property rights over genetic resources regarding ownership of the providers, might have had a strong effect on the contractual context. The reason is that stricter property rights, while being the foundation for linking southern conservation effort with financial incentives, might have also caused a novel contractual situation. From reviewing bioprospecting cases and interviewing bioprospecting stakeholders, we think that by the emergence of new stakeholders and socio-economic contract contexts, contract uncertainty might have increased, in turn increasing contractual hazard. Such contractual hazard can be ameliorated by the type of government involvement in the implementation of bioprospecting projects.

After the adoption of the Nagoya Protocol, a major critique has been raised against it, on the basis that the Protocol is ambiguous in parts, etc. Here, we put forward an additional idea: even a clear and specific protocol would in fact not be sufficient to overcome the high contract uncertainty built into any bioprospecting project due to their inherent heterogeneity in terms of both asymmetric information and expectations about the outcomes of such projects.

It is a fact that there are ample difficulties to implement benefit sharing for genetic resources at the international level. It is important to note in this context that if the allocation of private property rights over genetic resources is envisaged, special attention ought to be paid to the institutional set-up of bioprospecting projects. As private property rights might be further strengthened in bioprospecting cases, the role of governments become increasingly more important. But there is still much to be learnt about the way public and private stakeholders can efficiently and equitably interact to help achieve the CBD's goal of conservation, access to and benefit sharing of global biodiversity. It will be necessary to systematically assess how the Protocol has affected contract hazard in bioprospecting projects as new data on projects become available.

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Part II
The Construction and Evolution of Governance Regimes

Chapter 6
Political Transformation and Watershed Governance in Java: Actors and Interests

Martin Christian Lukas

6.1 Introduction

This chapter analyses transitions in watershed and forest governance in Java. It focuses on the reorganisation of political structures, changing actor constellations and the emergence of new management approaches. These transitions have been part of broader political transformations in Indonesia and reflect learning effects from experiences with previous management approaches, shifts in broader political and scientific discourses as well as developments on international scales. This chapter will particularly shed light on changes in the interplay of various actors, their interests and power relations and their discursive constructions of environmental issues and relate these changes to the emergence of new management approaches. It describes a transition from a pronounced state-led, centralistic, hierarchical style of government, claiming sovereignty of interpretation over environmental issues and being largely based on command-and-control structures, to more dynamic and vertically

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more fragmented, network-like forms of governance with an increasing involvement of non-state actors, new actor coalitions and a more diverse range of management approaches and initiatives.

The emerging new modes of governance can be characterised by their much 'flatter', more horizontal structures, for which different authors have used terms like 'networked' or 'nodal governance' (Parker 2007; Burries et al. 2005). However, these terms have to be specified in the context of Indonesian watershed governance. Despite revolutionary political transformations in the country and visible progress in many areas, simplistic, one-sided sociopolitical discourses have persisted in some realms, and some of the previously most powerful actors continue to dominate political processes and decision-making in the fields of natural resources use and environmental conservation. This is in line with one of the main arguments developed by Flitner and Görg (2008) that contrary to the intuitive implication of governance 'networks' describing trends towards less hierarchical, more participatory regimes, dominant centres of power and influence often continue to exist. Yet they are not situated in well-defined hierarchical levels as before but less visibly embedded in complex networks which may at the same time be more dynamic, with changing temporary constellations or 'nodes of power'.

Yet political rights and possibilities to exert influence of those who were previously excluded, repressed or at best seen as passive recipients or implementers of watershed interventions, have tremendously improved. Following their powerful, and at the same time environmentally disastrous, rebellion in the frame of a nationwide political upheaval in the late 1990s, they now have more freedoms to push their interests, to struggle for their rights, to set up networks and form alliances with new actors, supporting them in questioning established sociopolitical discourses and setting up new management approaches that are locally rooted but influenced by actors from various scales. Like previous political transformations during the twentieth century, the political transitions since 1997 have exacerbated environmental problems. But at the same time, they have provided the scope for developments towards new, socially and ecologically more sustainable modes of watershed governance.

6.1.1 Watershed Governance in Java: Long-Standing Investments, Long-Standing Challenges

The volcanic island of Java, by nature already a highly dynamic environment, has been rapidly transformed by human activities. The rapidly increasing magnitude of human impact and the islands' reputation as one of the most densely populated areas in the world pushed Java into the spotlight of Malthusian prophecies of severe environmental and human disaster even long before environmental concerns found their way into the global scientific and sociopolitical mainstream (see, e.g., Geertz 1964). Particularly deforestation, soil degradation, erosion, sedimentation and flooding, all classical watershed problems, became issues to be addressed in the frame of environmental management initiatives.

Concerns over the possible negative hydrological impacts of soil degradation were already raised in the late nineteenth and early twentieth century. Accelerated deforestation in many parts of the island after the mid-nineteenth century, the observation of serious erosion on former coffee and neglected tea plantations in the Priangan, the southern part of West Java, and the growing dependency of lowland agriculture, particularly the colonial sugar industry, on steady water flows for irrigation sparked calls for forest preservation (Oosterling 1927; de Haan 1936) and provided a rationale for the spatial extension and strict management of state forest territories by the Dutch administration.

Following the worldwide surge in river basin development initiatives as a favoured modernisation strategy between the 1940s and 1960s (Ekbladh 2002; Molle 2009), river basin development became a matter of national importance in Indonesia in the late 1960s. With support from international development organisations, a first series of river basin development projects was implemented in some of Java's major river basins, including Solo, Brantas and Citanduy. Watersheds were delineated as planning units nationwide, and watershed management authorities were set up. In 1969, the government established river basin development authorities directly under the Ministry of Public Works in a number of Java's major river basins, including the Citanduy, Citarum and Solo rivers. Starting in the 1970s, political interest gradually turned from agricultural reclamation, irrigation systems and flood protection in the lower river basins to upland conservation. Since then, substantial resources have been allocated for upland conservation measures in Java, both in the frame of donor-funded development projects and in the frame of the continuous national regreening programmes, which have focused on tree planting and terracing of agricultural land. Yet many watersheds in Java continue to be considered as degraded (Interviews April 2008, January 2010).

Research and interventions have mainly focused on optimising small holder farming systems and improving peasants' livelihoods (ADB 1996, 2006; Purwanto 1999; Tim Koordinasi Wilayah 1989; USAID 1985). But this can neither sufficiently explain nor address some of the most important issues and challenges related to upland degradation. My analysis of land use and land cover change and its drivers in the catchment areas of the Citanduy, Cimeneng and Cikonde rivers has shown that state forest areas rather than small holders' private plots seem to comprise the major hotspots of critical, erosion-prone land and that the long-lasting investments into upland conservation, which focused on small holders' private land, have had only limited effects. To understand the underlying causes of this situation and to assess the achievements, chances and challenges related to recent transitions towards new modes of watershed governance, it is important to look at the historically grounded interplay between the various actors, their interests, their power relations, their discursive constructions of environmental issues and their roles in designing and implementing management interventions.

This will be illustrated in the following sections, based on empirical findings of recent research on land use and land cover change and forest and watershed governance in the catchment area of the Segara Anakan lagoon, the estuary of the Citanduy, Cimeneng and Cikonde rivers. Section 6.2 will shed light on some of the major characteristics and patterns of forest and watershed governance during Indonesia's New Order Regime between 1965 and 1997. Section 6.3 will briefly describe the revolutionary, partly chaotic dynamics with regard to forest management following the fall of the New Order Regime between 1997 and 2001. The fourth and main section will explore transitions towards new modes of forest and watershed governance since then. It describes an increasingly open political arena providing opportunities for new or newly empowered actors to frame alternative discourses, to develop alternative management approaches and to challenge long-standing power relations. It further illustrates transformations of parts of the previously centralistic, hierarchical state apparatus into elements of more flexible networks encompassing various state and non-state actors and exposes the pervasive persistence of long-standing nodes of repressive power as major, albeit not easily visible obstacle to sustainable resources governance. The conclusion will reconsider the analysed developments towards new modes of governance, conceptualising them as transitions in the frame of a gradual shift from traditional hierarchical forms of government to networked or nodal governance.

6.2 Watershed and Forest Governance During the New Order Regime 1965–1997: Coercive, Non-participatory Approaches

6.2.1 'A Strong Alliance That Was Hated by the People'

Between 1965 and 1997, forest and watershed management like most other spheres of public life in Indonesia were dominated by a repressive, centralistic, hierarchical nation state. President Suharto's New Order Regime, which particularly during its first two decades was pervasively dominated by the military, was marked by forcibly coordinated command-and-control structures with state institutions on all levels being politically brought into line. The national government, the central locus of decision-making, with its hierarchical, top-down oriented administrative system and its enterprises claimed absolute political sovereignty, which was secured by police and military forces. Accordingly, the state with its sectoral authorities particularly under the Ministries of Agriculture, Forestry and Public Works and its province, district and sub-district administrations as well as its state corporations widely claimed sovereignty of interpretation over watershed and forestry issues and dominated related management approaches. Governance of natural resources, particularly forest resources, was closely entangled with the state's interest in natural resource exploitation.

But also international donor organisations, such as USAID, World Bank and Asian Development Bank (ADB), with their external experts as well as Indonesian scientists substantially contributed to watershed management and the formation of related discourses. However, their spheres of influence were rather limited to the fields of converging interest, where joint collaboration with the Indonesian government was sought by the participating parties, albeit for different reasons. For example, the US-led international surge of river basin development as a preferred modernisation strategy (Ekbladh 2002; Molle 2009) was perfectly in line with the goal of Indonesia's New Order Regime to achieve rice self-sufficiency and to consolidate state control. Hence, river basin development projects marked the beginning of a broader US-Indonesian economic partnership in the late 1960s, which provided the USA the opportunity to promote the western liberal model of modernisation after Suharto's military coup and the subsequent crush of Indonesia's communist movements in one of the bloodiest massacres in Southeast Asian history (Simpson 2008). The USAID-funded Citanduy I Project was one of the first large river basin development projects implemented. It focused on agricultural reclamation of the lower river basin, irrigation schemes and flood control measures. The selection of the Citanduy basin as implementation site was partly related to previous violent conflicts between the state and radical Islamic and 'communist' movements in that area. The river basin development approach as a state-dominated, large-scale intervention was certainly in line with the national government's strong interest to bring these movements under control or respectively prevent their resurgence and to establish or consolidate state control in this politically fragile area.

Aside from the fields of converging interest, the influence of international donor organisations, external experts and Indonesian scientists on management approaches was rather limited. The knowledge, data and recommendations they contributed were usually widely tailored to management approaches and planned interventions that had already been outlined before. Alternative interpretations and critical arguments were at best occasionally raised by a number of foreign researchers. Intransparent administrative structures and procedures were not only a machinery of corruption but could easily slow down unwanted initiatives. For example, the establishment of the socio-economic research unit that was integral part of the Citanduy II project design but apparently, and possibly for political reasons, not sought by the government was effectively hindered by administrative formalities (USAID 1984).

Notwithstanding the substantial achievements of some of the interventions, including the conversion of the lower Citanduy basin into a fertile rice bowl, problematic power and decision-making structures had partly detrimental effects. Besides the national government, provincial- and district-level authorities were in different ways involved in decision-making. But for communities, there was no scope for participation, and residents of the area in retrospect often subsume all state authorities on the various levels from the national government down to the sub-district and partly even the village level as 'a strong alliance that was hated by the people but was better not defied' (Interviews, February 2011). The people were mainly recipients, implementers or targets of management approaches and had very limited possibilities to participate in decision-making.

This constellation of actors and power and the resulting non-participatory, partly coercive management approaches and simplistic, one-sided political discourses had detrimental environmental effects and undermined the effectiveness of management interventions. This will be illustrated in the following two sections, which will exemplarily shed light on coercive forest management and upland conservation measures.

6.2.2 Coercive Forest Management and Forest Degradation

Management of Java's state forest areas, which account for no less than about one quarter of the surface area of this extremely densely populated island, was and still is assigned to the state forest corporation. During the New Order Regime, the state forest corporation was one of the most dreaded organisations within the 'strong alliance that was hated by the people but was better not defied' (Interviews, February 2011).

The state forest corporation has been one of the most relevant actors in watershed governance on Java but has often not received adequate attention in this context. Its forest management practices, its relations with forest margin people, its discursive power and its particular position within institutional arrangements have greatly affected both physical watershed processes and watershed management during the past decades.

Forest areas had been declared as state domain and had been mapped and delimited under the Dutch colonial administration in the second half of the nineteenth century and further extended thereafter. They had been managed in increasingly repressive ways with forest margin people facing severe restrictions of their traditional access to forest resources (Peluso 1992). The state forest corporation, the successor of the 'colonial forest empire', further strengthened the system of coercive power during the New Order Regime. A strong forest police, in case of need supported by the military, secured the state forests from peasants. Fatal shots on villagers were not uncommon.

The political rationale for this coercive forest management approach was rooted in two claims, complementing each other: the state's and its forest corporation's interest in exploiting the forests' productive resources and the argument that watershed conservation required professional forest management. The latter could thrive on the concerns about soil degradation that had already been raised in the late nineteenth and early twentieth century and that were further underpinned in the context of forest devastation during the Japanese occupation between 1942 and 1945, when peasants were encouraged to encroach state forest and plantation land to use it for food crop cultivation – partly officially with leaseholds and partly in-officially (Interviews, February 2011). The rise of environmentalism in Indonesia after the mid-1970s (Cribb 1988) provided additional ground for asserting the state's sovereignty over forest resources. The clear-cut narrative that forests should be managed by professionals for the sake of optimising production and watershed conservation, while peasants would not be able to manage forestland sustainably, entwined the state's and its forest corporation's interest in production and control over forestland (Peluso 1992; Interviews, April 2008, April 2011).

In reality, the approach of coercive management neither allowed optimised production nor was it a successful watershed conservation strategy. State forests remained contested, and even substantial efforts and repressive methods for guarding them could not entirely prevent peasants, who had been deprived of their traditional access to forest resources, from 'illegally' using them. This 'illegal' exploitation not surprisingly was inefficient and destructive, since people, if they took the risk and bypassed forest guards, tended to regard state forests as de facto open access resource, which could recklessly be exploited without paying attention to questions of resource sustainability and optimised production (Nibbering 1988). This resulted in forest and hence watershed degradation and comparably low levels of productivity (ibid., Peluso 1992; Singer 2009). It was not before the late 1970s and early 1980s that increasing hindsight into the failure of coercive forest management and the influx of ideas from the international level paved the way for the implementation of first social forestry programmes, albeit on a small scale and with limited success (Simon et al. 1992; Singer 2009).

However, in the context of watershed management initiatives, including national regreening programmes and donor-funded watershed management projects, degradation within state forest territories never received adequate attention. Professional forest management by the state forest corporation was commonly seen as synonymous with sustainable forests and watershed conservation. Hence, management of state forest territories and their protection from the people for the sake of production and watershed protection were entrusted to the corporation. While forest management by the corporation was discursively framed as epitome of sustainability, the historical roots of the seemingly irreconcilable, ecologically detrimental tensions between the corporation and peasants were never addressed. Also the corporation's own management practices, which have always involved regular large-scale clear cuts, including at steep slopes in close proximity to streams, have never been questioned in the context of watershed conservation, even though these practices obviously cause substantial erosion. The power of the discourse which portrayed forest management by the corporation as a means for sustainable forest and watershed management effectively prevented the emergence of any contrary views.

The persistent contradiction between the political pursuit of watershed conservation and related long-lasting efforts on the one hand and the failure to adequately discuss and address widespread degradation in state forest territories and its causes on the other hand can only sufficiently be explained with historically grounded tensions between the state forest corporation and peasants, the state's interest in the control over and exploitation of forest resources and related institutional arrangements. The latter granted the forest corporation the sole responsibility over state forest territories, while the spatial responsibility of watershed management authorities at both the scale of the watershed and the various administrative levels ended at the border to the state forest. Furthermore, the basin-wide watershed management authorities responsible for upland conservation were established under the responsibility of the Ministry of Forestry, which in turn has close ties with the state forest corporation.

6.2.3 Upland Conservation: Simplistic Political Discourses and Top-Down Approaches

While degradation within state forest territories was not adequately addressed, substantial resources were directed into upland conservation measures targeting small holders' private land all over Java. One hotspot of upland conservation efforts was the catchment area of the mangrove-fringed Segara Anakan lagoon. In this region, upland degradation was seen not only as a cause for the sedimentation of irrigation channels, seasonal flooding and dry season water shortages in the newly established ‘rice bowl areas’ of the lower river basin but also as a threat to the unique ecosystem of the estuarine Segara Anakan lagoon and the livelihoods of its residents (White et al. 1989; Olive 1997; Yuwono et al. 2007).

The causes of these problems were mainly attributed to the population in the upper catchment area with governmental organisations and experts pointing to population pressure, poverty and unsustainable upland farming as major issues. This framing of erosion and sedimentation processes as an environmental crisis caused by poor upland farmers provided a clear-cut narrative for the joint interest of international donor organisations, the Indonesian government, the forest administration and the state forest corporation to take collaborative action. Supporting poor upland peasants in terracing their land and encouraging them to plant more trees would contribute to reducing poverty, to inducing development in economically marginalised upland areas and to environmental conservation (ADB 1996, 2006; Tim Koordinasi Wilayah 1989; USAID 1985). These were shared interests of the government and international donor organisations and were in line with international environmental and development discourses. Enhancing the livelihoods of forest margin people and encouraging them to plant ‘their own’ trees also befitted the interest of the forest administration and the state forest corporation since it might help to reduce the pressure on state forestland. This coalition of actors with converging interests set a ‘machinery of discourse and management interventions’ in motion that has dominated watershed governance till today. Upland farmers’ private plots were discursively framed as major source of erosion; experts set up experimental and demonstration farms and developed agricultural extension packages; tree planting programmes were initiated, and farmers received training and seedlings. Departing from first pilot projects in the frame of the Citanduy I project in the late 1970s, these activities have been carried out for decades in the catchment area of the Segara Anakan lagoon and all over Java both in the frame of development projects, funded, among others, by USAID and ADB, and in the frame of the national conservation and regreening programmes (ADB 2006; USAID 1984; Interviews, May 2008–May 2011).

In both cases, planning and implementation were mainly under the responsibility of the Ministry of Forestry, which, after it had split from the Ministry of Agriculture in 1983, became one of the most powerful ministries. This constellation helps to explain why watershed conservation became synonymous with the enforcement of professional state forest management by the state forest corporation and with terracing and tree planting programmes on farmers’ private land, with the latter along the way potentially supporting the first.

That this ‘machinery of discourse and management interventions’ has barely been challenged for decades has to do with the dominance of a repressive, centralistic state with forcibly coordinated command-and-control structures and with development experts and Indonesian scientists being engaged as part of the machinery to contribute to the preparation and implementation of predefined interventions rather than to critically question them. It has to do with long-lasting simplistic assumptions regarding the role of professionally managed production forests for watershed conservation in Java and with internationally circulating, though questionable, discourses about the destructive hydrological impacts of marginalised upland farmers' cultivation practices in other parts of the world (see Ives and Messerli 1989; Forsyth 1996; Forsyth and Walker 2008). Ecological discourses, once formed and appropriated by a powerful actor coalition in line with their political interests, can serve as normative foundations for political action and effectively confine boundaries of knowledge production or render critical scientific inquiry seemingly unnecessary – a phenomenon conceptualised by Hajer (2000) as 'problem closure'.

More participatory approaches to forest and watershed management would likely have provided opportunities for discussing, investigating and addressing various other causes of lowland and lagoon sedimentation, including the state forest corporation's management practices, the underlying sociopolitical causes of forest degradation, considerable historical land use changes in the nineteenth and early twentieth century that were triggered by colonial exploitation, the digging out of slope toes by farmers to expand their wet rice fields and, last but not least, a series of volcanic eruptions within the past two centuries. This would likely have resulted in management approaches addressing watershed issues more effectively and efficiently since the entire range of drivers, partly including the historical courses of their development, would have been addressed according to their actual relevance.

More participatory approaches would also certainly have enhanced the effectiveness of the long-lasting investments in the frame of the conservation and regreening programmes. These were later judged to have failed to raise farmers' awareness for issues of erosion (ADB 1996) and were generally 'improperly carried out' (ADB 2006: 8). Despite the fact that the upland interventions obviously considerably increased the portion of upland dryfields being terraced, they in fact seem to have failed to reduce downstream sediment yields (Diemont et al. 1991; Purwanto 1999). This has been attributed to various reasons, including, among others, procedural shortcomings and site-insensitive delivery of one-type-fits-all upland agricultural packages (USAID 1985; Purwanto 1999). These problems can mainly be ascribed to non-participatory, mechanical approaches, inflexible top-down oriented decision-making structures and rigid regulations. The types of terraces farmers could construct were too restricted and partly locally not adapted, and the time slots for completing the work were inflexible. Not the farmers or communities but state authorities in collaboration with state-run nurseries decided which trees farmers could plant at what time (Interviews, November 2009–May 2011). Exertion of pressure and authority was reportedly not uncommon. In some cases, farmers were made to adopt a variety of different tree crops over time which they were told provided great economic opportunities but which finally failed, partly causing farmers to return to annual crops. For instance, the promotion of clove trees and subsequently declining clove prices, both related to business interests of the President Suharto's family, caused farmers to experience economic losses (Interviews, November 2009, February 2011).

The upland conservation and regreening programmes have raised the proportion of terraced agricultural land and substantially increased tree cover in selected sites. However, their overall contribution to watershed conservation has remained very limited as measured in terms of effort invested over decades.

6.3 Political Upheaval and Environmental Crisis

The revolutionary political upheaval in Indonesia that started in 1997 and involved the fall of Suharto's New Order Regime triggered environmentally disastrous developments but, at the same time, provided scope for the emergence of new, possibly more sustainable modes of watershed and forest governance. Environmental destruction was not only a result of chaotic upheaval and lawlessness, but itself has to be seen as an integral part of the rebellions against the state and its forest corporation, against their hegemonial ecological discourses, against coercive forest management and against decade-long repression and injustice. The environmental destructions and the subsequent and still ongoing rearrangements and renegotiations of political structures, of the configuration of actors, their levels of power, their political room for manoeuvre and their strategies have provided opportunities for *or* utterly necessitated the search for new management approaches.

The nationwide turmoil after 1997 provided opportunity for or was rather partially driven by large-scale upheavals of peasants, questioning established patterns of access to and control over forestland. The linking of local peasants' struggles with the emergence of national-level revolutionary forces brought land reform movements, which had been widely eradicated in 1965 and which could persist only as underground movements thereafter, back into the political arena (Peluso et al. 2008). After years of repressive modes of state forest management, a powerful countermovement of villagers spearheaded by influential individuals, partly including staff members of the forest corporation itself, used the opportunity of social tumult during the political transition after 1997 to plunder the state forests and therewith challenge consolidated power structures and get access to and control over resources they had been deprived of for decades or centuries. Peasants' re-appropriation of forestland, or what is commonly referred to as 'illegal logging', abruptly and forcibly shifted the control over forest resources from the state forest corporation to the people, altered the power relations between peasants and the forest corporation, and left behind critical, erosion-prone land. It needs to be remarked that illegal logging was not only driven by 'formerly repressed peasants' but was partly coordinated and accelerated by influential, wealthy individuals from towns or other regions, who provided chainsaws, trucks and safeguard and used the opportunity of political upheaval for their personal gain (Interviews, November 2009, February–May 2011). However, village residents in different parts of the catchment area of the Segara Anakan lagoon stressed that everybody participated and that illegal logging and occupation had not only been driven by personal economic hardships or the prospect of economic gain but that it had been a strategy 'to demonstrate who had the power' (ibid.). The devastated state forests are to be seen as a product or manifestation of decades of repressive government, coercive forest management and injustice.

The roots of these injustices partly go back to the late nineteenth and early twentieth century when the state forest areas were demarcated by the colonial forest administration and traditional uses started to be increasingly discriminated (Peluso 1992). In other cases, concrete land conflicts between peasants and the state forest corporation go back to evacuations of entire villages in the 1950s in the wake of political unrest triggered by the Darul Islam, a radical Islamic movement fighting for Indonesia to become an Islamic state. Following dubious state-organised land swaps, which left many of the displaced people landless, the villagers who returned to their land were forcibly evacuated in 1965 to make way for the expansion of state forest territory (Interviews, February–May 2011). Other land conflicts go back to disappropriations of peasants' land by plantation companies (ibid.). In all cases, the alliance of state authorities, companies, police and military effectively oppressed the displaced until 1997, leaving no room for resistance. Since then, the displaced have returned, cleared the forest, occupied the land and struggled for land titles (ibid.). My case studies suggest that as long as these land conflicts are not resolved, the land remains prone to erosion since farmers tend not to invest into soil conservation measures without any long-term perspective.

Decades of repressive forest management, which allegedly aimed at watershed conservation but in fact had always been detrimental to conservation goals, have finally resulted in massive large-scale forest devastation and hence watershed degradation. Erosion, landslides and more severe floods at the scale of sub-watersheds since the late 1990s have been the obvious consequences.

6.4 Towards New Modes of Forest and Watershed Governance

This ecological devastation together with the political transformations in Indonesia since the late 1990s has opened windows of opportunity for transitions towards new, possibly more sustainable modes of watershed and forest governance. These still ongoing transitions are driven by or respectively involve major reformations of the overall political system, altered power relations and 'new' actors entering the political arena. But they also reflect a paradigm shift that is related to learning effects from experiences with previous watershed and forest governance approaches and to influences of political and scientific discourses and societal developments on larger (international) scales. There has clearly been a shift from non-participatory, state-dominated, top-down oriented repressive command-and-control approaches towards more decentralised, participatory, incentive-based approaches. And there are clear signs of developments towards a better balance between the state's and the corporate's claims of sovereignty over production and conservation on the one hand and the local populations' resource use rights and needs and an acknowledgement of their factual substantial contributions to watershed conservation on the other hand. However, many of the achievements made are the results of fierce political struggles, and the persistence of established political discourses and some of the 'old' dominant centres of repressive power continue to pose major challenges.

Long-standing power relations and patterns of conflicting interest combined with habitual ways of thinking and communicating and related courses of action continue to affect the scope and outcomes of forest and watershed management approaches. Continued forest and watershed degradation, albeit less devastating than in the late 1990s and early 2000s, is a visible sign of still ongoing political struggles for and conflicts over new modes of governance.

Based on a number of governance and management issues encountered in a series of case studies in the catchment area of the Segara Anakan lagoon between 2009 and 2011, this section explores both transitions and patterns of persistence in watershed and forest governance since the late 1990s and discusses related achievements and challenges.

6.4.1 'New' Actors and Actor Coalitions: New Discourses and Management Approaches

With the fall of Suharto's New Order Regime and the following democratisation and decentralisation of the political system, the state rendered its sole sovereignty of interpretation and decision-making, thereby providing room for various previously unheard, excluded or oppressed actors to contribute their views to sociopolitical discourses, to influence decision-making processes on, form networks and take coordinated action, and to struggle for their rights and push their interests. Peasants, who had previously mainly been seen as passive recipients or implementers of interventions, have now enhanced opportunities to participate in decision-making at the community level, to have their interests represented by village heads, who are not necessarily an extended arm of the central government, and to form farmers' groups that are not anymore coordinated and brought into line by the central government (Interviews, November 2009, February–May 2011).

Such farmers' groups with support from non-governmental organisations (NGOs), many of which are run by or linked with groups of university students, collaboratively struggle for rights over land that was disappropriated in the 1950s and 1960s, that is not properly managed by plantation companies, or whose tenure is renegotiable due to expiring long-term leaseholds. Corresponding legal processes drag on over many years, but in a few cases, farmers have in fact already received land titles (Interviews, November 2009, February–May 2011). Particularly interesting with regard to the formation of new modes of forest and watershed governance are the local management plans that farmers with support from the NGOs have set up for disputed land in some areas and that challenge the long-standing simplistic watershed discourses. For example, Serikat Petani Pasundan (SPP), a leading farmers' organisation in West Java (see Rachman 2004), has in the process of claiming former plantation land in one part of the catchment area developed spatial plans for this land, with land partitions for all families of the surrounding communities and conservation-oriented management rules (Interviews, April 2011). The latter assign steep slopes to conservation and mixed forests and valley floors to terraced wet rice fields. These locally based management plans replicate the soil-conserving land use pattern that has already prevailed on small holders' private land for decades and challenge the long-standing discourse that claims small holders' allegedly unsustainable cultivation practices as main cause of watershed degradation. In fact, in contrast to state forests, a large proportion of small holders' private land in the lagoon's catchment area with its combination of terraced wet rice fields and stratified mixed forests, managed by selective logging rather than clear cuts, could serve as a model for watershed conservation. Acknowledgement of these spatial realities is an important basis for more targeted interventions. Hence, dismantling the long-standing misleading discourses is an important contribution towards new, more participatory and more sustainable modes of watershed governance. Initiatives of new actor coalitions like the one described above contribute to that.

The transformation of a repressive, centralistic state, claiming sovereignty of interpretation, into a more open political arena with various previously unheard actors being able to form networks and build coalitions provides new opportunities for sustainable watershed governance. From political processes previously excluded villagers have built networks with NGOs and urban-based student activists. The bundling of different experiences, competences and spheres of influence in such networks gives the participating actors the power to effectively push their views and interests, to question established discourses and to develop alternative, more participatory and locally based management approaches. Their links with broader level organisations, such as the Consortium for Agrarian Reform (Konsorsium Pembaruan Agraria, KPA), allow them to exert political influence on the national level (Afiff et al. 2005). As new emerging nodes of power and influence, such networks expand into the spaces opened up by the fragmentation of a long-standing, state-dominated, hierarchically integrated power apparatus.

Other emerging nodes of influence involve in addition to NGOs private sector organisations and universities and span various levels, including the international. In one part of the lagoon's catchment area, Bumi Hijau Lestari, an Indonesian nonprofit foundation established by furniture manufacturers from all over the world, contributes to increase tree cover on farmers' private land. The initiative, which focuses on Central Java, is similar to the regreening programmes that were run by the state for decades, but leaves more flexibility to farmers. The latter can choose between various species, and the foundation then provides seedlings and fertiliser and in collaboration with the Bogor Agricultural University organises workshops to teach farmers how to produce their own organic fertiliser and how to take optimal care of the trees. The farmers can freely choose when to harvest their trees and where to sell them but are encouraged to use the foundation's funding partners as marketing channel. Asked for a comparison with the previous state-run regreening programmes, village heads and farmers highlighted the better quality of seedlings and training and the flexibility and transparency. They also felt to play a more active role throughout the entire process (Interviews, November 2009).

It appears that a foundation that links furniture manufacturers from all over the world, who, driven by the social and environmental awareness of their customers, want to demonstrate their corporate social responsibility, with farmers, village heads and a university runs an alternative regreening programme without any involvement of the state and obviously with better results than the previous state-run programmes. However, in spite of being well accepted and successful at the local level, the initiative does not explicitly target the basin-wide hotspots of watershed degradation. Driven by the mission to contribute to social and environmental sustainability at the localities of timber production, the initiative does not necessarily select the sites of intervention according to degradation levels. In this context, linkages between the new network and the long-standing state-run watershed management authorities – i.e., linkages between newly emerging and long-standing nodes – might possibly be promising. In such a constellation, the state authorities could facilitate the smoothly running programmes of the foundation by contributing expertise on the hotspots of watershed degradation.

6.4.2 'Old' Structures and Actors in Transition: New Approaches

Besides creating space for 'new' actors and actor coalitions to emerge as new nodes of influence and power in an increasingly open political arena, the national political transformations have also at least to some extent reshaped the previous centralistic, hierarchical, top-down oriented state apparatus into a more fragmented network. Ambitious efforts towards decentralisation involved major shifts of power and authority between all levels from the national to the village level, with district governments emerging as important nodes of power and influence and communities gaining more autonomy (Resosudarmo 2005).

These transformations provide a basis for district and sub-district authorities in charge of natural resource management to play a more central role as regional facilitators rather than implementers of top-down oriented programmes, thereby providing space to experiment with more participatory, incentive-based approaches. Such developments are supported by the Ministry of Forestry. Following a recent replacement of the head of the ministry's Directorate for Land Rehabilitation and Social Forestry (Direktorat Jenderal Rehabilitasi Lahan dan Perhutanan Sosial, RLPS), funding for the long-standing regreening programme, which has briefly been outlined in Sect. 6.2, was phased out. Partly in collaboration with the Ministry of Agriculture, funding is now provided for more participatory programmes, which are facilitated by the district-level authorities (Interviews, February–May 2011).

The concept of these programmes establishes communities as central loci of initiative, decision-making and action. For example, in the frame of a recently launched smallholder rubber programme (Karet Rakyat), the participating farmers not only decide how many trees they would like to plant in which areas at what time but are supposed to organise the entire process themselves. They are encouraged to establish farmers' groups, who develop plans for village nurseries and planting programmes and negotiate with suppliers of material. Extension personnel from the district and sub-district offices, who are responsible for agriculture, forestry and watershed conservation, facilitate these activities and provide support in finding appropriate marketing channels. Once the farmers' group has a thorough concept, it can apply for public funding via the district administration, which would then contribute one third of the total investment costs. The head of a district extension service in the lagoon's catchment area emphasised the importance of his role as facilitator in the process since this would, among others, help to prevent the formation of exploitative farmer-trader relations. In this context, he also planned to invite investors for building a rubber processing factory in the area, thereby offering the possibility for farmers to directly sell their produce to the factory without relying on traders (Interviews, February 2011).

The example of the Karet Rakyat programme and its comparison with the previous regreening programmes described in Sect. 6.2 impressively illustrates the changing roles of some state authorities from being decision-makers to being facilitators. In this case, a district authority, which had previously as part of a hierarchical command-and-control system been responsible for implementing the blueprints designed by the ministry at the local level, has now become a facilitator of village-based initiatives, providing expertise if needed, channelling funds from the ministry to the village to support these local initiatives and establishing links with the private sector. In other words, some of the fragments of the previous state-dominated, hierarchical structure have been transformed into a loose network encompassing farmers together with state and market actors. Where this works well, it can be a prime example for environmental governance that bridges dichotomies between state and market approaches and between conservation and development and that is mainly based on incentive rather than regulation and command-and-control. First experiences with the new approach in the lagoon's catchment area are encouraging; further developments need to be seen.

Clearly, the performance of such new approaches heavily depends on the persons involved. Throughout all levels of the state administration, new reform-oriented, innovative forces are competing with long-standing conservative structures that hold on to top-down oriented command-and-control approaches. The latter combined with intransparent, shady procedures can easily impede initiative and trust at the local level. Part of the government of the same district whose Karet Rakyat initiative has been outlined above was recently under investigation over diverting village development funds for their personal gain (Interviews, February 2011). But also local capacity plays an important role for paradigm shifts and new approaches to materialise in successful natural resource management. For example, lacking local capacity to understand formal contracts and initiate legal processes provides more powerful actors, such as the state forest corporation, room for manoeuvre thereby obstructing transitions towards more participatory approaches and makes villagers extremely vulnerable to exploitation by thugs, who pretend to support communities, but only collect large amounts of money before they disappear (Interviews, February 2011). Understandably, the villagers' historically rooted lack of trust in state authorities and their habitual strategy of passive resistance towards state-led interventions are important challenges for any initiative. Changes in mindsets and communication patterns on both sides – among representatives of state authorities and among the people – are perhaps as relevant as the concepts of new management approaches. Furthermore, developing appropriate local level decision-making structures and processes is a learning process that requires time and also heavily depends on the persons involved.

The latter aspect plays an important role also on higher levels. Single-state authorities, such as the Ministry of Forestry, can be conceived as networks comprising rather conservative and rather reform-oriented elements, each of them being part of broader networks spanning all scales from the local to the international. Aiming at promoting new governance and management approaches, such as market-based instruments, international donor organisations, such as the United Nations Development Programme (UNDP), attempt to connect with and support some of the (more innovative) elements within the ministry rather than trying to overturn the entire structure (Interviews, January 2010). International organisations, selected elements of the ministry, national NGOs and various regional and local actors in the project sites then appear as a network, developing and promoting new governance approaches.

These developments have been enabled by broader political transformations in Indonesia and have been influenced by both learning experiences and international paradigms. In case of the state-run regreening programmes, insight into the limited effectiveness of the previous command-and-control approaches has contributed to the development of new, more participatory, locally based approaches. In fact, the concept of the previous regreening programmes as such had already been incentive based, but the way of implementation gave the programmes the character of regulatory top-down interventions, which often were not in line with local needs and interests and therefore not particularly successful. Last but not least, the international paradigm shift towards participatory, incentive-based approaches, integrating NGOs, private actors and markets, obviously contributes to the formation of new modes of watershed and forest governance in Indonesia through interlinked networks and flows of information and ideas.

6.4.3 Persistent Nodes of Power: Between Repression and New Modes of Governance

The discussion of transformations towards new modes of watershed governance in the previous two subsections has focused on developments in conjunction with management approaches for small holders' private land while excluding the further development of the state forest areas, which have been exposed as major hotspots of degradation in Sects. 6.2 and 6.3. What has become of the environmentally destructive conflicts over forest resources between villagers and the state forest corporation? Have new modes of forest and watershed governance transformed the state forests from political battlefields into sustainably managed woodlands?

It appears that state forest areas continue to be a hotspot of degradation and that watershed discourses and related interventions, in spite of converse spatial realities and alternative framings by new actor coalitions that have been discussed in Sect. 4.1, generally continue to be focused on small holders' private land. The notions of 'unsustainable upland farming' and 'population pressure' as major watershed issues continue to be recited like mantras by both state authorities and experts. These discourses are obviously an effective legitimation for non-action with regard to degradation in state forest areas. In fact, long-standing institutional areas of responsibility continue to limit the scope of action of the state authorities assigned to watershed management. The responsibilities of watershed and province-level planning authorities as well as district-level administrations still widely end at the border to the state forest areas (Interviews, May 2008, January 2010, February 2011). Hence, the sustainability of state forest management widely depends on the outcomes of renegotiations between the forest corporation and peasants over resource access and control.

The major result of these renegotiations to date is community forestry programmes. In the frame of these programmes, the forest corporation grants peasants the right to use forestland for food crop production during the first years after trees have been replanted and later shares some proportion of the benefit from harvesting the trees with the communities. This benefit sharing is supposed to promote the communities' sense of ownership, thereby reducing the risk of illegal logging. In fact, the state forest corporation attempts to confer the responsibility of safeguarding the forest upon the communities (Interviews, February 2011). Compared with the situation prevailing until the late 1990s, the community forestry schemes appear as promising new modes of forest governance, which have replaced coercive, state-dominated approaches by participatory, community-based approaches.

However, in fact, the programmes are not yet particularly successful in many parts of the lagoon's catchment area (land use mapping and interviews, May 2008–May 2011). The underlying causes are related to the corporation's domination of the entire process and its remaining as central locus of decision-making. This together with historically grown tensions between the corporation and peasants, involving a pronounced lack of trust and habitually conflicting patterns of communication and action, clearly undermines the sustainability of the community forestry programmes. While the programmes pretend to be participatory bottom-up approaches, they are in reality based on standardised blueprint contracts designed by the forest corporation to be signed by the community, and the process of establishing the community forestry groups and related structures is dominated by the corporation rather than based on participatory processes. All decisions regarding the planting, management and harvesting of the trees are made by the corporation. Peasants have no possibility to participate in decision-making with regard to the choice of species and planting or harvesting times. For example, they have no chance to effectively communicate their concerns over the corporation's practice of vastly expanding the area planted with pine, which, according to peasants in many areas, undermines water supply for irrigation and causes landslides. Any collaboration between the corporation and peasants is considerably complicated by historically grown patterns of distrust and confrontation. Peasants, for historical reasons, deeply distrust the corporation and see it as a repressive apparatus, as an enemy. The corporation's staff members in turn often exhibit attitudes of predominance towards peasants, whom they often view as uneducated, 'backward', ignorant and unable to sustainably manage forest land.

Consequently, the community forestry programmes have not been very successful in generating a pronounced 'sense of ownership' over state forest territory among peasants. As result, the trees, which are to be considered as property of the corporation, are partly not well maintained; tree cover is reduced, and cultivation of annual crops is understandably not accompanied by soil conservation measures.

Taking a historical and actor-based perspective, it appears that the design of the community forestry programmes was perhaps not primarily driven by the sincere motivation to experiment with and implement new participatory, community-based modes of forest governance for the sake of environmental and social sustainability but was a strategy of the forest corporation to regain control over its territories and its trees. Following years of large-scale illegal logging in the early phase of the political transformation, the corporation had no choice but to make concessions to peasants. Means of obvious coercion, as regularly used before the political transformation, would not be legitimate anymore. Thus, granting peasants limited cultivation rights, and sharing benefits with them was the preferred strategy to regain control over territory and trees.

The mere existence of community forestry programmes as new, seemingly participatory modes of forest governance should not hide the fact – and the problems outlined above illustrate this to some extent – that the state forest corporation remains as a node of enormous power and influence, as a part of the former repressive state that has not been as thoroughly reformed as government bodies. Large parts of the forest corporation widely remain as a centralistic, hierarchical, top-down oriented, intransparent apparatus.

The commitment of the corporation's upper management to introduce reforms and its political rhetoric – last but not the least motivated by its long-standing determination to seek certification by the Forest Stewardship Council (FSC) – are encouraging, the more so as certification would create additional international pressure (Interviews, January 2010, February–May 2011). But at the same time, the political rhetoric may hide both the enormous challenges ahead and some highly problematic nodes of power and repression. For example, the collaboration between the forest corporation and peasants in the frame of the community programmes is not only – as described above – complicated by historically grown patterns of deep distrust and a pronounced imbalance of power between the two parties. But authorities of the forest corporation continue to maintain close relationships with police and military forces. In these networks, fractions of the former hierarchically integrated repressive alliance of all state authorities live on and continue to exert influence, control and pressure – less obvious than before since the power is not organised in well-defined hierarchal levels anymore but is less visibly embedded in more complex networks (cf. Flitner and Görg 2008). Disputes between the forest corporation and peasants are not directly dealt with by the corporation's own forest police anymore but by regular police forces. But close (personal) networks allow the corporation to promptly deploy the police. Peasants claiming state forestland for historical reasons and waiting for legal processes to proceed have fear whenever approached by an unknown person since they 'never know whether it is a spy acting on behalf of the state forest corporation or the police' (Interviews, February–May 2011). District-level representatives of the military are members of the communication forums within the community forestry programmes (ibid.). Not easily visible, such nodes of power substantially contribute to the outcomes of new forest and watershed governance approaches. Repression lives on in the midst of an increasingly open political arena that is being reshaped by political transformations and that provides scope for promising developments towards new modes of governance.

6.5 Conclusion

Analysing the interplay between actors, their interests, their power relations and their discursive constructions of environmental issues and the emergence of this interplay over time sheds new light on present modes of environmental governance. Understanding current management approaches as an outcome shaped by present political structures and the processes of their emergence, by long-standing struggles over political power, resource access and control and by sociopolitical discourses and problem-solving paradigms provides a context that helps to explore the underlying causes of their 'success' or 'failure'. This context opens up perspectives on the achievements of and future challenges embedded in newly emerging modes of governance, which would possibly remain hidden if one would adhere to the assumption that the various actors and approaches necessarily aim at solving environmental issues (cf. Mayntz 2001).

Current modes of watershed and forest governance in Java have to be seen, among others, as an outcome of the historically emerged interplay between long-standing struggles over the access to and control of forest resources, simplistic sociopolitical discourses regarding the causes of upland degradation, international development and environmental management paradigms and broader revolutionary political transformations in Indonesia. The latter have triggered environmentally disastrous developments by transforming many state forest areas into political battlefields but have also opened windows of opportunity for transitions towards new, possibly more sustainable modes of watershed and forest governance. These transitions involve shifts from non-participatory, state-dominated, top-down oriented repressive command-and-control approaches towards more decentralised, participatory, incentive-based approaches.

Building on Burries et al. (2005) and Flitner and Görg (2008), the analysed transitions can be conceived and conceptualised as gradual shift from a very hierarchical style of government to networked or nodal governance. They are characterised by an increasing fragmentation of the previous centralistic, top-down oriented state apparatus into a more fluid network of nodes of power and influence. While some fragments of the 'old' hierarchically integrated structure continue to persist as powerful nodes of repression, others are being transformed into nodes of innovation, linking with newly emerging networks and thereby advancing the transition. Exploring the connections between the previous 'old' structure, i.e., the vertically integrated alliance of repressive state authorities, and the present nodes of power and influence exposes some of the major barriers to innovation and illustrates that some elements of the 'new' modes of governance might be more infused with 'old' repressive elements than visible at first glance. In fluid, diffuse and partly informal networks, which may span not only a multitude of governmental and non-governmental organisations but also individual actors on various scales, the localities of power and influence are less obvious than in formal, well-defined hierarchical levels (cf. Flitner and Görg 2008).

Indonesia's political transformation since the late 1990s, involving democratisation and decentralisation, has created space in an increasingly open political arena for new actors and actor coalitions to emerge as new nodes of influence. In the context of watershed and forest governance, peasants' organisations and village heads together with NGOs, private sector organisations and universities form networks, bundling different experiences, competences and spheres of influence, and spanning various scales. Expanding into the spaces opened up by the fragmentation of the long-standing, state-dominated, forcibly coordinated power apparatus, these networks increasingly question established discourses and develop new, more participatory, locally based management approaches. Some 'fragments' of the state administration – and these can be both individuals or entire authorities – also experiment with, support or establish new, more participatory, locally based management approaches. Both the newly emerging networks and these 'innovative' reform-oriented 'fractions' of the state administration appear to be the promoters of new modes of governance. Linkages between both – i.e., constellations where state authorities link with community-level actors, NGOs and private sector organisations – appear to be particularly promising. If these newly established approaches work well over the long term, they could be prime examples of environmental governance that bridge dichotomies between state and market approaches and between conservation and development and that are based on different types of incentives rather than on coercion and command-and-control. However, other 'fragments' of the former repressive power apparatus – including large parts of the state forest corporation, which has close ties with the police and military – continue to persist as powerful nodes of influence. Within an increasingly open political arena that is being reshaped by political transformations, continued forest and watershed degradation is the visible sign of ongoing political struggles for and conflicts over new, more sustainable modes of governance.

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Chapter 7
Watershed Development, Decentralisation and Institutional Change: Insights from the Mechanism Design Theory

Jayanath Ananda

7.1 Introduction

Rain-fed and semiarid areas of India are characterised by poor resource endowments, erratic rainfall and widespread poverty.[1] Watershed[2] development (WSD) programmes play a pivotal role in supporting rural livelihoods and reducing poverty in these areas. About 54% of agricultural lands in India are rain-fed (Reddy et al. 2009; World Bank 2008) where soil fertility and water scarcity are major constraints for agricultural production and the average productivity of dryland agriculture is low (Bouma et al. 2007). Watershed development has also been considered as an important rural development strategy in arresting environmental degradation in India and other developing countries (Kerr et al. 2000). The main premise in WSD has been the enhancing of the resource base in order to increase agricultural productivity which in turn would alleviate poverty and provide a more equitable distribution of income in rain-fed and semiarid regions.

There are several unique aspects to watershed development programmes when compared to other rural development programmes. Watersheds rarely correspond to human-defined boundaries. They are functional units established by physical relationships where upstream land use affects the downstream activities. Hence, the costs and benefits of watershed activities are unevenly distributed among upstream and downstream areas. Moreover, a watershed holds multiple, interconnected natural

[1] About 42% of the population in India lives below poverty line (US$1.25 a day) (World Bank 2008).

[2] Watershed is a topographically delineated area that is drained by a stream system.

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resources: soils, water (surface and groundwater) and vegetation. Impacts on one resource invariably affect the status of others.

Over the last two decades, the Government of India has increased its investments in rain-fed areas. Despite ongoing government investment in the order of US$500 million annually, the watershed development projects have delivered disappointing results to date (Samra and Sharma 2009). Although some programmes managed to increase the water availability, reduce soil erosion and increase cropping intensity, rural employment and incomes, there is evidence of rapid depletion of groundwater tables and fragile ecosystems where soil erosion continues. Overall, the success has been sporadic, intermittent and short-lived,[3] and returns to investment from WSD projects have been low.

The above problems have prompted a closer examination of the watershed governance arrangements which have been subject to much debate in recent times. Lately, many WSD programmes in India have taken a participatory approach, where state governments share costs and benefits with local communities. Such participatory approaches are not without significant institutional challenges. One of the main challenges facing WSD programmes is to coordinate investments across multiple scales and internalise the spillover effects within the watershed equitably. Moreover, in the past, the absence of an effective 'exit strategy' for governments and sustaining the project after the initial funding rounds has become a difficult task. These problems highlight the importance of establishing and supporting new institutional configurations to coordinate watershed development activities.

This chapter explores the institutional configurations of WSD programmes from an institutional design point of view. Using a theoretical framework based on the mechanism design theory (MDT), it analyses the WSD governance institutions in India providing insights on the institutional design and associated incentive incompatibilities. The remainder of this chapter is organised as follows. The next section provides an overview of the institutional apparatus that attends to the WSD governance in India with special reference to Andhra Pradesh. Section 7.3 presents a brief overview of the MDT highlighting its main tenets. Using the theoretical framework, Sect. 7.4 analyses the WSD programmes in India identifying several institutional features that may influence the institutional performance of WSD. Section 7.5 provides some concluding comments.

7.2 An Overview of Watershed Governance in India

The WSD programmes are implemented in over 300 districts of India under several flagship initiatives. The districts were chosen on the basis of their environmental, social and developmental status, and high priority is accorded to semiarid, low-rainfall

[3] Evaluation studies to support this finding include the studies conducted by the Indian Council of Agricultural Research (ICAR), National Remote Sensing Agency (NRSA), Ministries of Agriculture and Rural Development and Planning Commission and Reddy (2000).

Table 7.1 Details of WSD projects sanctioned and funds released from 1995–1996 to 2007–2008

Name of scheme	No. of project sanctioned	Area covered (lakh ha.)	Total funds Ind. rupees crores (US$ million)
Drought-Prone Areas Programme	27,439	130	2,838 (579)
Desert Development Programme	15,746	79	2,103 (429)
Integrated Wastelands Development Programme	1,877	107	2,798 (571)
Grand total	45,062	323	7,739 (1,579)

Source: Government of India (2008)
Note: Exchange rate as of 16 October 2011

regions with concentration of scheduled castes and tribes. For example, *Anantapur* District, an area selected for WSD projects, characterises hot and arid climate with erratic and unevenly distributed rainfall resulting in soil moisture stress, excessive evaporation and groundwater losses. Most WSD programmes are aimed at improving the quality of land resources through water and soil conservation. Three main WSD schemes have been implemented since the 1970s under different guidelines. Table 7.1 summarises the major WSD schemes implemented in India since the mid-1990s.

Andhra Pradesh, the fifth largest state in India with 23 districts, occupies the largest share of nationwide watershed projects implemented. One of the main WSD programmes in Andhra Pradesh included the Drought-Prone Areas Programme (DPAP), co-sponsored by the central and state governments with the objective of improving soil and moisture conservation, constructing water harvesting structures, afforestation and horticulture programmes. The DPAP has implemented projects over 5,000 watersheds, and it is the main driver of the watershed development programmes in Andhra Pradesh (Reddy et al. 2009).

In addition, Integrated Wastelands Development Project (IWDP), Desert Development Programme (DDP) and Joint Forest Management projects have been featured in the national programme. An important aspect of WSD programmes is its contribution to groundwater use in India. Over 80% of the farming community relies on wells for irrigation, and WSD programmes have contributed to this rise in groundwater use with improvements to the water table (Springate-Baginski et al. 2002).

7.2.1 Institutional Frameworks for WSD Projects

The institutional framework for WSD project implementation has undergone changes over time with the main focus being to steer away from a fragmented, single ministerial theme to a multi-theme approach. In line with India's three-tier governance model, the federal government (national government or the centre)

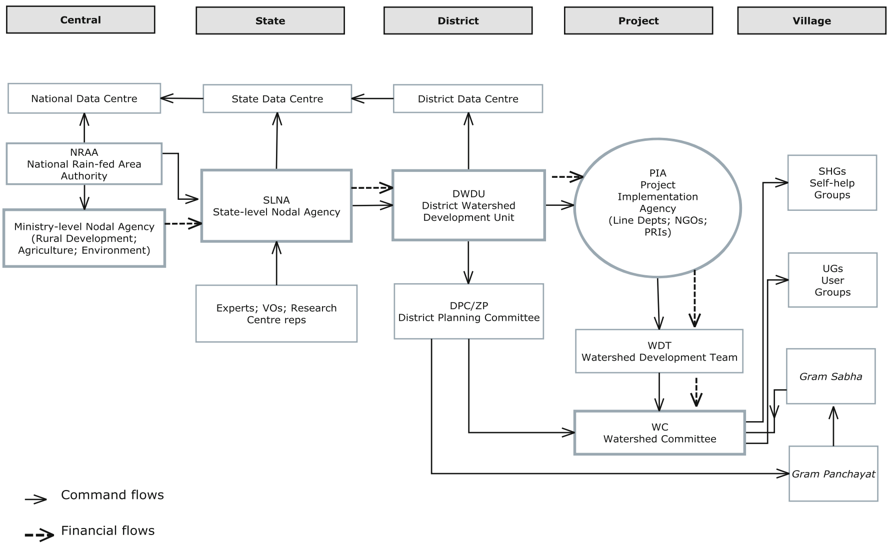

Fig. 7.1 Institutional structure for WSD implementation proposed by the Common Guidelines (2008)

formulates watershed development guidelines called Common Guidelines. The latest Common Guidelines were released in 2008 in collaboration with three federal ministries – the Ministry of Rural Development, Ministry of Agriculture and Ministry of Environment and Forests. These new guidelines have established dedicated institutions with multidisciplinary focus, delegated more power to states to oversee and implement WSD projects within their jurisdictions and provided additional financial assistance to dedicated institutions. The new WSD guidelines also took a cluster approach where geohydrological units are broadened to include additional watersheds in contiguous areas (Government of India 2008). The new institutional framework encapsulating the 2008 Common Guidelines is shown in Fig. 7.1.

Under the 2008 Common Guidelines, a new national-level institution – National Rainfed Area Authority (NRAA) – has been established to formulate overarching policies related to watershed development, undertake strategic planning and provide technical assistance including capacity building. A ministry-level nodal agency is set up as the operational arm of NRAA comprising multidisciplinary experts to facilitate the project management process. A national-level data centre with GIS capabilities manages data on watershed and land resource information.

A state-level nodal agency (SLNA) is responsible for facilitation and disbursement of WSD funds emanating from the centre. The SLNAs sanction WSD projects, maintain state-level data, provide technical support to district units and undertake monitoring and evaluation activities. More importantly, they approve project implementing agencies (PIAs). At district level, District Watershed Development Unit (DWDU) is established where the watershed is about 25,000 ha in size. They are responsible for identifying PIAs, providing technical support to PIAs, capacity building, maintaining district-level data and liaising with the District Planning Committees.

The project implementing agency (PIA) functions as the main implementing agency at the project level. The PIAs can take several forms including line departments, autonomous organisations under state/central governments, government institutes/research agencies, intermediate *Panchayats* and voluntary organisations. Selected PIAs will sign a contract or a memorandum of understanding (MoU) specifying well-defined outcomes with concerned DWDUs. Each PIA must form a Watershed Development Team (WDT) whose members are hired on contract to collaborate with technical experts at superordinate levels. The WDT will guide a Watershed Committee (WC) in the formulation of the watershed action plan which will constitute various user groups and self-help groups. The roles and responsibilities of WSD governance entities are summarised in Table 7.2.

7.2.2 Watershed Management and Decentralisation

After decades of top-down implementation approaches, watershed development programmes have evolved towards a participatory and decentralised management approach in recent times. The shift towards a participatory approach largely stems from the failure of the top-down approach. Figure 7.2 shows a generalised WSD institutional configuration before decentralisation at the district level. The usual convention is to create new institutions and assets, support new watershed technologies during the initial phase of the programme and over time, transfer management of these to local communities. It is envisaged that this participatory approach will lift the government's burden on operation and maintenance expenditures.

The 73rd amendment of the Indian Constitution provided an impetus for decentralisation by strengthening the local government, collectively called *Panchayat Raj Institutions* (PRIs), at district, block and village levels. These self-governing bodies have been given an expanded role in implementing WSD initiatives within a nested and decentralised institutional environment. *Zilla Parishad* (in Andhra Pradesh) is the district-level representation of the PRI system. *Zilla Parishad* has the responsibility for implementation of WSD programmes. They hold the ultimate power of financial and administrative matters. More importantly, they select project implementing agencies (PIAs) and approve WSD plans. There are many organisations such as government departments, NGOs, universities, research institutes and PRIs that are involved as PIAs for implementing WSD programmes (Fig. 7.3). Since the 73rd Constitutional Amendment on empowering local institutions, the role of PIA has been shifted largely to PRIs. However, in the case of Andhra Pradesh, the PRI involvement in managing WSD programmes at the village level has been minimal.

The next section outlines a theoretical framework to analyse WSD governance arrangements.

Table 7.2 Roles and responsibilities of WSD implementation

Institution	Roles and responsibilities	Membership
National Rainfed Area Authority (NRAA)	National-level coordination of WSD; strategic planning, capacity building and programme evaluation; maintains national data centre; provision of technical knowledge	Ministry of Agriculture (Chair), Ministry of Rural Development, Water Resources, Environment and Forest and *Panchayati Raj* and Planning Commission
Ministry-level nodal agency	Facilitates allocation and disbursement of funds from planning commission and other external agencies; coordination with line departments; prioritisation of watersheds for projects	Ministry of Rural Development, multidisciplinary experts
State-level nodal agency (SLNA)	Sanctions WSD projects, maintains state-level data, approves selection of PIAs, provides technical support to DWDU, prepares state plans, releases funds for implementation of WSD projects, monitors and evaluates state data centre	Chairperson nominated by the state government; representatives from NRAA, the nodal ministry and relevant departments; groundwater board; voluntary organisation representatives; experts from research centres
District Watershed Development Unit (DWDU)	Identifies PIAs in consultation with SLNA, coordinates with the District Planning Committee, signs a memorandum of understanding (MoU) with SLNA, technically supports PIAs, organises community training on WSD initiatives/technologies, allocates funds to PIAs	Project director, district collector, 3–4 subject matter specialists
Project implementing agency (PIA)	Responsible for implementing WSD projects: signs an MoU with DWDU, discusses, plans and implements activities proposed in the WSD project plan	Can include line departments, autonomous organisations under state or central governments, government institutes/research bodies, intermediate *Panchayats*, voluntary organisations (VOs)
Watershed Development Team (WDT)	Guides WC, SHGs and UGs, prepares resource development plans, common-pool resource management and equitable sharing	Four-member team (agriculture, soil science, water management, institutional building); women participation
Watershed Committee (WC)	Carries out WSD activities, liaisons with *Gram Panchayat*, maintains records including financial records	*Gram Sabha* elects the chairman and the secretary; ten members: WDT, SHG, UG and landless representation

Source: Common Guidelines 2008 (Government of India 2008)

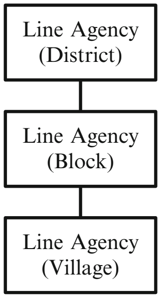

Fig. 7.2 The institutional structure (district level and below) of WSD before decentralisation

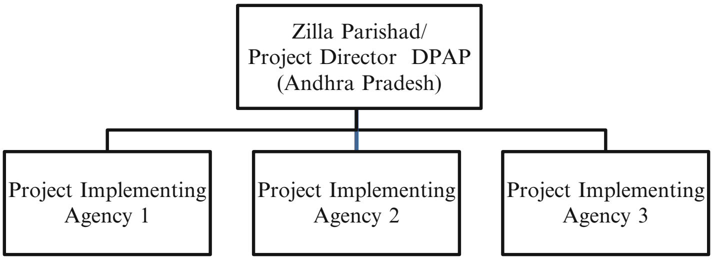

Fig. 7.3 Post-decentralisation institutional structure (district level and below)

7.3 The Mechanism Design Theory

An important feature in collective choice situations is that individual preferences are not publicly visible. Consequently, the information must be elicited. The extent to which the information revelation problem constraints the ways in which collective decisions can respond to individual preferences is known as the mechanism design problem (Čihák 2008). The mechanism design theory (MDT) (Hurwicz 1960) concerns the problem faced by a principal or planner in designing a 'mechanism' by which a set of agents with productive capacities or consumption needs will interact with one another to yield resource allocation outcomes. The term 'mechanism' refers to the institutions and 'the rules of the game' (who decides what, communicates with whom, in what fashion and how eventually allocations are made) that govern economic activities (Mookherjee 2005). Often, perfect information relating to the technology, productive capacities and agents' preferences are not available to the planner.

A major thrust in the information literature is about the dispersion of information among numerous economic agents. It is also at the core of explaining the failure of central planning (Hurwicz 1960). Moreover, there is a lack of incentives for economic agents to share their information truthfully with others, especially with the government. This is referred to as the incentive compatibility problem (Hurwicz 1972). In fact, Hurwicz showed that even with compatible incentives, an optimal outcome cannot be guaranteed because of private (asymmetric) information. On the one hand, a decentralised system requires less communication and information processing requirements compared to a centralised system. A decentralised system provides incentives to individual agents and aligns incentives with agents' motivations better compared to a centralised system. If the incentive costs are substantial, the choice between centralisation and decentralisation will involve a trade-off between incentive (costs) and benefits (communication, information processing). On the other hand, centralised systems are better equipped to deal with problems of externalities, public goods, increasing returns and distributional equity (Mookherjee 2005).

The revelation principle (Myerson 1979), a major branch of MDT, simplifies this allocation problem by calculating the most efficient rule of the game for getting people to reveal their private information truthfully (Čihák 2008). This implies that centralisation is at least weakly preferred to any decentralised system. According to the revelation principle, the central government could design optimal contracts to extract locally available information (Greco 2003).

7.3.1 Communication and Information Processing Costs

A strand of MDT literature that deals with the design of decentralised communication costs is loosely referred to as 'message space' literature. Although it had a limited impact mainly due to a high level of complexity and mathematical abstraction, costly communication and computation remain central to the theory of economic organisation.[4] Most models deal with optimal delegation[5] in simple hierarchical contracting with two productive agents whose communication is restricted only between adjacent layers. The main focus is on the problem of vertical control, and problems of horizontal coordination across different branches have received less attention (Mookherjee 2005).

Delegation beyond two productive agents or delegation in complex hierarchies is prone to problems of vertical control loss and coordination across horizontal branches. The effectiveness of alleviating communication problems depends on the ability of the superordinate entity (principal) to provide the subordinate entity (agent) with a high-powered incentive scheme. Delegation of production and contractual rights also creates moral hazard problems, as the agent's preferences are not perfectly aligned with those of the principal.

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[4] More recent works on this can be found in Radner (1993) and Mount and Reiter (1995).

[5] For a delegation to be optimal, certain conditions including the ability to observe the contract costs, top-down contracting and risk neutrality have to be met.

7.3.2 Assigning Governance Functions Across Multiple Scales

The governance arrangements of common property resources are often organised as decentralised, hierarchical structures. They also encompass multiple geographical scales. Mapping various governance functions across multiple scales and entities within a decision hierarchy is a challenging task. Undoubtedly, the success of decentralisation largely depends on the task assignment and execution. Therefore, identifying tasks or governance functions becomes the first step of any decentralisation process.[6]

The subsidiarity principle provides some limited guidance in this regard. Notwithstanding external resources, two variables are pivotal for mapping tasks to governance structures. First, the capacity[7] of the entity to carry out a given function is critical. Executing a task satisfactorily at a given level partly depends on sufficient access at that level to all dimensions of capacity. Moreover, although problematic, representation of all relevant actors who have an interest in the task also generally contributes to the successful execution of a given task (Marshall 2008).

Attempts have been made to develop a generalised model for allocating governance functions across a decision hierarchy, but these have not always proven successful. For example, Hurwicz (1973) presented a model of designing informationally decentralised systems thus:

> Two difficulties make the problem non-trivial: calculation and information transfer. First, consider the calculation of the maximising values for the variables of the problem. Assuming even that all the relevant information concerning the parameters of the problem is in the hands of the computing agency, this agency needs a well-defined computational procedure (algorithm) to find solutions. Even when there is an algorithm … it may be that the information processing capacity of any agency is inadequate. (Hurwicz 1973, pp. 4–5)

Not surprisingly, solving for an efficient optimal solution in this context usually proves too complex, if not impossible.

7.4 Analysis of Institutional Performance

Institutional innovations in WSD programmes in India have been supported by varied factors in various states. In Andhra Pradesh, the political autonomy of the state and the central government's willingness to embrace new modes of WSD governance have contributed to the institutional change process. Moreover, clear political

[6] The task selection for decentralisation can be approached from four main perspectives: constitutional, economic, managerial and social (Dollery et al. 2006).

[7] The term 'capacity' encompasses several dimensions including financial, physical, human and social capacities – including leadership.

leadership backed by the good governance and increased donor funding have also helped the uptake of WSD projects in Andhra Pradesh.

This section examines the performance of watershed development institutions based on the mechanism design theory principles. Before analysing institutional performance through a lens of MDT, it is important to reflect on the institutional design process for WSD projects in India. Not surprisingly, who gets to choose the mechanism is pivotal for determining outcomes. In the case of WSD in India, the mechanism design is still dominated by a top-down approach, as public involvement in designing the overall programme is weak. However, the Common Guidelines (2008) emphasise a participatory approach to watershed project implementation.

7.4.1 Information Requirements and Costs

Watershed governance entails significant information requirements. Pannell et al. (2009) identified some of the information costs pertaining to public investments in the natural resources and the environment. Similar information requirements apply to watershed development projects in India as well. They include the assessment of the current condition of the watershed, the main threatening processes (e.g. soil erosion), the current and expected damage to the watershed due to those threatening processes and the likely time lag between the treatment and impacts. Information on changes in current management practices and establishment of new watershed assets (e.g. water harvesting structures such as check dams, terraces for soil and water conservation) are also required. Most of this information is embedded in local institutions. Tapping to traditional institutions and local knowledge can enhance the effectiveness of WSD programmes.

The success of a WSD project depends on the likely reduction in damage if proposed intervention or management practices were implemented and the likely improvements in agricultural productivity and the poverty situation. Most WSD activities entail spillover effects where upstream activities impact on the downstream resource base. To design an optimal WSD plan for a region, the magnitude of potential spillover effects (positive and negative) from the intervention must be ascertained. Moreover, the likely rate of adoption by landholders, mechanisms to encourage adoption of the proposed works and devising appropriate financial incentives are pivotal. Finally, the administrative and political feasibility of the intervention including capacity constraints of the institutional forms involved need to be taken into consideration.

The revelation principle assumes that there are no communication and information processing costs. Put differently, the implicit assumption here is that local, state and central governments share the same objective and have no conflicts. Hence, there is no reason for local governments not to fully and truthfully reveal their information to the centre. However, self-interest and the political economy of governments and their subordinate organisations cloud this argument.

The governance structures at the federal and state levels (Fig. 7.1) have a higher technical capacity to process and interpret the watershed data when compared to structures at a lower level. In Common Guidelines of 2008 (Government of India 2008), the above information processing issues have been addressed to some extent, by establishing data centres at national, state and district levels. Notwithstanding vertical control and information sharing problems, these data centres can contribute to better institutional performance. That said, even at the district level, the gathering of information can be less than perfect due to asymmetric distribution of information and moral hazard issues. In a watershed development context, the mechanism design theory framework considers various stakeholders in a given watershed as economic agents (e.g. landholders) having specific objectives and risk preferences. They also hold information about the costs and benefits and production possibilities of various watershed development initiatives such as soil conservation measures. The agents (landholders) also have perfect information about the costs and benefits of, say, soil erosion structures of their land, and they might not truthfully reveal that information to the policy planner at the next superordinate level because of moral hazard reasons. Asymmetric distribution of information is regarded as one of the main causes of market failure (Alkerlof 1970). Moreover, the mechanisms to transfer information are less than perfect.

There are multiple 'mechanisms' or PIAs to implement WSD activities (see Sect. 7.2.1). With enhanced decentralisation, PRIs as proxies of DWDU can identify potential PIAs and outsource WSD activities. These new governance arrangements create institutional competition for WSD funding which can increase the rate of return to investment in WSD. However, there is room for adverse selection at the state level when selecting PIAs by the state-level nodal agency (SLNA). Although DWDU has been the responsibility to identify potential PIAs to carry out the WSD activities, the ultimate decision on which institution should be given the responsibility to implement the intervention on the ground lies with the SLNA.

7.4.2 Incentive Incompatibility

The available evidence indicates that farmers show little enthusiasm for adopting WSD technologies. Inappropriate watershed technologies, high initial investment, high operation and maintenance costs are some of the reasons for low adoption of WSD technologies. It appears that financial incentives provided for WSD activities do not align well with the local realities in certain instances. The budgetary allocations specified in the guidelines appear too restrictive. For example, WSD works have been allocated 50% of the total budget, whilst livelihood activities and microenterprises are allocated 10 and 13% of the total budget, respectively (Government of India 2008). These allocations do not take into account the diversity of the livelihood strategies of the rural poor. In certain areas, it would be more beneficial to allocate more money for self-help groups on income-generating enterprises than WSD structures which may not provide economic benefits for a particular area.

Another aspect of incentive incompatibility relates to the private-public benefit dichotomy. Certain WSD activities may produce greater net private benefits than net public benefits. Unveiling benefits from a check dam to an individual farmer is another example of incentive incompatibility. In low-rainfall areas, rainwater harvesting using check dams is widely adopted, but benefits are accrued to a few farmers who use groundwater in the closer proximity to the check dam. There is no incentive for individual farmers to reveal their true benefit due to the intervention. The current guidelines specify a minimum of 10–90 private-public cost split (10% of the cost to be borne by the landholder and 90% by the project) for natural resource management activities on private lands. These cost-sharing rules for WSD structures need to be explicit as different activities yield different private and public benefits.

Over-subsidisation of certain WSD activities such as soil and water conservation would lead to inefficient outcomes. If households are over-subsidised, the relative costs and benefits of conservation play no role in their long-term commitment to the proposed management practice. Available evidence also indicates that NGO-treated areas have a higher level of success in terms of getting households to invest in soil and water conservation compared to government intervention (Bouma et al. 2007). High initial investments and operational and maintenance costs together with the requirement for high technical input make certain WSD technologies unattractive for farmers at least in the long run.

7.4.3 Mismatch of Scales and Poor Linkages

A project implementing agency (PIA) usually manages an area about 5,000–6,000 ha, and plans were underway to extend this up to 10,000 ha which may have exacerbated the coordination problems further.[8] This also ignores the high heterogeneity within a project area in terms of hydrology and socio-economic opportunities and barriers. In other words, these arrangements would have increased the likelihood of project failure due to information processing problems discussed earlier.

The latest WSD guidelines specify that a project area is confined to 500 ha. Whilst this obviously has some appeal in terms of matching local context with project aims and design, it makes the control of groundwater externalities difficult. The best technology to suit local conditions may not be the one suited for all areas in the project. Although the project documents stress the need to adapt proven technologies to suit local conditions, techniques not preapproved under the project design were not supported, demonstrating little flexibility. The same logic applies when selecting income-generating activities under the project. It has been pointed out that the number of activities that a typical project undertakes is too large and difficult to manage and reducing the number of activities in favour of those that provide most benefits would reduce per hectare cost of land treatment. It is also noted that long-term environmental benefits of these interventions are rarely computed.

[8] A larger scale has merit in managing certain WSD assets such as groundwater resources.

It is imperative that watershed planners understand the status quo of watershed management institutions. This involves clarifying boundaries of the resource system in question. A watershed resource system encompasses a wide array of resource subsystem but can be broadly divided into (a) forest, (b) agricultural, (c) pastures, (d) surface water and (e) groundwater. Then an understanding of property rights and access rights structures surrounding these watershed resource systems is warranted. The property rights structure and the governing rules greatly influence how the resource is managed. Most watershed resources belong to common property. Group size and homogeneity, exclusion right and power structure within the community are factors which condition the collective action (Baland and Platteau 1996; Ostrom 1990).

7.5 Concluding Remarks

WSD programmes play a pivotal role in reducing poverty and arresting resource degradation in semiarid regions of India. The progress of WSD programmes under various sponsorships has been well documented with clear evidence of under-achievement and poor adoption of WSD technologies.

This chapter analysed the performance of decentralised watershed governance structures in India. The analysis was limited to institutional design (scale of decentralisation), information and incentive incompatibility issues. These problems have received somewhat less attention in the relevant literature. It did not discuss various inherent political economy issues relevant to WSD governance. The use of MDT as the theoretical frame for the analysis was useful as it places a greater emphasis on information sharing and information processing[9] in decentralised institutional configurations.

The interaction between the WSD programmes and recent institutional changes poses several challenges. WSD projects operate within a complex hierarchical decision-making structure leading to high transaction costs. In this context, information gathering and processing costs are significant. Lower level entities may not have the access or capacity to collect and interpret technical and socio-economic data pertaining to WSD activities. Certain PIA modalities such as PRIs are likely to have lower communication costs than others. However, with limited capacity, they are given the responsibility for numerous functions at the local level.

It is recommended that the nature of the contract between a PIA and the state must be thoroughly examined in the future iterations of the Common Guidelines for watershed development. More attention must be paid to renegotiating opportunities and monitoring and evaluating modalities and compliance mechanisms between various contracting parties. It appears that certain aspects of the new institutional

[9] Other aspects of MDT include contract complexity, collusion among agents and incomplete commitment and renegotiation (Mookherjee 2008).

guidelines may suffer from incentive incompatibility problems. Reforms to current institutional settings should address incentive compatibility problems of contracting espoused by the Common Guidelines. Further, the PIA and village selection process for WSD projects lacks transparency. The PIA selection process, in particular, may be prone to adverse selection.

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Chapter 8
Sharing the Costs and Benefits of Marine Protected Areas: Implications for Good Coastal Resource Governance

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8.1 Introduction

Marine protected area (MPA) is "any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher 1999). The establishment of MPA is a major strategy towards achieving global sustainable development goals. Management strategies for MPAs can range from full protection to allowing multiple use activities (IUCN-WCPA 2008). The importance of MPAs is recognized at the global scale, with the United Nations Council, the International Union for Conservation of Nature and the various world congresses and international agreements (e.g. World Parks Congress, World Summit on Sustainable Development in 2002, Convention on Biological Diversity, Evian Agreement) calling for the systematic establishment of MPAs. The worldwide target based on the recommendations of the Durban Action Plan developed in 2003 is to establish MPAs for 20–30% of the world's oceans by 2012. The Millennium Development Goal 7 on environmental sustainability includes protection of marine, as well as terrestrial areas, among the indicators.

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Many countries have independently set up their own targets on MPA establishment and prepared their own marine conservation plans in relation to the global goals. However, as of February 2009, only about 5,000 MPAs encompassing 0.8% of the world's ocean surface have been established worldwide (Spalding et al. 2008). The progress towards global marine protection goals proceeds very slowly (Wood et al. 2008). For instance, it is estimated that it would take 100 years to place 10% of coral reefs in the Philippines under protection (Aliño et al. 2006).

One strategic mechanism to expand and sustain management of MPAs is through networking of MPAs. An MPA network is defined as "a collection of individual MPAs or reserves operating cooperatively and synergistically, at various spatial scales, and with a range of protection levels that are designed to meet objectives that a single reserve cannot achieve" (IUCN-WCPA 2008). Guerreiro et al. (2010) consider transboundary networks of marine protected areas and transboundary marine protected areas as important forms of international cooperation that will help meet international targets in biodiversity conservation and ecosystem management. Networks of MPAs can also contribute to sustainable development goals by promoting integrated ocean and coastal management through ecological, social and economic benefits and functions (IUCN-WCPA 2008).

The establishment of MPA networks for systematic protection of the marine environment has a sound technical and scientific basis. This is because unlike terrestrial areas, different areas in the marine environment are closely connected through thermal and migratory patterns, water circulation and climatic factors, as well as spawning and recruitment patterns (UNEP-WCMC 2008). Thus, while individual MPAs may not be adequate to protect the ecosystem and its processes, natural networks of MPAs may be able to do this at larger scales (Gaines et al. 2010; IUCN-WCPA 2008).

Apart from ecological or natural networks of MPAs which are based on biophysical connectivity among sites, networking can also be in the form of social networks, where people and institutions managing individual MPAs link up and connect with each other through exchange of information, sharing of experiences and good practices, as well as resources. MPA networking can be a good strategy to achieve economies of scale, thus reducing the cost of managing MPAs (Balmford et al. 2004). However, the best MPA network arrangement may necessarily be an ecosystem-based approach where natural network of MPAs is complemented by social network so that natural connectivity and social arrangements are integrated into one holistic and synergistic programme of action for the MPAs.

Managing MPAs, whether as single units or as a network, has cost and benefit implications. The creation of an MPA may be regarded as an investment in natural capital. As capital goods, they generate valuable marketed and non-marketed services, the production of which entails both private and social costs and benefits. To illustrate, environmental improvement due to MPA establishment has both a benefit (which is the damage cost avoided) and a cost (which is the opportunity cost or benefit foregone) representing something that has to be given up, such as income of fishers when a certain fishing ground is closed. Recent studies that have examined

MPA costs include Armsworth et al. (2011), Ban et al. (2011), McCrea-Strub et al. (2011), Angulo-Valdés and Hatcher (2010), Balmford et al. (2004) and Smith et al. (2010). A study by Rudd (2007) looked at MPA benefits. Lutchman et al. (2005) studied both costs and benefits of MPA, and Sanchirico et al. (2002), Alban et al. (2006) and Butardo-Toribio et al. (2009) looked further by also exploring the distributional problems involved.

The question of who is footing and sharing the costs and benefits onsite needs to be carefully examined, along with how the practices of good governance relate to cost-effectiveness and cost-efficiency in the management of single MPAs (Butardo-Toribio et al. 2009), as well as network of MPAs. Assessing the distributional issues involved in the sharing of costs and benefits (e.g. who are the gainers and who are the losers) can yield valuable information on the need for subsidies and incentives and could serve as basis for building diverse and sustainable financing portfolio for the MPA and network of MPAs. Moreover, although there is general acceptance of the MPA approach in the Philippines as indicated by the rising number of MPAs nationwide (Arceo et al. 2008), policymakers need guidance on the technical, financial, economic and governance dimensions of MPA establishment in order to fully support and accelerate the implementation of this resource management strategy nationwide.

The cost-benefit analysis was used in this study to achieve two objectives. Firstly, the study aimed to examine the costs and benefits of establishing and implementing an MPA and how these are being shared on-site under various types of management arrangements, including under the network approach. Secondly, the study tried to explore how governance practices with emphasis on transparency, accountability and public participation relate to effective and efficient management of MPAs under various types of institutional arrangements. This is an important objective since the governance aspect of MPAs remains largely unstudied (see McCay and Jones 2011; Jones et al. 2011 for some of the current discussions).

8.2 Methods

8.2.1 Study Area

The study covers six MPAs situated in five municipalities in the provinces of Cebu and Zamboanga del Sur, Philippines (Fig. 8.1). These MPAs were established for the twin goals of enhancing local fishery resources and conserving marine biodiversity. These sites were chosen because they represent different types of MPA management arrangements, size and age – which are the factors deemed in this study to influence the streams of MPA costs and benefits and quality of governance practices. A brief profile of the study sites is shown in Table 8.1. For the network level study, the experiences of the network of MPAs in Illana Bay in the province of Zamboanga

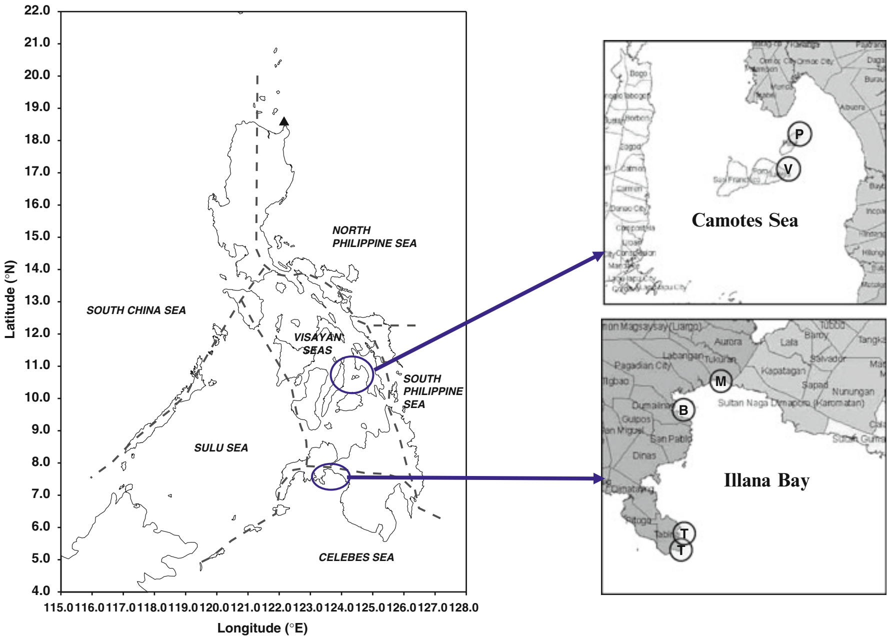

Fig. 8.1 General location of the six study sites

del Sur and Camotes Sea in Cebu province were examined. Each of the six case study sites belongs to either of these MPA networks.

The study municipalities of Pilar and Tudela on Camotes Islands in the province of Cebu are part of the Camotes Sea Coastal Resource Management Council. This council has a Protected Area Committee that oversees the management of all MPAs within the member municipalities. At least nine MPAs, including the two MPA case study sites, form the Camotes MPA Network.

The case study municipalities of Dumalinao, Tabina and Tukuran in Zamboanga del Sur province are active members of the Illana Bay Regional Alliance 9. This regional alliance of nine component local government units including the provincial government serves as a coordinative body as well as financing and technical arm for the coastal resource management programmes of the province. This regional alliance is composed of two subclusters, with four local government units each as members. Cluster 1 called PaTuLaD covers 7 MPAs while cluster 2 called SanTaDiDi covers 11 MPAs. This regional alliance adopts a bay-wide management strategy to protect coastal and marine resources shared by member local government units. Inter-local government unit collaboration in coastal and marine resources management has legal basis in the Philippine Fisheries Code of 1998 and the Local Government Code of 1991.

Table 8.1 Brief profile of the MPAs used in the study

MPA	Location	Total size (ha)	Year established	Management arrangement[a]
(PMMP)	*Barangay* Lower Poblacion and *Barangay* Villahermosa, Pilar (Camotes Island, Cebu)	179.4	2005	Co-management between municipality and *barangay*
Villahermosa Marine Sanctuary	*Barangay* Villahermosa, Tudela (Camotes Island, Cebu)	69.3	2004; although already existing in 2002	*Barangay*-managed with some funding assistance from municipality
Bibilik MPA	*Barangay* Bibilik, Dumalinao, Zamboanga del Sur	20.0	2003; although already existing in 2002	Co-management between municipality and *barangay*
Tambunan MPA	*Barangay* Malim, Tabina, Zamboanga del Sur	103.0	2003	Municipality-managed
Talisay MPA	Sitio Pangalaran, *Barangay* Malim, Tabina, Zamboanga del Sur	32.8	2004; although protected by the PO since the 1980s	People's organization (PO)-managed
Militar, Sto. Niño, Sugod and Tagulo (MiSSTA) MPA	*Barangay* Tagulo and *Barangay* Sugod, Tukuran, Zamboanga del Sur	160.0	2003	Co-management between municipality and *barangay*

[a]*Barangay* (village) is the smallest unit of government in the Philippines. It is within a municipality or city

8.2.2 Assessment of Costs and Benefits

Primary data were collected from semi-structured interviews and focus group discussions. Interview results were validated with secondary data gathered from biophysical assessment reports, participatory resource assessments, fisheries profiles and MPA plans, and available MPA and municipal government records. Data on MPA costs and benefits (revenues) were obtained from actual values provided by the municipal, *barangay*, and donor agency key informants. The costs of volunteer labour, municipal staff labour and *barangay* labour were derived from estimates provided by key informants about the quantity of such labour and their opportunity costs.

Costs and benefits of the MPAs are categorized as direct and indirect and divided into two phases: establishment and operational (or implementation). The total cost was derived by summing up these two costs. The establishment costs include capital costs (e.g. boat, guardhouse/outpost), costs associated with site delineation, installation of marker buoys, as well as organization and management planning activities. The expenditures considered as implementation or operational costs are annual administrative costs (personnel, office supplies and materials, staff travel, etc.) and operations/activity costs (law enforcement, IEC, training, site rehabilitation, etc.).

Positive and negative effects arising from MPA establishment and operation were also assessed from anecdotes provided by key informants. Information on how costs and benefits are being shared on-site was obtained through key informant interviews, as well as from existing community and municipal records.

8.2.3 MPA Management Effectiveness Rating

To measure the effectiveness of MPA management, the MPA performance rating developed by Cebu Coastal Environment Foundation (White et al. 2004), as modified by the United States Agency for International Development/Philippine Environmental Governance 2 Project, was utilized. This rating system considers five MPA management effectiveness levels as follows: level 1 (the MPA is initiated), level 2 (the MPA is established – fair), level 3 (the MPA is enforced – good), level 4 (operations of the MPA are sustained – very good) and level 5 (the MPA is institutionalized – excellent rating). A minimum set of criteria/activities have to be satisfied for MPAs to be considered as having achieved a particular management level (see Annex A). The modified rating system has also been integrated with indicators that measure the adoption of good governance principles of functionality, transparency, accountability and public participation.

8.3 Results and Discussion

8.3.1 MPA Costs

The estimated total direct costs incurred during the establishment and implementation stages of the six MPAs studied are presented in Table 8.2. Total cost is generally higher during the implementation stage than the establishment stage. The activities that generally required higher expenditures during the establishment stage include the construction of guardhouse, multipurpose building and boardwalks and the installation of marker buoys. In the implementation stage, the more costly items were law enforcement and habitat rehabilitation activities (i.e. mangrove replanting, coral transplantation, seagrass rehabilitation). Costs also varied depending on the nature of labour employed (paid or unpaid, lower or higher opportunity cost), type of equipment used (e.g. motor patrol boat vs. paddle boat) and building materials (e.g. permanent vs. temporary guardhouse). Total cost was highest for Tambunan MPA, and lowest for Pilar Municipal Marine Park, which is the largest among the MPAs studied.

In general, the MPAs that are being managed by the municipality (whether directly or under a co-management arrangement with the *barangay*) incurred higher average annual costs (derived by dividing the total cost with the age of the MPA) than those being managed by the *barangay* or people's organization (PO) (Table 8.2). This observation might be attributed to the municipalities' greater funds access which allowed them to spend more on MPA activities. When MPA size is taken into account, the larger MPAs incurred lower annual cost per hectare than the smaller MPAs. For example, the average annual cost per hectare of the four largest MPAs was only Php 4,500 (around US$104) compared to the average annual cost of the two smallest MPAs (Bibilik and Talisay), which is around Php 16,000 (around US$368). Moreover, the two smallest MPAs incurred the highest cost per hectare during both the establishment and operational stages, which suggests that it may be more costly to establish and implement smaller MPAs than larger MPAs, due to the economies of scale.

Indirect costs, as used in this study, refer to the unfavourable effects that occurred within the community as a result of MPA establishment. Data obtained from anecdotal reports provided information on the indirect costs (Table 8.3). The costs mainly included additional time and fuel to relocate to areas where fishing or other resource extraction activities are allowed. To illustrate, illegal fishers shifted to legal fishing practices to avoid fines and penalties when enforcement started, but, as a consequence, they suffered a decline in their income due to increased fishing cost.

Table 8.2 Total estimated cost incurred in the establishment and implementation of the six study MPAs (undiscounted, in Philippine pesos, from establishment to 2006 only)

MPA	Establishment[a]	Implementation[b]	Total cost	Average annual cost	Average annual cost/ha
PMMP	445,082 (2,481/ha)	612,153 (3,390/ha)	1,057,235	528,617	2,947
Villahermosa	377,867 (5,453/ha)	808,898 (11,672/ha)	1,186,765	237,353	3,425
Bibilik	799,159 (39,958/ha)	1,427,326 (71,366/ha)	2,226,485	445,297	22,265
Tambunan	840,778 (8,163/ha)	1,999,942 (19,417/ha)	2,840,720	710,180	6,895
Talisay	357,576 (10,902/ha)	970,452 (29,587/ha)	1,328,028	332,007	10,122
MiSTTA	741,081 (4,632/ha)	1,574,016 (9,838/ha)	2,315,097	771,699	4,823

Source: Butardo-Toribio et al. (2009)

[a]Includes capital costs (e.g. boat, guardhouse/outpost), installation of marker buoys, ordinance formulation, organization and planning activities; value in parenthesis is cost per hectare

[b]Includes administrative costs (personnel, office supplies and materials, travel, etc.) and activity costs (law enforcement, IEC, training, rehabilitation, ecotourism facilities, etc.); value in parenthesis is cost per hectare

Table 8.3 Negative effects caused by MPA establishment in the various study sites

MPA	Who were affected	How were they affected	Remarks
PMMP	Owners of navigational boats	Dry-docking within the vicinity of the MPA was prohibited	Permanent, boat operators now dry dock in a neighbouring province
	Illegal fishers and fishers who operated inside the MPA	2–3 fishers using chlorine lost their earnings of P190 each	Offset by good harvest using legal gears and in areas outside of the MPA
		Spear/cyanide fishing (2–3 boats with net income of P5,000/boat, net share per crew of P400–P500)	Illegal fishers shifted activities to other areas
		Fishing activities, even legal, have to be relocated	
	Sand quarrying operators	Loss of net income of P650/day each for two families	Temporary, operation was relocated to another *barangay*
	Mangrove cutters	40–55 persons who engaged in mangrove cutting to sell wood and fuel lost income	Mangrove wood and fuel now bought in the neighbouring municipality of Tudela
Villahermosa	Shellfish collectors	Prohibition on shell collection inside core zone affected subsistence and commercial collectors	Minimal effect since collection of shellfish for food still allowed in the buffer zone
	Fishers who formerly operated in present no-take/core zone	Greater distance and, hence, time and cost to go to the areas where fishing is allowed	Net fishers affected, hook and line fishing still allowed inside the buffer zone; however, since fishers shifted activities to outside the boundaries of MPA, fishing pressure have shifted there
Bibilik	Fish pen/cage operators	Area of operation had to be relocated in another area	Affected individuals still against the MPA
	Subsistence fishers who formerly operated inside the MPA	Five families (additional 1-km walk or 20-min travel by nonmotorized banca or 2–3 min if motorized (fuel cost is P5))	Affected fishers had to walk farther to move to another location, fishers shifted operation just immediately outside the MPA
		Around 50 families affected by prohibition on shellfish collection	

(continued)

Table 8.3 (continued)

MPA	Who were affected	How were they affected	Remarks
Tambunan	Boat owners	Prohibition on passing and anchoring except along designated passage way meant additional travel time and fuel (20 motors and 10 paddle boats)	Temporary, prohibition not strictly enforced
	Fishers and shellfish collectors	Activities had to be done outside the MPA	Offset by improved catch at present
Talisay	Boat owners	20 boat owners resisted the MPA because of the effect on navigation	Remaining opposition to the MPA due to partisan politics, not linked to MPA operation
	Fishers	More than 50 fishers engaged in hook and line fishing and ten families involved in octopus fishing; area of operation had to be relocated in another area	There is no dislocation of livelihood, and the impact is in terms of extra paddling time and fuel for motorized due to the need to relocate fishing activities
MiSTTA	Boat owners and majority of fishers	Additional expense for motorized boats and 5- to 20-min difference in paddle boat time	Offset by improved catch at present, resistance addressed through frequent meetings

8.3.2 Sharing of MPA Costs

Partnership in the management of MPAs among stakeholders has been shown to be a good strategy for defraying the costs of local resource conservation. Partnership also helped in promoting the value of shared environment stewardship. In general, the direct costs (establishment and operational) of the six MPAs studied are shared among five sources: (1) local government units (*barangay*, municipality, province), (2) local revenue streams (net revenues from livelihood, user fees, fundraising, etc., which are generated and ploughed back to MPA management), (3) national government agencies (e.g. Department of Environment and Natural Resources, Department of Agriculture-Bureau of Fisheries and Aquatic Resources, Philippine National Police-Coast Guard), (4) donors and assisting organizations (aid organizations, private sector donor, non-government organizations, etc.), and (5) the host local community, which is the main source of volunteer labour (Butardo-Toribio et al. 2009). The distribution of costs among the five sources was dependent on the type of management arrangement. In the municipality-managed or comanaged MPAs, the municipal government provided the largest contribution of the total cost while the *barangay* and community had the highest contribution in the *barangay*-managed Villahermosa MPA. However, the PO-managed Talisay MPA had higher contribution from outside grants or donations than from the community. In fact, external grants/assistance contributed significantly in defraying the total costs of all the MPAs, especially a large part of the establishment cost for both PO-managed (80%) and *barangay*-managed (44%) MPAs. Donor assistance provided about 37% of the total cost of comanaged MPAs, 28% of the *barangay*-managed MPA and 59% of the PO-managed MPA (Butardo-Toribio et al. 2009). Meanwhile, the government agencies contributed very little, if any at all, to total MPA direct costs.

The type of management arrangement also influenced the sharing of labour cost, which accounts for as much as 50% of the total MPA cost (Butardo-Toribio et al. 2009). The municipal government provided the largest percentage of labour cost in municipality-managed or comanaged MPAs by mobilizing more municipal staff to oversee these MPAs. In contrast, very limited municipal staff time was provided by the host municipalities in the *barangay*-managed and PO-managed MPAs. For the latter MPAs, *barangay* officials, community volunteers and members of the PO or fishermen's association provided at least 80–90% of the labour needs of these MPAs.

Indirect costs of MPA establishment and implementation are primarily borne by fishers, whether using legal or illegal practices, who were displaced from their customary fishing areas. This is observed in all of the six MPAs studied (Table 8.3). The other groups that incurred indirect costs from the MPA include boat owners (when navigation is regulated or prohibited within and in the vicinity of the MPA) and other resource users, such as shellfish collectors, mangrove cutters and sand quarrying operators, whose activities have to be relocated elsewhere.

8.3.3 MPA Benefits

Based on anecdotal reports, the host local communities have observed socio-economic and ecological improvements since the establishment of their MPAs. Some of the perceived benefits include improved fisheries, higher coral cover and marine biodiversity, improved environmental awareness, enhanced community solidarity and community empowerment. The benefits of the MPAs did not only trickle down to fishers and resource users who benefited from improved resource conditions but also generally shared by the whole village through increased environmental awareness and solidarity.

The community's perceptions of improved fish catch and coral recovery are consistent with the results of annual biophysical monitoring conducted in all six MPAs. Increasing trends in fish abundance and biomass and improved hard coral cover have been observed inside all MPAs (Butardo-Toribio et al. 2009), although these improvements in reef conditions have not been translated to monetary values. However, the indicative annual economic value of mangrove resources in three of the six study areas has been estimated using the benefit transfer method to be US$113,712 (Php 5.2 M) for Pilar, US$10,712 (Php 494,359) for Talisay and US$7,004 (Php 323,235) for Tambunan (Butardo-Toribio et al. 2009). Generally, the estimated annual economic benefits far exceed the total annual direct costs to manage these MPAs.

8.3.4 Management Performance and Cost-Efficiency

Based on the results of the MPA rating, the management level (i.e. management performance) of the study MPAs ranged from level 2 (MPA is established) to level 3 (MPA is enforced) (Table 8.4). All MPAs with a level 3 rating and Pilar actually qualified for level 4 (MPA is sustained) based on the original criteria (White et al. 2004), but deficiencies in governance practices gave them a lower rating under the modified MPA rating system used in the study. Overall, positive ecological effects as seen by improved reef conditions are observed in all the six MPAs regardless of management level. Management performance does not seem to be influenced by the type of management system, implying that institutional arrangement may not be a critical factor in the success of the MPA. However, the MPAs in this study are still relatively young; thus, the importance of institutional arrangement which may be critical in their sustainability has not been observed yet. For instance, past experiences on community-based coastal resource management initiatives in the Philippines have shown that local communities must work in partnership with the government to ensure that their efforts will be sustained (Rivera and Newkirk 1997), and contributions from the municipal government is a critical factor that can influence the success of community-based MPAs (Pollnac et al. 2001).

Table 8.4 Status of MPA management and effectiveness in the study MPAs relative to their costs

MPA (*management arrangement*)	Management rating (*as of 2007*)	Indicative ecological effects (*based on changes in coral reef conditions*)	Average annual cost (pesos)	Annual cost/ha (pesos)
PMMP (comanaged by *barangay* and municipality)	Level 2 (MPA is established)	Improved hard coral cover, fish density and biomass inside the MPA; hard coral cover and fish density improved in the vicinity of the MPA	528,617	2,947
Villahermosa (*barangay*-managed assisted by fishermen's association)	Level 3 (MPA is enforced)	Improved hard coral cover, fish density and biomass inside and in the vicinity of the MPA	237,353	3,425
Bibilik (comanaged by *barangay* and municipality)	Level 2 (MPA is established)	Improved fish density and biomass inside the MPA; no baseline data on coral cover	445,297	22,265
Tambunan (municipality-managed)	Level 3 (MPA is enforced)	Improved hard coral cover, fish density and fish biomass inside and outside the MPA	710,180	6,895
Talisay (PO-managed)	Level 2 (MPA is established)	Improved hard coral cover and fish biomass inside MPA	332,007	10,122
MiSTTA (comanaged by *barangay* and municipality)	Level 3 (MPA is enforced)	Improved fish density and biomass inside the MPA; no baseline data on coral cover	771,699	4,823

Source: Modified from Butardo-Toribio et al. (2009)

With the exception of Pilar, MPAs with lower management levels (Bibilik and Talisay) incurred the highest annual cost per hectare despite their small sizes and presumably simpler management needs. In contrast, the more effectively managed MPAs have lower annual cost per hectare. These results suggest that cost-efficiency is a good indicator of management performance. However, in the absence of more studies, this observation can perhaps also be attributed to the effect of the economies of scale since these two MPAs are also the smallest among the case study sites.

8.3.5 Cost and Benefits of MPA Networks

MPA networking can reduce costs of managing MPAs due to economies of scale (Balmford et al. 2004). In addition to the cost of managing the individual MPAs in their localities, the local government units that joined the MPA network incurred additional cost in terms of regular funding contribution to the network operations. However, while this meant higher budgetary requirement on the part of the LGUs, the increase in the size of coastal and marine area effectively managed and guarded has been shown to translate to higher cost-effectiveness. For instance, an initial study on the Camotes Sea Council shows that the cost of coastal law enforcement per square kilometre of municipal waters with a municipality enforcing the law individually is much higher (average of US\$72) and the effective enforcement coverage (5 km^2) is much lower than when the municipalities collaborate together (effective enforcement of 10 km^2 and average cost to each local government of US\$39) (Arceo et al. 2008).

Holling (1973), as cited by IUCN-WCPA (2008), believes that the ecological interconnectedness between and within ecosystems through MPAs that are strategically located can strengthen the resilience of the systems against stresses. In addition, networking enables the creation of a biologically and administratively coordinated system of MPAs that can provide a consistent approach to design, finance, management and monitoring (Ballantine 1994; White et al. 2006). In the case study sites, for instance, network-level activities have resulted in the creation of functional composite teams for each thematic programme (e.g. enforcement, education/advocacy, monitoring and evaluation, financing, etc.) in the network action plan. The activities also provided opportunities for sharing knowledge based on actual experiences in MPA management, such as during regular network-wide MPA forum. This supports the observation of White et al. (2006) that networking of MPAs leads to a social network where individual MPA stakeholders coordinate with each other and share experiences resulting in complementation and synergy of efforts.

For both the MPA networks (Camotes MPA Network and IBRA9 Alliance) studied, joint activities include information/education campaigns and advocacy, law enforcement and participatory monitoring and evaluation. The members of the MPA networks have formalized their collaboration through the signing of a memorandum of agreement. Based on their experiences, networking can result in the following benefits (EcoGov 2011):

1. Facilitate resolution of conflicts between or among municipalities (e.g. delineation of contested municipal water boundaries, harmonization of inconsistent fishery ordinances).

2. Develop complementary and synergistic approaches and coordinating mechanisms, thereby reducing transaction costs. For instance, in the MPA network studied, common policies, standard operating procedures, monitoring and evaluation and feedback and response systems have been developed.
3. Facilitate sharing of technical expertise and establishment of decision support systems linked to incentives and MPA performance and impact evaluation.
4. Promote sustainable financing schemes based on synergized funds leveraging.

Various interacting factors that can affect the type and magnitude of costs and benefits of MPA network establishment (and therefore the net benefit) may include those described below:

Purpose of the MPA Network: MPA networks may be created for any or combination of the following objectives: (1) biodiversity protection, (2) sustainable fisheries management, and (3) development of non-extractive uses of the ecosystem like ecotourism and other recreational activities (Alban et al. 2006). The purpose of the establishment of the network could affect the costs and benefits because of its implications on institutional arrangement/management regime, rules and prescriptions, property rights, operations and ecological and socio-economic impacts. Moreover, assigning competing or non-complementary objectives to the MPA network could cause conflicts among various stakeholders and policy actors, which could raise the transaction cost and undermine the sustainability of the network.

Type, Organization and Activities of the Network: A network that is established based on factors of natural ecological connectivity may theoretically be expected to result in greater economic impact due to improved ecological synergy, resilience, biodiversity and sustainability. Overall benefits from a network primarily established based only on social, institutional or political considerations might not be as maximal. However, the formation of a network based on social connectivity could presumably result in lower transaction and other management-related costs due to a higher capacity of MPA managers to work together due to shared values and interests. The degree of complexity and effectiveness of the institutional or social arrangement, including the number and type and complexity of activities the network is involved in, may also be expected to affect the magnitude of network management cost and, therefore, the net benefits from this type of management arrangement.

Design, Size and Geographic Distance: The size and design of individual MPAs are important inasmuch as ecological connectivity, migratory patterns, potential for spillover, life history stages of species and needs of fishing communities have to be considered for optimum benefits. At present, only a limited number of MPAs meet their management goals because too many are set up in the wrong places or with unrealistic expectations (Jameson et al. 2002). In addition, the number of MPAs involved in a network, as well as the extent of geographic separation among them, can affect the magnitude of costs and benefits due to the combined effects of transaction cost and economies of scale. Gaines et al. (2010) therefore recommend size, spacing, location and configuration guidelines in designing marine reserve networks to promote both conservation and fisheries management. In addition, IUCN-WCPA (2008) provides five ecological guidelines for the design of resilient MPA networks.

Degree of Maturity or Level of Management Effectiveness of the MPA Network: It is expected that as the maturity level (e.g. due to history of external assistance and age of the MPA network) of the MPA network increases, the gains in social capital would be greater and the operation of the network would be more efficient due to more developed organizational capacity. The streams of costs and benefits would also be logically different at various stages of the MPA network. For instance, transaction cost is expectedly higher at the establishment stage because of the higher expense on searching, negotiation and contracting. Transaction cost could be lower at the enforcement stage when rules and operating procedures are already established and stable, including in terms of cost, personnel, material resources and technical expertise-sharing arrangements. Also, from the social and economic perspectives, costs could also be higher at the establishment stage as fishers temporarily suffer a setback in their welfare due to the loss of fishing grounds. Such costs can be expected to be lesser at later stages when the livelihood options have already been secured due to improved fish biomass and ecological conditions.

8.4 Summary and Conclusion

The study showed that establishing and managing an MPA can have significant financial and human resource cost consideration, particularly during the initial stages. Costs varied with local contexts, and the operation of the economies of scale is apparent. Cost sharing and partnerships among local stakeholders helped in defraying MPA costs, with the local communities contributing significantly to MPA management through their volunteer labour. National government agencies provided limited support, but there are many opportunities for them to get more engaged such as in providing needed technical assistance. Meanwhile, MPAs managed by people's organizations and *barangay*s would benefit from increased local government support, which the study showed to be minimal overall and tend to focus only on MPAs that the local government themselves directly manage or comanage. Though external funding support is critical particularly during the initial stages, there is a need to pursue building self-generated funds to enhance MPA management and sustainability. It is also important to design a system for equitable sharing of costs and benefits and to provide early incentives to marginalized MPA managers and cooperators.

The cost of establishing and managing an MPA appears to represent only a minor fraction of the potential benefits that can be derived from it. However, MPA establishment is not a sufficient ingredient in managing coastal areas. The value of MPAs lies in their limiting fishing effort and in providing spillover for the fishing population.

MPAs must be designed so as to maximize this goal and the net benefits that can be derived. Furthermore, networking of small, isolated MPAs based on ecological connectivity and integration of MPA efforts through the creation of social networks can lead to greater effectiveness by increasing spatial scales pertinent to fisheries management and biodiversity conservation as well as to higher cost-efficiency in the long run.

Monitoring of coral reef benthos and reef fish conditions provides some inferential support to community anecdotes about improving fish catch and coral conditions. Further studies are needed to establish the validity of this observation. In addition, local key informants reported various positive socio-economic benefits from MPA establishment. However, the establishment of the MPAs caused some unintended effects on the livelihood strategies of some community members which can undermine the goals of the MPAs. The study thus underscores the need for an integrated approach that links MPA management to overall coastal resource management and social and economic development strategy of the local government.

Effective and efficient MPA management does not necessarily depend on the kind of management arrangement involved. Political will, the ability to muster needed local and external support, and good governance practices are important. In particular, good governance will help MPAs in projecting their stability, legitimacy and credibility to the general public and to the donors, as well as offer long-term protection from threats and food security (Juinio-Meñez et al. 2007).

While the establishment of an MPA is by itself a worthy exercise, networking of MPAs would theoretically enhance ecological benefits through synergy, improved resilience, biodiversity and sustainability. MPA networking may hasten the achievement of MPA goals and targets (i.e. time is money; the longer the MPA is able to realize the various benefits, the more inefficient it is).

A critical feature of MPA networks concerns the people and the institutional arrangement they forge in order to operationalise and sustain the partnership as well as address benefit-cost sharing and other socio-economic issues associated with setting aside protected areas. The success or failure of the network can depend on how the parties are able to work effectively together and sustain the partnership. An analysis of the social and institutional context of the MPA network, complemented by an economic assessment, can provide a more holistic understanding of its dynamics.

In summary, MPA networks could enhance and scale up benefits of natural processes and municipal-level management interventions in terms of biodiversity conservation, fishery productivity, livelihood of fisherfolks, recreational and aesthetic value of the environment and effectiveness of local environmental governance. To realise optimum benefits, MPA networks should be within the context of an integrated fisheries management at the municipal and intergovernmental levels (e.g. Cicin-Sain and Belfiore 2005). This can be achieved through effective enforcement of fishery laws and common regulation of fishing activities to ensure social equity. Substantial benefits will eventually accrue to municipal fishers and local community as a result of this ecosystem and resource management strategy.

A limitation of this study is that the costs and benefits of the MPAs and network of MPAs have not been fully quantified and compared. While theoretical, anecdotal and short-term field observations support the soundness of this management approach, detailed and longer-term studies will be needed. Gaining full understanding of the dynamics involved will inform decisions on design of MPAs and MPA networks to make them more cost-effective. This study, however, is able to suggest a framework for analysing the implications of costs and benefits and how these are shared on site as well as the importance of the governance dimension in strengthening the value of MPAs and network of MPAs.

8.5 Annex A EcoGov Modifications of CCEF-Developed MPA Rating System

Level of MPA management	Criteria/activity satisfied
Level 1 – MPA is initiated	MPA concept accepted
	Management body membership tentatively determined
	Preliminary management plan drafted
	Resolution and/or ordinance drafted
	Site surveyed using standard methods with baseline assessment complete, preferably conducted in a participatory process
	Education programme raising awareness about MPA functions and benefits started
Level 2 – MPA is established	Community acceptance gained and documented
	Ordinance passed and approved by the Municipal Council
	Management plan adopted and legitimized by the LGU or PAMB
	Boundaries delineated
	Signboards/billboards posted
	MPA outpost or other structure constructed
	Management activities started (e.g. patrolling and surveillance)
	Biophysical monitoring includes local participation
	IEC activities conducted
	Budget for year 1 implementation allocated
Level 3 – MPA is enforced	Management body active and supported by legal instrument
	MPA billboards, boundary markers/anchor buoys maintained
	Collaborative patrolling and surveillance conducted by mandated-enforcement group and local community volunteers
	Regional participatory biophysical monitoring being conducted

(continued)

(continued)

Level of MPA management	Criteria/activity satisfied
Level 4 – MPA sustained	MPA management plan and/or ordinance reviewed in a participatory process
	Management body capable to run the MPA independently
	MPA billboards, boundary markers/anchor buoys maintained
	Enforcement system fully operational
	Annual participatory biophysical monitoring and timely feedback of results being implemented for at least 2 years
	Socio-economic monitoring regularly conducted
	Management body regularly monitored and evaluated
	Budget from LGU or from other sources is being allocated and accessed for 2 or more consecutive years
	Environment-friendly enterprise and/or fees collected as a sustainable financing strategy
	Illegal and destructive activities stopped inside and within the vicinity of MPA
Level 5 – MPA institutionalized	Ordinance passed by the Provincial Council giving MPA stronger political support
	Management plan refined for adaptive management
	MPA management plan incorporated in LGU development plan
	Management body capacitated for fund sourcing
	Expansion strategies or enhancement programmes initiated
	Effective coordination with appropriate national and local agencies on CRM/MPA policies and with other LGUs achieved
	Support facilities constructed/added
	Evaluation of impacts on ecology and socio-economy conducted, completed and feedback mechanisms are in place
	Performance M&E linked to an incentive system regularly conducted
	IEC programme on MPAs maintained over the years
	Advance IEC materials developed and disseminated with assistance from partners and/or private sector grants
	MPA emphasizes on public education and is being used as a study tour site, residents advocate for MPAs
	Revenues from enterprise and/or fees sustained and accounted for

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Chapter 9
Indigenous Development Through Payments for Environmental Services in Arnhem Land, Australia: A Critical Analysis

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9.1 Introduction

Across a range of indicators—median income, life expectancy, child mortality, education and employment—wide gaps in outcomes distinguish indigenous[1] and nonindigenous Australians (Steering Committee for the Review of Government Service Provision 2009). The current debate on indigenous development emphasises participation in the mainstream economy as the critical element in overcoming this socio-economic disadvantage (Council of Australian Government 2009). Economic participation is equated to entrepreneurship and employment, and government policies and public/private cooperation aim at providing job opportunities for indigenous Australians in manufacturing, mining, agriculture, forestry, retail and other services (Australian Employment Covenant 2009; Council of Australian Government 2009). This model of economic participation may suit many indigenous Australians. Others, particularly in remote areas,[2] may face economic conditions that make employment in the mainstream economy a difficult challenge. Around 26% of the indigenous population live in remote and very remote regions of Australia (Australian Bureau of Statistics 2008), where job opportunities are limited and economic participation requires relocation or increased mobility, potentially resulting in further economic disadvantage (Biddle 2010). Job creation for indigenous people in remote communities has also had limited success in and around major mining projects (Altman 2009). Furthermore, the low agricultural potential in

[1] Throughout the chapter, I refer to the aboriginal and Torres Strait Islander population as indigenous people.

[2] For a definition of remote and very remote areas, see Australian Bureau of Statistics (2007).

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large parts of indigenous-owned land does not allow for large-scale investment in this sector (Luckert and Whitehead 2007). Indigenous Australians may also have different sets of incentives and cultural demands precluding direct transfer of non-indigenous models of entrepreneurship and employment (Austin-Broos 2003; Lindsay 2005).

Commercialisation of ecosystem services through market-based instruments is an alternative form of indigenous economic participation. Muller (2008) and Greiner et al. (2009), among others, advocate payments for environmental services (PES) as an opportunity for indigenous landowners to support their environmental and cultural management activities in northern Australia. A mix of market-based instruments—including the creation of markets for environmental services—and government funding is also at the core of proposals for establishing 'conservation economies' in indigenous-owned land (Altman and Whitehead 2003; Hill et al. 2007; Luckert et al. 2007; Woinarski et al. 2007). Market-based instruments are also at the core of the Australian government's Environmental Stewardship Program for the protection of environmental resources (Department of the Environment, Water, Heritage, and the Arts 2008). The idea of using market-based instruments and specifically PES schemes, linked to natural resource management (NRM) carried out by indigenous landowners and indigenous land and sea management groups, is gaining momentum in both academic and government circles in Australia.

Contrasted to other forms of economic participation, PES schemes and market for environmental services linked to indigenous NRM have several advantages. First, unlike other landowners (e.g. farmers), indigenous communities have ownership of cultural and environmental assets of outstanding, tangible and intangible, internationally significant values (Altman et al. 2007). They own the assets that produce the environmental/cultural goods and services that could be sold through PES or similar schemes. Second, as many of these assets are in remote and very remote regions of Australia, indigenous landowners have a locational advantage over other potential environmental service providers. Indigenous landowners are geographically well placed to address the complexities of many environmental issues in remote regions of the Australian continent (Luckert and Campbell 2007). Third, the indigenous labour force possesses or has access to indigenous ecological knowledge that is an essential skill for NRM in remote areas. Fourth, indigenous employment in NRM for remote regions matches many people's aspirations to live on and care for their country (Northern Land Council 2006; Whitehead et al. 2009a, b). Indigenous knowledge, location and ownership of important environmental assets could give indigenous landowners a competitive advantage in markets for environmental services.

It is uncertain, however, if participation in markets for environmental services is also sufficient to promote indigenous development. Research shows that a condition for sustained improvements in well-being is genuine decision-making power (Hamilton 1999; Hill 2003; Hunt 2008). Arguably, with the exception of Samuelson's ideal world of perfect competition (Samuelson 1957), the exercise of power is always present in market transactions. The exercise of power is dependent on which side of a non-clearing market an agent is positioned (Bowles and Gintis 2007). Agents whose demand (supply) is less than the supply (demand) of the other market side may exercise power by threatening to withdraw from the transaction. The position in the market is linked to the agent's ability to move his/her endowment across uses. That is, the more specific the agents' endowment, the harder it is to find alternative uses that can gain the owners some negotiating power (Hart 1989). Note also that the exercise of power involves the threat and use of sanctions. According to many political theorists, sanctions are actually the defining features of the exercise of power (Bowles and Gintis 2008; Parsons 1967). The aim of this chapter is to investigate if market exchanges of environmental services through PES or similar instruments help indigenous communities to gain authority and capacity over their own resources and futures or if they tend to perpetuate dependence and inequalities.

I approach this question firstly through a theoretical analysis. I review the theory of PES and contrast it with secondary data and evidence on the environmental conditions of the indigenous estate.[3] This analysis seeks to understand if the requirements for successful implementation of PES schemes and creation of environmental markets and the conditions of their estate are likely to place the indigenous landowners in the side of the market with the greater ability to exercise power and hence drive development according to their goals and aspirations. I then illustrate the experience of the Dhimurru Aboriginal Corporation (hereafter Dhimurru) and Djelk rangers in participating in PES and other contractual arrangements for the provision of services. Primary data was collected during stakeholders' interviews, participatory observation and action research, as described in the next section. This summary of the rangers' experience highlights where the actual power resides in negotiating contractual arrangements with government agencies and private companies for the provision of environmental and other services. In the final section, I elaborate an overall assessment of PES and pseudo-PES schemes for indigenous development in Australia.

9.2 Research Methodology

The research methodology was based on the collection and analysis of primary and secondary data. A review of the literature and informal talk with experts in indigenous NRM issues provided the secondary data used to contrast the theoretical framework of PES with the indigenous economic and cultural context. Primary data was collected through extensive fieldwork started in September 2009 with regular 4-week visits to the field sites. In the field, I used four different approaches for data collection. During the earlier stages of the fieldwork, I interviewed the senior indigenous and nonindigenous staff of the two organisations. These were semi-structured

[3] The indigenous estate is land held by—or on behalf of—indigenous people under a corporate or group title. It covers around 20% of the Australian landmass and has internationally significant environmental and cultural values (Altman et al. 2007).

interviews focused on the history of the organisation, the motivations for their conservation activities, the types of conservation activities they undertake, the financial and capital needs and the forms of government support they receive. I then analysed internal and unpublished documentation related to the organisations' revenues and cost structure and contractual arrangements with their commercial partners. I also used participant observation methodology complemented with an action research approach to gain a better understanding of the organisations' conservation activities, constraints and risks. This methodological approach helped to develop better reporting methodologies to keep track of inputs, outcomes and environmental outputs of organisations' activities.

9.3 PES and the Indigenous Context

The theoretical roots of PES can be traced to the neoclassical economic theory of externalities and public goods as systematised, for instance, by Coase (1960). The Coasian approach stipulates that negative environmental externalities can be potentially traded by the affected parties. Environmental service (ES) beneficiaries make direct, contractual and conditional payments to landowners for adopting management practices that reduce negative external effects and thus secure ecosystem conservation and/or restoration (Engel et al. 2008; World Bank 2009; Wunder 2005). According to Engel et al. (2008), the successful implementation of PES requires that (a) natural ecosystems are mismanaged because many of their benefits are externalities from the point of view of the owners, (b) buyers are identified and willing to pay for environmental services, (c) sellers of environmental services are also identified, (d) transactions are voluntary, (e) environmental services are well defined, and (f) payments are conditional on effective service provision (conditionality).

While this framework is quite straightforward, these requirements are not sufficient *per se* to distinguish PES scheme from other conservation approaches. In fact, this list of requirements has created serious ambiguities among practitioners and scholars so that nowadays the PES label applies to any sort of programme (see, e.g. Muradian et al. 2010). What sets PES apart from command-and-control approaches to environmental conservation is that PES programmes encourage *a change in output composition* by increasing the private benefits associated with the conservation of environmental assets—so that these private benefits are larger than the opportunity cost of conservation. Command-and-control strategies, taxes and subsidies, on the contrary, affect the revenue structure, the volume and/or the cost of production to encourage or restrict the firms' production. Under this distinction, a money transfer to farmers, for instance, to shift their assets from the production of a commercial crop to the generation of ES is a PES scheme. However, a money transfer to a conservation agency (be it a national park, a non-governmental organisation or a community-based ranger group) for the generation of ES is a subsidy. There is no change in output composition, but a change in production volume. This distinction, largely overlooked in the literature, is important because these mechanisms are meant to address different problems: PES target the generators of negative externalities, while subsidies, for instance, support agents providing public benefits in the form of positive environmental externalities. Disregarding the fundamental features of these instruments runs the risk of promoting ineffective policies for conservation and development. In Australia, for instance, several government grant programmes targeting farmers to promote conservation have had negligible environmental outcomes while costing taxpayers several hundred million dollars (Kingwell et al. 2008). A better understanding of the different features of these instruments is also fundamental when designing policies to address emerging threats such as climate change. For instance, policymakers need to recognise the likely time delays associated with changing output composition rather than sponsoring agents that already generate some desired environmental outcomes. In addition, policymakers need to consider the likely impacts on every sector of society of shifting inputs to other productive uses, so as to select the instruments that better fit society's preferences and priorities. Finally, one needs to consider that the lack of secure property rights for many indigenous communities is at the core of their development problem, and the issue of property rights cannot be assumed away by proposing PES programmes.

In order to encourage changes in output composition, PES programmes require that secure and transferable property rights over ES are clearly and unambiguously assigned to landowners rather than to the public. This requirement sanctions the user-pays principle: users must pay for the ES they enjoy, and suppliers must be compensated for delivering them (Engel et al. 2008; Pagiola et al. 2008). From a development point of view, this is what makes PES attractive: market exchanges of ES owned by economically disadvantaged individuals or communities have the potential to improve their livelihoods. Security and transferability of property rights over ES are linked to security of land tenure that in turn affects the availability of real alternative uses for landowners' assets. Tenure issues are indeed critical for participation in PES programmes (Pagiola et al. 2008), as lack of secure and transferable rights hinders investments, changes in assets' production role, disinvestment and exit from a market. As noted by Van Hecken and Bastiaensen (2010), landowners must have alternative uses for their assets other than the provision of ES so that they can voluntarily choose the most profitable option.[4] Security of tenure, and hence the existence of profitable alternative uses for the landowner's assets, gives credibility to threats to withdraw from market transactions. That is, high opportunity costs of conservation put the agent in the position to exercise power in markets for ES.

[4] Unlike Van Hecken and Bastiaensen (2010), I do not regard freedom to choose among real alternatives or the fact that a supply response is triggered by a price incentive as the features distinguishing PES from command-and-control approaches. Facing a pollution tax, for instance, a firm could choose to pay the tax, invest in cleaner technology or exit the market. Also, a subsidy to buyers increases the price they are willing to pay, hence increasing the sellers' revenue and making production economically viable. PES aim to change the composition of the firm's output, while command-and-control measures, taxes and subsidies affect revenues, volume and production costs.

There are marked differences from this conceptual framework of PES and the economic and cultural context of indigenous communities in remote regions of Australia. First, in many cases, the benefits of NRM activities for indigenous landowners can be assumed to be higher than benefits from other forms of resource use. This reflects two factors: (a) large areas of indigenous land have low natural fertility (Greiner et al. 2009; Luckert and Whitehead 2007); hence, indigenous NRM has low opportunity costs, and (b) cultural beliefs and philosophies underpin indigenous ecological knowledge and drive indigenous NRM. Indigenous philosophies and ecological knowledge are then important inputs in the generation of environmental, social and health benefits for indigenous people (Altman 1987; Williams 1986). The implication of these factors is threefold. On one hand, it means that indigenous landowners potentially have a comparative advantage in environmental service provision. They can provide environmental services at the highest relative efficiency in terms of the other goods and services that can be extracted from their land. On the other hand, low opportunity costs and the private benefits of indigenous NRM for indigenous landowners imply limited bargaining power when negotiating payments for environmental services. Under these conditions, indigenous landowners may actually become 'forced to trade' (Muradian et al. 2010) and sell cheap, not out of choice but out of lack of power (Martinez-Alier 2002). Further, as indigenous landowners have limited alternatives to NRM, there is little scope to intervene on output composition through PES. Rather, increasing ES from indigenous NRM requires grants and subsidies.

Second, indigenous management practices are usually considered sustainable. Hunting and fishing according to indigenous customs ensure that resources are not exhausted (Dhimurru 2006). Williams (1986), for instance, describes a set of indigenous harvesting activities—including fire management, fish trapping and gathering bush products—that are meant to avoid waste, assure regeneration and maximise productivity of the land. Still, indigenous landowners are dealing with several environmental issues in their lands, such as species decline and changing landscapes (Woinarski et al. 2007). There are several causes, such as the breakdown of precolonial indigenous NRM and European models of agricultural exploitation (see Wilson et al. 2010). Alien species such as mission grass and buffaloes were introduced for commercial and agricultural purposes (Parsons and Cuthbertson 1992; Smith 1995). Research has demonstrated the negative impacts of introduced species such as pigs and water buffaloes on indigenous harvest of native species (Bradshaw et al. 2007; Fordham et al. 2006; Lonsdale 1994). These environmental effects are negative externalities created on indigenous landowners by other economic agents. Depopulation is also a cause of declining environmental conditions (Altman and Whitehead 2003). As many remote areas are no longer populated, fire regimes have radically changed, resulting in increased carbon emissions (Ritchie 2009). It may be argued that depopulation is partly a symptom of increasing opportunity costs for indigenous people. Residing in remote or very remote areas requires foregoing economic opportunities that urban settings may offer. According to Biddle (2010), however, indigenous people that move to urban areas do not do as well—in terms of employment—as those already residing there and may do worse than those that stayed in remote centres. Depopulation, invasive species and breakdown of precolonial indigenous NRM are direct consequences of policy distortions and the failure to charge economic activities to reduce negative externalities. Removing such distortions is the obvious first-best solution (Heath and Binswanger 1996) for solving environmental problems in the indigenous estate.

Third, indigenous land tenure is regulated by two systems. Under the Native Title Act 1993, native title includes a bundle of rights and interests over land and water. These rights are recognised by law conditional on providing evidence of a continued system of traditional law and custom giving rise to these rights and can be extinguished after consideration of the merit of any conflicting legislation creating new property rights (Commonwealth of Australia 1993). Further, the jurisprudence holds that native title is recognised only for personal, domestic and non-commercial purposes (Gerrard 2008). Hence, native title legislation limits indigenous people's ability to underwrite economic enterprise, possibly precluding access to ES markets as well, unless they provide incidental economic advantages (Gerrard 2008). Under the Aboriginal Land Rights (Northern Territory) Act 1976, as well as other state-level land rights legislations, indigenous communities that successfully claimed their traditional land are granted with it under inalienable freehold title (Commonwealth of Australia 1976). The title to indigenous land is held by a land trust for the benefit of indigenous landowners, and it is a communal title that cannot be bought or sold (Central Land Council 2007). It is nowadays recognised that land rights legislation provides more secure land tenure to indigenous people than the Native Title Act, and it does not hinder indigenous aspirations to participate in economic activities (Gerrard 2008).

On theoretical grounds, it is questionable that PES programmes and participation in ES markets are appropriate instruments to address environmental problems and development issues in indigenous communities. PES and ES markets require secure land tenure, but many indigenous communities lack property rights over their ancestral lands. PES schemes target the generators of negative environmental externalities, but many serious environmental issues on indigenous land are not caused by indigenous activities; hence, indigenous landowners may not be able to effectively tackle environmental problems in the lands. Further, low opportunity costs of many indigenous lands, and the cultural responsibilities to protect environmental resources limit the ability of indigenous landowners to use their endowment for alternative uses; as a result, indigenous ES providers may not have negotiating powers and the ability to threaten to withdraw from market transactions.

The clear mismatch between the theoretical requirements of PES schemes and the economic, cultural and environmental conditions of remote regions under indigenous ownership may suggest that labelling any approach to environmental conservation (including grants and subsidies) as PES is more a political expediency than a rigorous application of the theory. Incidentally, this implies the endorsement of the neoliberal discourse of privatising public environmental goods and services, of shifting financial responsibilities for development to the private sector by granting rights over public goods and of increasing efficiency of public expenditure (Büscher 2010). I now turn to the experience of two indigenous land and sea management groups with PES and other similar arrangements in order to assess if this theoretical mismatch is relevant on practical grounds.

9.4 The Two Study Areas

The two indigenous land and sea management organisations that participated in this research are based and operate in Arnhem Land, in the far north of the Australian continent. They operate as corporations, that is, as service providers with limited liability for the benefits of indigenous landowners. Both organisations manage vast areas of indigenous-owned land held by the Aboriginal Land Trust as disciplined by the Land Rights (Northern Territory) Act 1976. They have a secure system of land tenure. These areas include two Indigenous Protected Areas (IPAs). An IPA is a land voluntarily assigned to the protection of biodiversity and conservation of heritage and cultural resources by the indigenous landowners. IPAs have no legal status, even though they are part of Australia's National Reserve System. Federal and state governments support IPAs through a series of grant programmes.

9.4.1 Dhimurru Aboriginal Corporation

Yolngu[5] people established Dhimurru in 1992 to monitor and minimise the impact of an increasing nonindigenous population that followed the establishment of a bauxite mine and processing plant on their traditional lands. Yolngu people run and control the organisation through the Dhimurru Board. It includes representatives of 17 clans with interests in the region. A Yolngu managing director, a senior cultural advisor, Yawarrin (men) rangers, Miyalk (women) rangers and a permit officer are responsible for Dhimurru's daily operations. Nonindigenous staff includes an executive officer, three project facilitators and an administrative personnel. Dhimurru currently employs 16 indigenous and 6 nonindigenous staff members.

In 2000, Yolngu people declared the Dhimurru Indigenous Protected Area. The IPA covers around 92,000 ha of land and 9,000 ha of adjacent marine areas in the Gove Peninsula (Fig. 9.1). The IPA contains areas of important cultural and environmental values, hosting a significant representation of Australia's Arnhem Coast sub-bioregion ARC-3 (Department of the Environment, Water, Heritage and the Arts 2010a, b). Environmental values include high plant diversity, intact faunal assemblages and significant feeding and nesting sites for threatened species of marine turtles and seabirds (Dhimurru 2008). The Dhimurru IPA surrounds land leased to Rio Tinto Alcan for bauxite mining and processing and the townships of Nhulunbuy, Yirrkala and Gunyangara.

The primary focus of Dhimurru's activities is the protection and enhancement of the natural and cultural values of the IPA (Dhimurru 2006, 2008, 2009). Dhimurru fosters 'both-ways' management by integrating Yolngu and nonindigenous sciences. The IPA is also managed according to IUCN category V guidelines for protected

[5] Yolngu are indigenous people from east Arnhem Land.

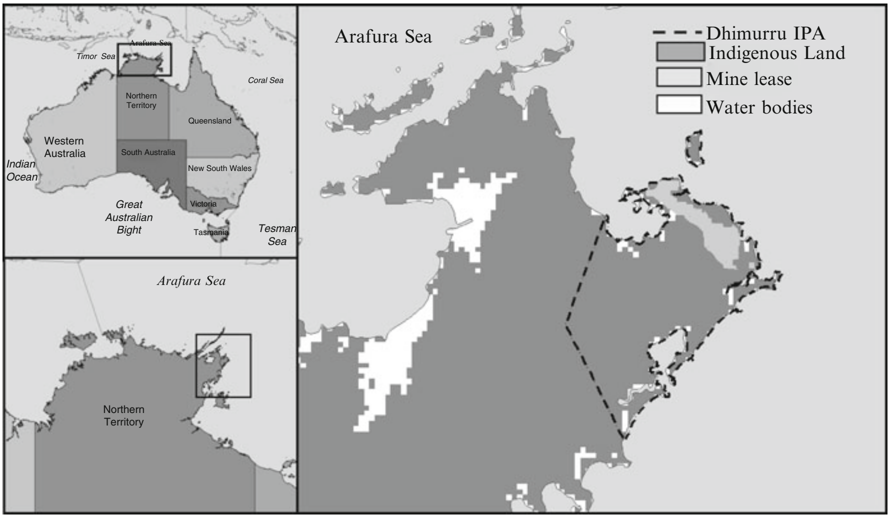

Fig. 9.1 The Dhimurru IPA

areas (Dudley 2008). Dhimurru's activities have interconnected goals: people management, environmental monitoring, conservation and restoration, and heritage and cultural protection.

Dhimurru limits and monitors the nonindigenous use of, and access to, recreational areas. Limiting access protects sites of cultural and environmental significance by avoiding damage caused by vehicle movements (spreading of weeds and non-native ants, fire scars, bush and tree damage, opening of new tracks, disturbance of fauna, damages to nesting sites, etc.) as well as by inappropriate behaviour (vandalism, extirpation of specimens). Environmental management and conservation include crocodile trapping, tagging and relocating; weed monitoring, treatment and eradication; discarded ('ghost') net recovery and turtle rescue; and beach clean-ups from marine debris.

Dhimurru's budget for the financial year 2009–2010 is around AUD2.3 million. Seventy-eight per cent of the budget comes from public funding—mostly from three programmes of the Commonwealth government: the IPA programme, the Indigenous Heritage Program and the Working on Country (WOC) programme. Income earned from fee-for-service contracts and private grants and revenues from permits and merchandise make up the remaining 22%. These resources are used to address environmental problems originating outside the boundaries of Dhimurru's jurisdiction. The inflow of nonindigenous people, for instance, is largely driven by the industrial development in the mine lease, over which the Yolngu people have no control. Weeds and ghost nets are a major threat to native flora and fauna, and again they are not linked to indigenous landowners' activities. Rather, they are negative externalities generated by other landowners or economic agents.

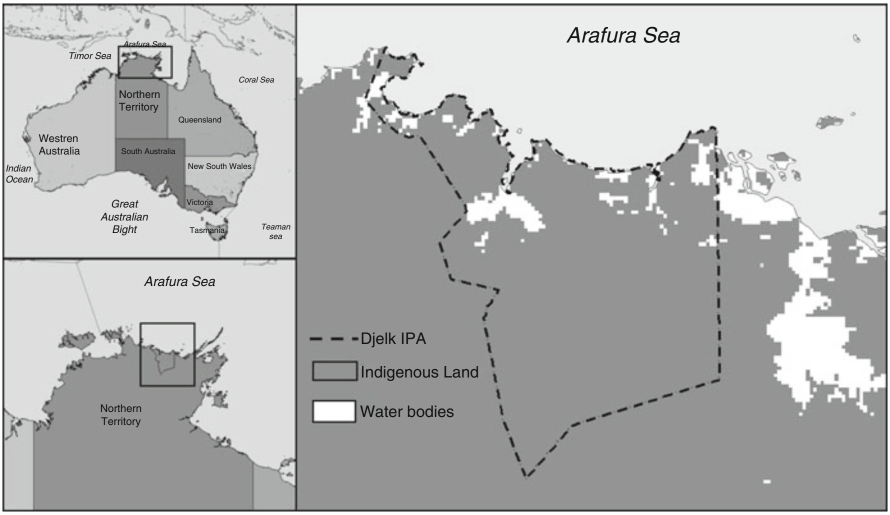

Fig. 9.2 The Djelk IPA

9.4.2 *Djelk Rangers*

Djelk rangers were established under the auspices of Bawinanga Aboriginal Corporation in 1991. The Djelk IPA was declared in 2009. It extends over 673,200 ha stretching from the central Arnhem Plateau to the Arafura Sea in the Arnhem Coast sub-bioregion ARC-2 (Fig. 9.2). The Djelk IPA comprises a biodiversity-rich landscape, home to iconic species such as saltwater crocodiles and the richest variety of reptiles in the world (Department of the Environment, Water, Heritage and the Arts 2010b). Senior indigenous owners guide and control the Djelk rangers and the management of the IPA through an advisory committee consisting of representatives of 107 landowning clans.

Djelk currently employs 35 indigenous rangers, a nonindigenous ranger coordinator and a special project officer. Rangers are divided in three groups (sea, land and women rangers). For the financial year 2009–2010, Djelk rangers had a budget of around AUD2 million, 83% of which comes mostly from the IPA and WOC programmes and 17% is revenue from fee-for-service contracts.

Major responsibilities of land and women rangers include fire management and selective burning, feral animal control and weed treatment. These activities aim at maintaining the biodiversity and the productivity of the land through the use and transfer of both indigenous and nonindigenous knowledge (Bawinanga Aboriginal Corporation 2009). There are substantial differences between Dhimurru and Djelk IPAs. Dhimurru rangers focus mainly on people management. The proximity of the Dhimurru IPA to the mine and processing plant and the large nonindigenous population pose the greatest threats to conservation of cultural and natural assets. Djelk's activities centre on the provision of environmental conservation services. However, for both ranger groups, their activities primarily aim to mitigate the impact of environmental problems that originated outside the boundaries of their jurisdictions. Efforts to control invasive species, for instance, are measures to alleviate the impacts of a negative externality.

9.5 Indigenous NRM and PES

Both Dhimurru and Djelk rangers are involved in several fee-for-service contracts with private companies and public agencies. Some of these contracts are PES arrangements as they aim to provide outputs that the rangers would not have provided otherwise—hence, they change the output composition using the same production inputs. These outcomes are ES, but in many cases, the link between contracted activities and environmental outcomes is not measurable, so payments are based on activities rather than on outputs. In other instances, the contracted activities have indirect environmental outcomes but aim at supporting natural resource management. Hence, the contractual arrangement is simply a payment-for-service scheme. However, the ranger's experience with these arrangements can still provide important insights on their ability to negotiate with private companies and public agencies the provision of services. When questioned about the need for these contractual arrangements, senior management staff in both organisations stated the necessity for stable funding streams that do not depend on governments' shifting policy priorities.

The Western Arnhem Land Fire Abatement (WALFA) project is an outcome-based PES scheme involving several indigenous groups—including landowners of the Djelk IPA and the Djelk rangers—and Darwin Liquefied Natural Gas (DLNG) (a subsidiary of Conoco Phillips, the third largest energy company in the world). The WALFA project is the first large-scale commercial provision of environmental services in indigenous Australia. According to this agreement, indigenous rangers and landowners implement fire management following indigenous cultural protocols and practices. The project goal is to reduce the number of highly destructive, high in greenhouse gas (GHG) emission, late season fires through prescribed early season burning across 28 million ha of Western Arnhem Land. The reduction in GHG emissions and enhanced environmental protection offset the environmental impact of the DLNG gas plant in Darwin. The target reduction of 100,000 tonnes of CO_2 equivalent per year relative to a 10-year baseline (1995–2004) has been regularly exceeded (May et al. 2010). In return, DLNG pays the traditional owners AUD1m per year (in 2006 dollars) for 17 years. This amounts to AUD10 per ton of carbon equivalent, but the unit price is susceptible to revision every 5 years. Djelk rangers received around AUD100,000 in 2009. The WALFA project started in 2005, even if the resources were made available in 2006. In August 2011, indigenous landowners and DLNG have started renegotiating the financial terms of the agreement.

Scientists and indigenous landowners worked for several years before the start of the project to accurately assess the quantity of GHG emissions abated through wildfire management. Unless this could be credibly and reliably gauged, it was unlikely that any abatement would receive financial backing. The Northern Territory Government then proposed it as an offsetting scheme as a condition for DLNG to obtain permission to build its plant. The Northern Land Council—an indigenous representative body—brokered the agreement on behalf of the indigenous groups involved in the project. For the Djelk rangers and the indigenous landowners, the motivation to produce the carbon credits was not entirely or exclusively monetary. Major motivational factors included the protection of the Arnhem Land Plateau from wild and highly destructive fires through the use of traditional burning practices, the transmission of traditional knowledge to the younger generation and assisting landowners to return to their traditional estate. The economic benefits of the WALFA project were also important, but they had a lesser role. Indeed, Djelk rangers receive around 10% of the total payments, even if they carry out most of the strategic burning in their IPA.[6] While successfully delivering the expected outcome, the project has also highlighted important issues relating to benefit sharing and potential conflicts between commercial and customary uses of natural resources. The lack of a national framework for carbon trading and emission abatement from fire management is also considered a major hurdle in scaling up the project (Whitehead et al. 2009a, b).

Australian Quarantine Inspection Services (AQIS) and the Northern Territory Department of Primary Industries, Fisheries and Mines run a ‘fee-for-services’ programme through which AQIS contracts indigenous landowners and rangers to provide weed, insect, illegal foreign fishing vessels (IFFV) and marine debris monitoring services (Muller 2008). AQIS pays for vehicle and vessel time and provides full pay for up to two rangers to collect samples and patrol the coasts. AQIS contracts run from year to year, and they offer no support for start-up costs, such as purchase of vehicles and vessels. As the programme is offered to indigenous landowners all over the Northern Territory, AQIS needs to make sure that the data is collected with the same techniques and frequency across a very large area. Hence, AQIS contract is a take-or-leave arrangement. Indigenous landowners have no ability to negotiate the terms of the contract. They can only decide if and when to perform the contracted activities. There is little data about the outputs and outcomes of this programme. Dhimurru earned around AUD8,000 in 2008–2009 and around AUD1,500 in 2009–2010 from the AQIS contract. For the same financial years, Djelk rangers earned, respectively, AUD50,000 and AUD11,000. As the rangers’ budgets are over AUD2 million, the AQIS fee-for-service scheme has clearly little financial impact. The drastic decrease in revenues from AQIS for both organisations is due to AQIS budget cuts and new biosecurity priorities. In 2010–2011, AQIS has contracted Dhimurru and Djelk only for IFFV and marine debris monitoring. During informal talks with AQIS officers, it emerged that indigenous rangers are not always willing

[6] The remaining 90% is distributed to indigenous landowners and to the Warrdeken rangers that manage the Warrdeken IPA.

to take up weed and insect monitoring, but are usually eager to participate in marine debris patrols. One could speculate that little indigenous knowledge and practice is involved in collecting weed and mosquito samples and that the financial incentives alone are not enough to motivate the rangers to undertake AQIS activities.

A third scheme is the indigenous ranger programme run by Australian Customs and Border Protection Service (Customs). Under this programme, Customs engages indigenous rangers in maritime surveillance and biosecurity services. The set of environmental services to be provided by the rangers is not clearly specified. The programme started as a pilot project in 2005 with a fee-for-service agreement between the Djelk rangers and the Customs. Under the agreement, Djelk rangers initially received around AUD250,000 to employ two rangers to carry out 150 patrols per year. In 2009–2010, the Djelk rangers spent nearly 2,000 h patrolling approximately 10,000 km^2 of sea, including islands, up to three nautical miles off the coast of the Djelk IPA. Their activities focused on surveillance and marine debris control. In the last financial year, Djelk rangers received around AUD272,000. Thanks to these resources, they have been able to set up their sea ranger group to carry out coastal monitoring and protection of marine sacred sites, as prescribed by their cultural responsibilities. While the terms of the contractual agreement are non-negotiable, the Djelk rangers have a high degree of discretion and flexibility in carrying out the contracted activities. For them, coastal patrolling is usually a multipurpose activity, often linking surveillance with cultural activities. Djelk rangers have intercepted several illegal fishing vessels and provided evidence for successful prosecution, and these outputs are the results of engaging indigenous rangers in ways that match their cultural obligations. In 2007, the scheme was extended to involve other ranger groups in the northern Australian coast through a AUD623,000 commitment by the federal government (Australian Customs Service 2007). Dhimurru asked to be involved in the project, but the Customs has so far not included them in its indigenous ranger programme.

Since 2005, Rio Tinto Alcan (RTA—one of the largest mining multinational in the world) has contracted Dhimurru to carry out some ethno-ecological monitoring in the bay adjacent to the RTA bauxite refinery and shipping facilities, on the northwest border of the Dhimurru IPA. The contract requires Dhimurru to develop and provide ongoing maintenance of an ethno-ecological database, as well as supporting sampling activities in the bay. RTA committed to regular payments of around AUD40,000 per year. Dhimurru's major interest in the contract was to improve rangers' skills and capacity in the view that this would help them to provide ES to other customers. The contract expired last year and has not been renewed. According to RTA, the global financial crisis forced them to rescind the contract.

9.6 Assessment of PES and Fee-for-Service Contracts

This rangers' experience in trading environmental and other services, albeit limited, highlights several issues related to power in market transactions, independence from shifting policy priorities and motivations of indigenous NRM. One of the alleged benefits of PES schemes is the increased independence from government funding (Pagiola et al. 2008). In the case of the WALFA project, this independence from government funding comes at the cost of relying on governments to set up the framework that identifies buyers and forces them into market transactions. Without government pressure, potential buyers may participate in ES markets for philanthropic reasons, motives of corporate responsibility and scientific needs. When other business priorities emerge, these agents can easily withdraw from the market, as in the case of AQIS and RTA and their contracts with the Dhimurru rangers. Hence, the indigenous landowners' ability to negotiate in environmental service markets needs the backing of the government. When this is lacking, contracts are bound to be short term, driven by the buyers' needs, with minimal consideration of indigenous interests and cultural priorities. Without a policy framework that forces agents to trade ES, buyers are located on the short side of the market and hence able to exercise power. This hardly seems like a good recipe for a sustained improvement of socio-economic conditions of indigenous communities.

However, a policy framework may not be in itself sufficient for indigenous providers to gain power in the ES market. There is no guarantee that indigenous landowners end up in the short side of the ES market or that they can successfully compete with other ES providers. As a policy framework for ES provision creates a demand that in turn creates a supply, it is plausible that new agents would enter the markets. Indigenous landowners may have some cost advantages—location, indigenous knowledge, etc.—but they may be outcompeted in areas such as technology adoption and marketing. Further, it is hard to predict the form of the ES market. For instance, without international coordination, an ES market in one country may drive potential buyers to move their investment in other countries. The resulting ES market could be a monopsony, where one or very few firms have some market power on the price they pay for ES. In short, the possible scenarios resulting from the creation of ES markets are multiple. The final outcome for indigenous ES providers can only be assessed empirically once such markets are established. Of course strong government regulation may favour indigenous ES providers. As labour markets are regulated to protect the workers—the weaker party in labour contractual arrangements—so should the regulation for ES markets protect the indigenous landowners. A clear legal framework for trading carbon credits and biodiversity services is needed to ensure that indigenous landowners are not the losers in market exchanges.

With or without a policy framework, however, it can be already noted that PES schemes may tie indigenous development to global markets. The example of Dhimurru's contractual relationship with RTA indicates that local indigenous economies may be subject to the volatility of resource prices and international trade, over which indigenous landowners have little agency—and possibly limited opportunities for hedging. Also, one should not disregard the potential adverse effects of exposing indigenous NRM to markets. Market exchange through PES can be realised only for environmental and cultural elements that can be commodified. This risks the conflation of a set of systemic cultural and environmental complexities into commercialised elements, with potentially detrimental effects on the system (Kosoy and Corbera 2010; Norgaard 2010). Indeed, indigenous rangers stress the strong links between environment and culture and demand the recognition of the environmental outcomes of their cultural activities (Dhimurru, personal communication, 2009). However, some indigenous landowners may want to embrace market exchange as a way to produce financial and cultural benefits (Comaroff and Comaroff 2009), as long as market exchange does not hinder indigenous control over resources (Morphy and Morphy 2009). Again, effective indigenous empowerment is a key factor to ensure indigenous interests are not overlooked.

As indigenous Australians may have different sets of incentives and cultural demands, one should not expect that indigenous landowners automatically take up PES. Monetary incentives may not be enough to ensure the delivery of contracted service, the undertaking of conservation work or the changes in management practices. Contracted activities have to guarantee a degree of autonomy in their implementation, so that they can fit around other cultural and environmental responsibilities, such as in the case of the WALFA scheme and Customs programmes. The activities required for the generation of the contracted environmental services need to match indigenous management practices. Whenever this match does not occur, and when conflicting cultural priorities are apparent, effective indigenous empowerment is required to ensure indigenous and nonindigenous interests converge (Albrecht et al. 2009).

9.7 Conclusions

Commercialisation of environmental goods and services through PES schemes is emerging as an alternative to other forms of economic participation of indigenous people in remote Australia. PES schemes have the potential to engage the human and knowledge capital to promote indigenous development and environmental conservation. This chapter has examined the theoretical framework of PES and market for ES and how it applies to environmental problems in the indigenous estate. It also contains a review of the experience in the use of PES of two indigenous ranger groups.

It is generally recognised that indigenous NRM is based on cultural and philosophical beliefs and that indigenous landowners have strong cultural incentives to conserve and protect environmental resources. Evidence suggests that indigenous landowners have low opportunity costs since the natural productivity of most of their estate precludes investments in resource extraction such as agriculture and forestry. Further, the evidence suggests that environmental degradation on the indigenous estate is linked to policy distortions and failures to price negative environmental externalities. Notwithstanding these problems, many indigenous landowners are undertaking NRM activities and thus provide a set of environmental public goods and services.

Given their low opportunity costs, and the cultural beliefs and philosophies driving indigenous NRM, indigenous landowners have little power to negotiate in market exchanges or with government agencies; they cannot credibly threaten to withdraw from ES markets. The experience, albeit limited, of PES and other fee-for-service arrangements in indigenous Australia seems to confirm that indigenous landowners have little capacity to negotiate environmental service delivery according to their ecological knowledge, cultural obligation and kin responsibility, as well as financial resources, unless buyers are forced by governments into negotiation.

The clear mismatch between the theoretical requirement of PES—secure property rights and targeting generators of negative externalities—and the conditions of indigenous-owned land in northern Australia, coupled with the lack of a legal framework for market exchanges of ES, leaves indigenous rangers exposed to market forces driving them into short-term contractual arrangements with little cultural relevance and possibly few social, environmental and economic benefits. PES programmes are drawn from textbooks of neoclassical economics and are possibly instigated by the neoliberal discourse of shifting governments' responsibility to the private sector and reducing public expenditure. PES cannot substitute for the lack of public investment on the indigenous estate, the lack of coordination of public policies, the removal of policy distortions and the resolution of externality problems that are often at the core of environmental and development problems in many indigenous remote communities.

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Chapter 10
A Nested Institutional Approach for Managing Bundle Ecosystem Services: Experience from Managing *Satoyama* Landscapes in Japan

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10.1 Introduction

In the past few decades, numerous institutions, which govern the behaviour of a group of individuals and interactions among them, have been established to address specific environmental and developmental issues as they emerged. These approaches to environment and development look at ecosystem services and human well-being in a compartmentalised manner, either from a socio-economic or from an environmental perspective. Furthermore, existing institutions are often designed to address issues related to specific ecosystem services, not taking into account interactions and trade-offs among different ecosystem services, multiple use of those services nor multiple user groups involved in managing resources (Armitage 2008; Berkes 2006; Folke et al. 2005; Rodríguez et al. 2006). Understanding the costs and benefits of management options for a range of ecosystem services is critical, and institutions need to be crafted to address a bundle of ecosystem services that directly and indirectly benefit the society (Farley and Costanza 2010; Kosoy and Corbera 2010).

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This chapter aims at analysing environmental institutions in the light of managing bundled ecosystem services. It starts by providing an overview of the concept and features that describe common goods and services. It then provides an analytical framework for designing institutional arrangements that takes into account the complex interactions between ecosystem services. In doing so, this chapter develops a concept of the new "commons" introduced by the Japan *Satoyama-Satoumi* Assessment in 2010, a system of co-management of ecosystem services and biodiversity within private and public land, particularly through the use of a traditional communal system of shared property, called *iriai*, applied in managing a certain type of landscape in Japan featured by the mosaic composition of different ecosystem types that are managed by humans to produce a bundle of ecosystem services called *satoyama* (JSSA 2010). Through conceptual discussions and descriptions of the case of the management of *satoyama*, this chapter illustrates the effectiveness of a nested institutional approach for managing a bundle of ecosystem services, taking into account the multiple uses and values of services, existence of multiple property regimes and multiple user groups.

10.2 Management of Commons

Issues related to the management of commons have been one of the key themes studied and discussed in the field of environmental governance. Formulating a generic definition of "commons" is not an easy task, as the term has been used by experts from different disciplines. Furthermore, the shared interests and values that define commons are themselves in constant flux, with fluid and often unpredictable groupings and initiatives across industries, historical public spaces and cultural identities (Holder and Flessas 2008), adding complexity to develop a generic definition of "commons."

In general, goods and services can be classified according to *excludability* and *rivalry*. For instance, a *non-excludable* good, such as the aesthetic enjoyment of a sunset or the health benefits of clean air, has benefits whose distribution is difficult to control (Brown 2007). These goods and services can often be examined as externalities (Heal 1999; Kaul et al. 1999). Take for example a city with a forested area. The forest filters and cleans local air, reducing smog, air pollution and associated health risks—these benefits are also felt by neighbouring cities whose municipal taxes do not contribute to the maintenance of the park. Stakeholders (ab)using global goods or services, such as the atmospheric sinks for carbon, are also not easily excludable.

Furthermore, an *excludable* good or service is one whose access can be limited (Brown 2007). A garden is *excludable*; a wall around the garden limits access. A flow of water down a river is difficult to exclude; it is constantly moving and thus requires additional effort to limit access, such as laws that regulate the volume of water that may be withdrawn per household. Access can be restricted through several means, depending on the institutions surrounding the good or service and the way it is treated by the market. Many goods and services are *excludable* through human institutions such as property rights; therefore, this property stems from a human relation with the resource rather than arising from the resource itself.

The second feature concerns the consumption of a *rival* good or service by one individual that diminishes the quantity or quality available to other individuals (Brown 2007). This is an inherent characteristic of the good. Examples can be applied or the atmosphere as a sink for carbon; the apple can only be eaten once, and each unit of carbon released into the atmosphere decreases its absorptive capacity for future emitted units.

A *non-rival* good or service can be consumed infinitely, because consumption of one unit *does* decrease the amount available to others (Brown 2007). These goods are not deemed scarce by the consumer and thus have no market value (Farley 2010). A good or service whose consumption is *non-rival*, such as enjoying the aesthetics of a garden or a sunset, is sometimes *non-rival* only to a certain point of congestion—the garden could have so many visitors at some point that it is less enjoyable for all. Thus, these congestible goods are treated as *non-rival* to a certain point of congestion, after which they are treated as spatially *rival* (Bergstrom and Randall 2010; Farley 2010).

The combination of these two characteristics applied to environmental goods and services leads to the division of these goods and services into a new classification with four different groups: private, public, commons and open access.

The commons have features defined by their *excludability*, indicating that their physical nature is such that controlling access by potential users may be costly, and *rivalry*, indicating that each user is capable of subtracting from the welfare of other users. These resources could include, for instance, an ocean ecosystem from which fish are harvested or a forest from which timber is harvested.

As indicated by Oakerson (1992), commons can have a fixed location, or it can occur as a fugitive resource and can be renewable or nonrenewable. They can be indivisible over large areas such as oceans and the atmosphere or others like small pastures. The key challenge for these commons is how to coordinate use by numerous actors to achieve an optimal level of their production or consumption.

The debates related to the management of commons were stimulated after the release of the article by Hardin (1968), “Tragedy of the Commons,” where he used the word “commons” to describe “common grazing land,” a pasture shared by local herders, without shared rules regulating its use. By illustrating the potential scenario of resource degradation caused by economically rational behaviour of herdsmen who keep increasing the number of livestock in an open pastureland to maximise their individual benefits while sharing the costs of overgrazing with all the other members, he argued that centralised government and private property could be the only solutions to manage commons sustainably in a long term. In his description of the problem of sustaining resources that everybody is free to overuse, Hardin also introduced multiple aspects of commons, such as the commons in food gathering (farm land, pastures, hunting and fishing areas), as well as the negative commons of pollution, such as the commons as a place for waste disposal or a sink for unwanted by-products.

Hardin's work was criticised later by various prominent scholars for not taking into account the fact that many social groups, including the herders on the common grazing lands, have struggled to manage their resources successfully by developing and maintaining self-governing institutions (Dietz et al. 2003; McCay and Acheson 1987; Netting 1976; Ostrom 1990). Criticism also came from researchers aware of the existence of diverse common-property institutions in the field, pointing out that Hardin failed to distinguish between common-property and open-access conditions where no rules existed to limit entry and use (Dietz et al. 2002).

As discussed by Ciriacy-Wantrup and Bishop (1975) and as discussed above, the concept of common property contains the nature of *excludability* of resources, whereby those who are not either owners themselves or have some arrangement with owners to use the resource in question are excluded. By using an example of hunting and gathering societies where the use of resources was regulated through informal institutions such as customs, taboos and kinship, Ciriacy-Wantrup and Bishop explained how these informal institutions could function effectively in managing commons. It is also worth noting the fact that policy reforms introduced by mid-1980 through transforming from governance of resources as common property by local communities to public and private governance in many cases made the situation worse, both for the resources and their users, through imposing a new set of rules that might not be considered legitimate locally (Dietz et al. 2002).

However, it should be recognised that in reality, a variety of property rights regimes are used to regulate the use of common-pool resources, ranging from the broad categories of government ownership (*state property*), private ownership (*private property*) and communal ownership (*common property*). When there are no clearly defined property rights on the users of the resources and regulations of their uses, a common-pool resource is under an *open-access* regime (Dietz et al. 2002). In practice, resources are often held in overlapping combinations of these regimes based on the degree of *excludability* and *rivalry* of resources, with variation within each. In many cases, combinations of property right regimes may work better than any single regime, and the success of local-level management, for example, often depends on the legitimization by central government. This type of nested systems and cooperative management arrangements is critical for sustainable common-property resource management (Berkes et al. 1989).

10.3 A Nested Institutional Approach

In the last decades, numerous institutions were established to respond to various environmental crises, such as Kyoto Protocol dealing with climate change or more recently an intergovernmental science-policy platform on biodiversity and ecosystem services (IPBES) being established as an international platform dealing with biodiversity and ecosystem services from a global scale. However, many of these institutions tend to focus on single problems, exemplified by various multilateral environmental agreements that are designed to address single problems, ignoring system-wide interactions and complexities. For example, addressing climate change under the framework of the United Nations Framework Convention on Climate Change through forest plantations may bring about negative impacts on the state of biodiversity and ecosystems, which is addressed under the Convention on Biological Diversity. Thus, existing institutions tend to address issues in a compartmentalised manner, without paying sufficient attention to interactive effects of individual drivers (Walker et al. 2009). There is growing recognition that in order to address a wide range of environmental issues effectively, greater interaction among existing institutions is critical.

In their efforts to identify effective governance and institutional arrangements for managing common-pool resources, commons scholars have examined diverse resource system where multiple user groups exist. Their work has resulted in the development of design principles and enabling conditions for management of common-pool resources (Agrawal 2002; Ostrom 1990). However, the development of commons theory has been based largely on the studies of relatively simple local-level cases and single-use resource management regimes (Armitage 2008; Berkes 2006).

Recognising the importance of cross-level linkages and governing the commons as complex adaptive systems, various innovative approaches for governance of commons have been suggested by researchers, such as *adaptive co-management* (Olsson et al. 2004), which emphasises the role of local groups in self-organising, learning and shaping change and working with institutions and organisations across levels and scales and *adaptive governance* (Folke et al. 2005), which recognises the importance of the roles of social capital, focusing on networks, leadership and trust. Furthermore, *polycentric or multilayered governance approaches* (Ostrom 2005) highlight the need for increased attention to vertical and horizontal linkages that allow social actors and institutions to respond to change and adapt and cope with uncertainty, and *resilience management* (Walker et al. 2002) proposes a framework for stakeholders to analyse resilience in social-ecological systems (SESs) as a basis for managing resilience.

A nested institutional approach discussed in this chapter is one that allows institutions to coordinate horizontally across geographic space to manage a mosaic of ecosystem types that produce a bundle of ecosystem services, while enabling institutions to also interact vertically to manage the provision of ecosystem services across political boundaries and secure an even distribution of those services across actors. A nested institutional approach attempts to bring some degree of consistency and coordination both vertically and horizontally among existing institutions and builds on the idea that goods and services should be managed according to their degrees of *rivalry* and *excludability*.

Figure 10.1 describes types of goods and services according to their degree of *excludability* and *rivalry*. *Excludability* relates to the costs of demarcating the good or benefit stream and formulating the necessary rights so that only the owner can control it. *Rivalry* in use or consumption implies that when someone uses a good, others cannot use it as well (Vatn 2005). As discussed previously, according to the degree of *excludability* and *rivalry*, resources or goods are broadly categorised into *private goods*, *open-access goods*, *common goods* and *public goods*. The figure also

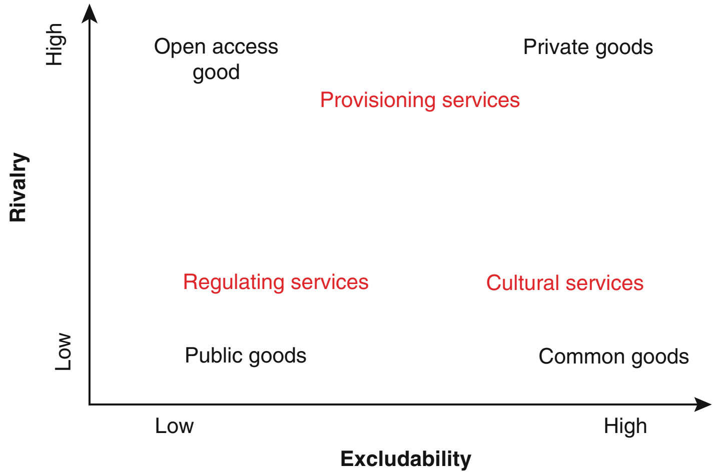

Fig. 10.1 Characterisation of goods based on concepts of *excludability* and *rivalry* in use or in consumption and types of ecosystem services (Adapted from Randall 1983)

attempts to map different types of ecosystem services according to different types of property regime. Although it is difficult to come up with general categorization, many *provisioning* services, such as food, fuel and timber, tend to be managed under private property or markets, while many of the *regulating* services, such as climate regulation, protection against storms and protection against noise, tend to be managed under the state/public property. Some of the *cultural* services, such as the provision of cultural, historical and religious heritage, are managed under the common property.

The above understanding considers nested and hierarchical institutions to manage the new commons. In other words, a market system that manages *provisioning* services which are characterised by high levels of *rivalry* and *excludability* should be controlled by an intermediate level of collective institutions that take care of goods and services that are *non-rival* and difficult to *exclude*; this will include most of the *regulating* and *cultural* ecosystem services (See Fig. 10.2). Therefore, contrary to the past where public and communal institutions managed public and collective goods and services and contrary to the new fashion with markets as the single most prevalent institution to manage all kinds of goods and services, we argue that an arrangement of various institutions comprising private, communal and public institutions is needed to produce the necessary conditions to maintain and deliver those goods and services that vary in degrees of *rivalry* and *excludability* but are however provided within a landscape. This approach will therefore lead to the maintenance of critical natural capital that is necessary for the sustainable supply of these services across temporal and spatial scales.

However, and despite the relevance of this nested institutional approach that takes into account the multiple uses and values of services, existence of multiple property regimes and multiple user groups to manage bundle ecosystem services,

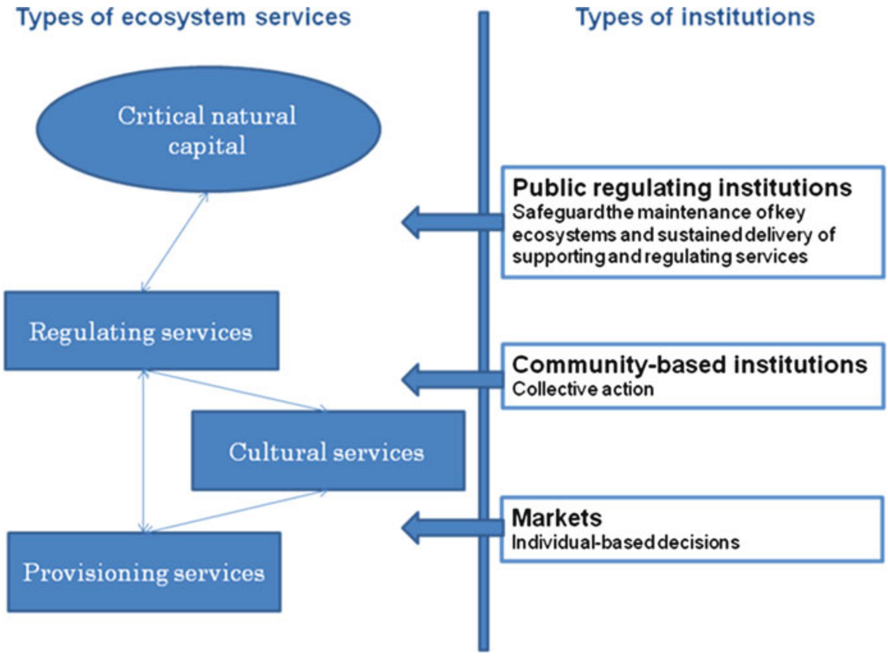

Fig. 10.2 Matching institutions and ecosystem services. The *column on the left* represents the ecosystem services provided within an ecosystem or landscape, and the *right column* represents the appropriate institution that manages each ecosystem service according to its definitional properties (*rivalry* and *excludability*)

we should not forget the importance of distribution and equity as structural outcomes from a fair environmental management system. Recent studies show that rising inequities in industrialised countries have led to the erosion of social capital (Wilkinson and Pickett 2009). The disruption of social practices can also be conducive to increases in environmental pressures as non-cooperative behaviour prevents collective solutions, promoting mistrust and resentment among social actors (Ostrom 1990). Equity must be independently analysed as outcomes from different institutional arrangements will vary in the degrees to which distribution of ecosystem services happens in practice.

10.4 A New Commons Management: Experience from *Satoyama*

Building on the previous work and knowledge accumulated by researchers in the field of the management of common-pool resources discussed in the previous sections, we now focus on elaborating the idea of the "new commons" introduced by the Japan *Satoyama-Satoumi* Assessment in 2010, a system of co-management of ecosystem services and biodiversity within private and public land, applied in managing *satoyama* (JSSA 2010; Takahashi et al. 2012).

The shift from purely conservation-focused activities to sustainable use increases the potential use of traditional practices in increasing the resiliency of local ecosystems and therefore the supply of ecosystem services for human well-being. The case study presented here describes the management of a type of landscapes in Japan, where such traditional practices are applied.

Satoyama is a term that couples *sato*, which means village, and *yama*, which means woodland. *Satoyama* is a traditional concept in Japan associated with land, which emerged in the seventeenth century. As shown in the way, the term *satoyama* is formulated; the concept implies an intimate relationship between human communities and woodlands (Terada et al. 2010). *Satoyama* is a type of landscapes that comprise a mosaic of different ecosystem types including secondary forests, agricultural lands, irrigation ponds and grasslands, along with human settlements, which has been managed to produce bundles of ecosystem services for human well-being (JSSA 2010). These landscapes have been formed and developed through prolonged interaction between humans and ecosystems and are most often found in the rural areas of Japan (Duraiappah and Nakamura 2012). Being featured by its mosaic structure of the landscape consisting of a range of different ecosystem types producing a bundle of ecosystem services for a variety of different social groups, the *satoyama* landscape offers a unique opportunity to understand how different property regimes, including its traditional common-property regimes, operate within the landscape.

Satoyama possesses numerous significant values, which are derived from its ecological, social, cultural and economic functions through the use of the ecosystem services flowing within these areas. Besides its role of being a home for humans, *satoyama* pools various ecosystems—including agro, forestry, wetlands, grassland and coastal ecosystems—and biodiversity, to provide ecosystem services that contribute to human well-being. For instance, the ecosystems in *satoyama* provide direct-use values such as food, fibre, fuel wood and water among others. At the same time, *satoyama* also produces a number of indirect-use values that include flood and water regulation, water purification, cultural services and pollination among many others. In recent years, partly due to the decline in agriculture and forest industry in Japan, the public benefits of *satoyama*, many of them cultural and regulating services rather than provisioning services, have been increasingly recognised (Kumagaya and Endo 2011).

The values of these ecosystem services that contribute to human well-being differ across different social groups. For example, local communities value many of the direct uses like rice production, fish production and water regulation much higher than urban residents who might be able to acquire these services from other sources. Urban residents on the other hand might place high values on the indirect uses such as climate regulation and cultural services. These different values of ecosystem services held by different social groups influence the perceptions and attitudes towards *satoyama* and its use in preserving biodiversity and the sustainable supply of the different ecosystem services. Recognising and respecting these differences in perceptions and attitudes are important if *satoyama* landscapes are to be used to reduce the rate of loss in biodiversity and maintain the sustainable supply of ecosystem services.

However, there has been a rapid decline in *satoyama* in the last half century, caused by a rage of factors (JSSA 2010), one of which relates to the changing nature of land ownership. Historically, a communal system of shared property, called *iriai*, has been used in Japan to manage common lands (*iriai* lands or *satoyama*). At *iriai* lands, members of communities informally formulate strict management rules that are enforced effectively by rotational patrolling and severe punishment schemes, allowing them to manage their *satoyama* sustainably from which they extracted grasses, firewood and charcoal (Kijima et al. 2000).

This communal system of a shared property in Japan emerged between the thirteenth and sixteenth centuries (McKean 2002). During the Tokugawa period (1603–1868), the idea of common property and techniques of sound commons management further evolved, while the common lands also underwent series of crises due to their massive conversion to cultivated fields as well as serious deforestation (McKean 1991; Mitsui 2010).

After the formulation of the Meiji government in 1868, as part of the land-tax reform, the government introduced the legislation that aimed at replacing *iriai* lands either by private property or state property. It resulted in the weakening of the *iriai* system, which had supported farming villages in Japan for centuries (Mitsui 2010; Uzawa 2004). This trend of the decline in the *iriai* system continued through the post-war period due to rapid economic growth that caused the boom and subsequent depression of domestic forestry (Mitsui 2010).

Furthermore, during the industrialisation, with the migration of rural communities to urban centres, the *satoyama* landscapes became fragmented. Rapid urbanisation led to a physical loss of *satoyama* landscape because of its conversion to other uses such as housing and golf courses, as well as the decline in a rural population that reduced the number of people available to make use of and manage *satoyama* landscapes (JSSA 2010). This also contributed to the declines in the *iriai* system.

At the same time, availability of substitutes for forest products and the government policies promoting tree plantations in the 1950s led to the conversion of secondary woodlands in *satoyama* to tree plantation, and monoculture plantation gave negative impacts on some of the ecosystem services provided by *satoyama* such as water, flood control and soil erosion prevention and deteriorated the mosaic nature of the landscapes. Furthermore, the supply of cheap timber from foreign markets led to the weakening of the Japanese forest industry, leading to the abandonment of *satoyama* (JSSA 2010; Mitsui 2010; Yamashita et al. 2009). The abandonment of *satoyama* woodlands led to the decrease in biodiversity as well as negative impacts on scenic beauty and recreational potential, making *satoyama* prime targets for illegal waste disposal (Terada et al. 2010).

There are two types of *iriai* rights defined by Articles 263 and 294 of the Civil Code of 1896: one type where a group of local people has exclusive ownership and use rights, while in another type, a group of local people has collective-use rights over *iriai* lands owned by individuals or other entities. In the second type, the rights of common are exercised on lands owned by the state, prefecture, municipality or persons. *Iriai* rights cannot be formally registered with the government, but the rights are effective as long as practices of collective forest management continue in the *iriai* lands. However, the decline of *iriai* lands or *satoyama* as mentioned above, and particularly the diminished role of *iriai* lands in people's daily life and activities, has led to the situation where *iriai* right holders do not place much importance on their rights or even consider waiving their rights (Yamashita et al. 2009).

While there have been continued declines in *iriai* lands and diminishing role of *iriai* rights in Japan, there have been some initiatives in recent years that can be considered as a new type of *iriai* land management or the "new commons" management (Mitsui 2010). This trend was also captured in one of the findings of the Japan *Satoyama-Satoumi* Assessment (2010), where the importance of "new commons," understood as a system of co-management of ecosystems and biodiversity within private and public land, was highlighted.

These emerging initiatives try to manage ecosystem services provided by *satoyama* landscapes that are of public benefits, many of which are regulating and cultural services such as water purification, flood control and soil erosion regulation, as well as spiritual and recreational services. Based on the increasing understanding and emphasis among the public on the importance of these services for public benefits, as well as the growing recognition of the merits of *iriai*-type management systems, some new initiatives emerged in Japan that use the concept of *iriai* through active engagements of urban dwellers in maintaining and utilising *satoyama*.

One of such examples is a joint effort undertaken in Aso, Kumamoto Prefecture, among the association of local farmers with *iriai* rights and urban dwellers who have interests in protecting environment in Aso, facilitated by the incorporated foundation called Aso Green Stock and supported by the government, which aims to maintain and protect *iriai* lands and promote local agriculture, forestry and livestock industry. By joining these efforts, urban dwellers are given access to these common lands through being granted special *iriai* rights.

In Aso, most of the grasslands are owned by the public sector but have been managed through the *iriai* system (Mitsui 2010). However, in recent years, the number of *iriai* rights holders and households engaged in agriculture and livestock farming has continued to decrease (Ministry of Environment 2009). This new type of initiative is considered as one of the innovative approaches to protect and maintain *satoyama*, by building on the traditional *iriai* commons management system in Japan. Unlike the traditional *iriai* which is a closed system at the village level, this new type of *iriai* is an open system, where interactions and an active involvement of urban dwellers in maintaining and protecting *iriai* lands are encouraged.

As is the case in the initiative in Aso, in undertaking efforts to manage *satoyama* by involving local communities and urban dwellers, the role of the intermediary type of organisations with support from the public sector under the appropriate legal framework is critical. Although there are some local ordinances on *satoyama* conservation, there are no integrated legislations or *satoyama*-conservation laws that aim directly at conserving and managing *satoyama*. Till the 1980s, the main mechanism used at the local level for the conservation and management of *satoyama* was the procurement of land around urban centres and the promotion of tree planting and greening this space. Then from the 1990s onwards, protection of secondary forests was promoted by procuring hilly areas and forests on hilly areas. The 1990s also saw a wave of legislation whereby federal authority was decentralised to regional governments giving the opportunity of local ordinances to take initiatives that were more place based and relevant to their respective constituencies (Duraiappah and Nakamura 2012).

In the 2000s, various local governments took the initiative to establish specific *satoyama*-conservation ordinances with the primary goal of conserving and preserving *satoyama* landscapes for their scenic and cultural heritage. These local ordinances outlined the rules of implementation of governance with participation from citizens and non-profit organisations in each specified region or unification of regions to be conserved and managed. Although many of these rules did not explicitly state the conservation of *satoyama* in their respective ordinances, the principle outlined in these local laws related to the use of agricultural land and the promotion of urban agriculture bore relevance to the management of *satoyama* landscapes. This type of support, including financial support by the public sector for citizen's groups and non-governmental organisations that are actively engaged in the protection of *satoyama*, as well as the promotion of the collaboration between communities and urban dwellers, is critical (Ishiura et al. 2005; Uesugi 1998).

Another area where legal options could be provided to promote the conservation of *satoyama* landscapes is related to land ownership and use, particularly through the Parks Law. In the case of Japan, national parks have a special connotation. In 1957, the "Parks Law" was introduced which gave legal status to quasi-national and prefectural parks. Unlike other countries, production activities with local communities actively involved in agriculture, forestry and other production activities are permitted. The ownership of the land in these protected areas is also not public but a myriad of ownership regimes including private lands. The national park systems in Japan offer a first step in providing an institutional landscape for addressing the multiple services provided by *satoyama* landscapes and used by a variety of stakeholders. However, the amount of land under national parks is still relatively small in comparison with the total land in Japan. The total area of Japan's national parks amounts to 2,065,156 ha and consists of land owned by the government (1,278,844 ha; 61.9%), local governments (253,257 ha; 12.3%) and private owners (533,026 ha; 25.8%). Parkland set aside exclusively for national parks and put under the ownership of the Ministry of the Environment accounts for only 0.2% (4,695 ha) of the total area of the national parks (Norihisa and Susuki 2006). The challenge is to scale up the concept of national parks to a level that is commensurate with the scale needed to provide the ecosystem services critical for well-being including in particular the regulating and cultural services that markets will have problems assigning prices.

10.5 Conclusions

Existing institutions are often designed to address issues related to specific ecosystem services in a compartmentalised manner. However, as described in this chapter, previous research undertaken by commons researchers has shown that when designing institutions, it is critical to take into account the interactions and trade-offs among different ecosystem services, multiple use of those services and multiple user groups involved in managing resources.

This chapter discussed different property rights regimes that are used to regulate the use of common-pool resources, including government ownership (*state property*), private ownership (*private property*) and communal ownership (*common property*), highlighting features of resources managed under these regimes, from the perspectives of *excludability* and *rivalry*. A review of various literatures revealed that in reality, a variety and the combination of property rights regimes are used to regulate the use of common-pool resources, and in fact, resources are often held in overlapping combinations of these regimes, with variation within each. Such nested systems and cooperative management arrangements are deemed critical for sustainable common-property resource management.

Building on this finding, this chapter elaborated on the nested institutional approach, which allows institutions to coordinate horizontally across geographic space to manage a mosaic of ecosystem types that produce a bundle of ecosystem services and take into account the complex interactions between ecosystem services. An example of the management of *satoyama* (or *iriai* lands)—a type of landscapes in Japan composed of a mosaic of different ecosystem types including secondary forests, agricultural lands, irrigation ponds and grasslands, along with human settlements, which produces bundles of ecosystem services for human well-being—was introduced to describe the nested institutional approach. As described in this chapter, *iriai*—a communal system of shared property that has been used in Japan to manage common lands—has been applied in managing *satoyama*, where both public and private lands exist.

In recent years, the importance of public benefits of *satoyama* has been increasingly recognised, and the merits of *iriai*-type management systems have also been realised. Based on this trend, some initiatives emerged in Japan which use the concept of *iriai* through active engagements of urban dwellers in maintaining and utilising *satoyama*. These emerging initiatives try to address ecosystem services provided by *satoyama* landscapes that are of public benefits, many of which are regulating and cultural services such as water purification, flood control and soil erosion regulation, as well as spiritual and recreational services provided by ecosystems. In this sense, it is an example of the active use of a commons management approach in public and private lands, which, unlike the traditional closed system, is an open system where active interactions between communities and urban dwellers across spatial scales exist. The importance of the support provided by the public sector in promoting these innovative approaches, through legal frameworks, was also highlighted in this chapter. The discussion in this chapter is a first step in conceptualising a nested institutional approach for managing a bundle of ecosystem services. Further research is required to elaborate on the concept through careful examination of various other examples from the world.

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Chapter 11
Institutional and Historical Analysis of Payments for Ecosystem Services in Madagascar

Cécile Bidaud, Philippe Méral, Fano Andriamahefazafy, Georges Serpantié, Louison Cahen-Fourot, and Aurélie Toillier

Abbreviations

AGIRE	Amélioration de la gestion intégrée des ressources en eau dans la Haute-Matriatra
ANGAP	Association Nationale de Gestion des Aires Protégées (national protected areas management association)
APMM	Association pour la Protection des Montagnes de Madagascar (association for the protection of Madagascar's mountains)
CBNRM	Community-based natural resource management
CI	Conservation International
COFAV	Fandriana-Vondrozo Forest Corridor
CSPF	Commission Spéciale sur la Pérennisation Financière (special committee for sustainable funding)
ERI	EcoRegional Initiatives
FAPBM	Fondation pour les Aires Protégées et la Biodiversité de Madagascar (Madagascar foundation for protected areas and biodiversity)
GELOSE	Gestion Locale Sécurisée (secure local management)
GRET	Groupe de Recherche pour les Echanges Technologiques (research group for technology exchange) (professionals for a fair development)
HIPC	Heavily indebted poor country
ICDP	Integrated conservation and development programme

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IRD	Institut de Recherche pour le Développement (institute for research for development)
IUCN	International Union for Conservation of Nature
IWRM	Integrated water resource management
JIRAMA	JIro sy RAno MAlagasy (Madagascan public water and electricity company)
LDI	Landscape Development Initiatives
NEAP	National environmental action plan
NGO	Non-governmental organisation
ONE	Office National de l'Environnement (national environment board)
OPCI	Organisme Public de Coopération Intercommunale (public body for inter-commune cooperation)
PES	Payment for environmental services
REDD	Reducing emissions from deforestation and forest degradation
SNAP	Système National d'Aires Protégées (national protected areas system)
UNDP	United Nations Development Programme
USAID	United States Agency for International Development
USDA	United States Department of Agriculture
VOI	Vondron'Olona Ifotony (grassroots community)
WCS	Wildlife Conservation Society
WWF	World Wide Fund for Nature

11.1 Introduction

The international emergence of payment for environmental services (PES) has been considered a prominent feature of environmental policies since the mid-2000s, particularly in developing countries (Engel et al. 2008; Muradian et al. 2010; Farley and Costanza 2010; Wunder 2005; Pagiola 2008). First adopted in Central and South America (e.g. Mexico, Costa Rica, Ecuador and Bolivia), PES systems have now been introduced in Southeast Asia, China, Africa and elsewhere. From an analytical standpoint, such a diversity of cases gives ample scope for assessing and discussing these new instruments.

While most studies undertaken thus far focus on the way PES are managed, via monitoring and evaluation, more and more authors suggest adopting a historical perspective to examine how these new instruments have emerged. Vatn (2010) points out, "PES systems are not created in an institutional vacuum." He puts forward an idea expressed by Engel et al. (2008, p. 668) that "PES mechanisms are not created in a vacuum by social planners or economic theorists. They develop in particular environmental, economic, social, and political contexts, and are subject to the push and pull of many stakeholders (path dependence)." In a recent article, Daniels et al. (2010) stress the idea of "institutional path dependency" as a way to better understand the real impact of PES in Costa Rica. In this work it is proposed that the success of PES in that country is closely connected with forest management decisions taken years earlier. Legrand et al. (2011) develop a similar argument, and

Pagiola (2008, p. 712), through a historical reading of PES in Costa Rica, points out, "the PSA[1] program did not start from a blank slate." Pagiola argues that one reason for the success of PES in Costa Rica was that a system of forest management subsidies was already in place when PES was introduced.

These authors' institutional analyses of current PES implementation reflect a need to go beyond the simplistic view of PES as an environmental policy tool that may be appropriate at a particular time and place following a precise theoretical model. The path dependency approach represents a new way of accounting for the patterns of PES implementation encountered in practice. Popularised by Mahoney (2000) and Pierson (2000), the path dependency concept shows that institutions do not necessarily develop in a rational way determined solely by the search for maximum efficiency. On the contrary, the way they develop depends largely on past choices and subsequent current policies. There are several versions of path dependency theory, varying from a weak, intuitive description that simply states "history matters" to Mahoney's strong version emphasising the "lock-in" effects to explain the course of events. Other authors such as Thelen (2003) adopt a more intermediate position. They show that institutional change is usually incremental with institutions adapting to external contingencies and showing a high degree of inertia as they evolve. Practice shows that institutional innovations prompted by circumstances at a particular time do not take place in an institutional vacuum nor are they blocked by an irreversibly locked-in historical path. The result is institutional hybridisation, sedimentation and conversion. Other authors have stressed the trans-scalar dimension in their analyses of institutional change (Djelic and Quack 2007).[2]

PES analysts that follow a path dependency approach often find that policies promoting PES are simply old policies that have been reinterpreted. Though "history matters," it is not to be taken literally. It means that the way in which PES is (or will be) applied in practice largely depends on how existing institutions interpret it, incorporate it and adapt the innovation to their given circumstances.

This chapter illustrates the analytical method of an institutional path dependency approach taking Madagascar as an example, a country recognised as a biodiversity hotspot (Myers et al. 2000) and where conservation policies have occupied a central place for over 25 years. Given the recent emergence of PES in Madagascar, the country offers a highly pertinent case study for analysing path dependency.

The four and only ongoing water PES in Madagascar are analysed in this chapter.[3] Trained observation of PES implementation and events and semi-structured

[1] Pagos por servicios ambientales (PSA) – payment for ecosystem services (PES).

[2] "The complexity of path generation increases considerably when we move our focus from technology and organisational fields to national institutional systems. It increases even further if we treat national institutional systems as potentially open systems in the double sense that they interact with each other while being embedded or nested within transnational institutional structures" (Djelic and Quack 2007, p. 167).

[3] PES here is defined through Wunder's definition (2005) but differs in two aspects: the presence of intermediaries between providers and beneficiaries for the furniture of the ecosystem service and the fact that the water PES studied here within are not yet contracted. The PES definition here is therefore used in a broader sense.

interviews of key stakeholders at the national, regional and local level represent the core of our methodology.[4]

We begin with the history of environmental policy in Madagascar, highlighting the influence of international donors and resulting international standards and norms applied to a multitude of recent PES schemes. We then examine four water-related PES schemes and show how a historical analysis can provide relevant input for a PES analysis.

11.2 The Emergence of PES in Madagascar

Madagascar was a French colony from 1896 to 1960. After a socialist interlude from 1975 to 1989, the island nation then reopened to the world. Despite the support of the international community, which began in the mid-1980s with the first structural adjustment plans and was strengthened in the 1990s, Madagascar's economic situation has remained precarious until today. In 2010 it ranked 135th out of 169 in the UNDP index, among the "low human development" countries. At the same time, Madagascar is one of the 25 biodiversity hotspots identified by Myers et al. (2000) leading world conservation circles to take an intimate interest in the country.

The conjunction of rich biodiversity and a severe human and economic development inertia largely explains the influence of donors and conservation NGOs in the management of Madagascar's environmental (i.e. conservation) policy (Kull 1996; Gezon 2000; Duffy 2006; Horning 2008; Chaboud et al. 2007; Froger and Méral 2009; Corson 2012).[5]

11.2.1 Origins and Development of Environmental Policy as the Background to the Emergence of PES

Environmental policy in Madagascar began taking shape in the late 1980s and has advanced through three phases under a national environmental action plan (NEAP) prompted by the World Bank.[6] Aided mainly by the United States, France, Switzerland and Germany, the policy began (1990–1996) by strengthening government institutions and implementing the most urgent conservation actions, creating a national network of protected areas and establishing various agencies such as the *Office National de*

[4] This chapter draws on a number of ongoing studies conducted by the Serena programme, which receives funding from the *Agence nationale de la recherche* under the SYSTERRA programme (ANR-08-STRA-13) http://www.serena-anr.org/

[5] There is also the purely political dimension. Madagascar has suffered serious political instability since the early 1990s with a succession of four presidents between 1993 and 2009, each succession accompanied by serious social unrest, some governments being overthrown and one president impeached.

[6] The three phases are referred to as *Programme Environnemental* 1, 2, 3 or PE1, PE2, PE3.

l'Environnement (ONE, national environmental board). This first phase received US$85.5 million in external aid[7] and focused on the creation of national environmental agencies and integrated conservation and development projects (ICDP) implementation.

Following the international agenda, the emphasis of the second phase (1996–2002) was on decentralising natural resource management, as was happening in the rest of sub-Saharan Africa (Bertrand et al. 2006), and on examining how environmental actions could be funded in the long term. Donor funding for this phase, estimated at US$150 million, was used mainly to set up community-based natural resource management (Toillier et al. 2008).

The NEAP's third phase began in 2003–2004 with the announcement of the Malagasy president during the IUCN Park Congress in Durban to triple the extent of the country's protected areas.[8] A group of international conservation NGOs and donors known as the Durban Vision group played a central role for the implementation of these "new protected areas" and the resulting "National System for Protected Areas" (the older ANGAP[9] network plus the new protected areas). The principal concern of the Durban congress entitled "Benefits Beyond Boundaries" was of the seeking out of sustainable funding for protected areas worldwide. Around the same time, the World Bank published a cost-benefit analysis showing the positive economic returns on investment in protected areas (Carret and Loyer 2003).

The Durban congress, attended by a Malagasy delegation largely made up of conservation NGOs, highlighted the problem of ensuring conservation funding, echoing a number of international networks that were emerging at the same time (Carbon Finance Alliance, Katoomba Group, Forest Trends, Ecosystem Marketplace, etc.).

The design, pilot stages and implementation of Madagascar's PES schemes were undertaken gradually during this third phase. Several key points are worth noting. Firstly, PE3 with its focus on the funding problem was included in the NEAP from the outset. Secondly, as PE3 was progressing and the environmental plan as a whole came to the end, stakeholders realised that environmental actions would not be planned so strategically in the future. The NEAP has been assessed as only partly successful, and the World Bank has turned its focus on strengthening the national protected areas system.[10] As a consequence of the expectation of the impending end of a policy framework to structure future conservation actions, what emerged was a mosaic of projects in which NGOs engaged directly with local stakeholders

[7] Andriamahefazafy and Méral (2004) have shown that the Madagascan government was able to provide only 2% of the total funding to set up and manage the network of protected areas for PE1 and 15–20% during PE2, donors providing the rest.

[8] Speech by the President of the Republic of Madagascar at the Durban congress, interpreted by the IUCN as equivalent to the standard of 10% of each country's land area under protection, which for Madagascar meant an increase from 1.7 to 6 million hectares (Borrini-Feyerabend and Nigel 2005).

[9] *Association Nationale de Gestion des Aires Protégées* (national protected areas management association).

[10] PE3 should have ended by 2009 but due to the political crisis has been more or less suspended. World Bank and GEF just lend US$50 million to strengthen the protected area system.

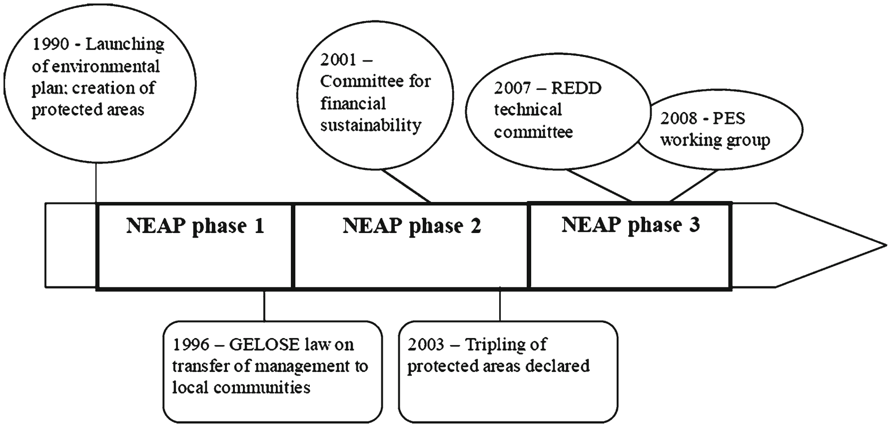

Fig. 11.1 Chronology of the main national-level environmental actions

(decentralised authorities, mayors, private operators, etc.). Thirdly, the climate change agenda and the REDD-plus mechanisms have increasingly captured the attention of the main national institutions encouraging the incorporation of market-based instruments (PES as an example) into environmental policy. Fourthly, political problems around the presidential elections of March 2009 led donors to retreat for diplomatic concerns. Conservation NGOs and private operators were then on their own to propose and implement environmental initiatives.

In 2008 a platform run by the Katoomba Group was set up for promoting and developing PES schemes.[11] Carbon capture came first, as the international influence has strongly favoured initiatives to control greenhouse gas emissions.[12] Biodiversity and water PES schemes then followed, still at the encouragement of donors, private operators and conservation NGOs (Fig. 11.1).

11.2.2 Embedding PES at the National Level

The general concept of environmental services (ES) and PES and the issue of sustainable long-term funding for such activities were taken into account from the initial planning stages of the environmental action plan in Madagascar (Andriamahefazafy

[11] The Katoomba Group is an international network of actors working to promote and improve PES schemes. Madagascar joined the Katoomba Group in 2008 at the instigation of WWF, WCS, IC and USAID, among others.

[12] Examples are the sales on the open market of carbon from the Markira protected area with IC and then with WCS; the funding application submitted to the World Bank BioCarbon Fund (BioCF) to buy emission reductions with the project to restore and conserve the Maromiza-Mantadia-Zahamena corridor; various REDD pilot projects (e.g. FORECA by GTZ and Intercoopération Suisse, PHCF by GoodPlanet/ActionCarbon and WWF).

Table 11.1 Goals of the NEAP

Environmental programme 1 1991–1997	1. Laying of foundations for environmental management (policy, legislation, build institutional capacity, etc.)
	2. Performing emergency operations (protect biodiversity in protected areas with neighbouring communities, soil conservation projects)
Environmental programme 2 1997–2003	3. Promoting sustainable use of natural resources such as soil, water, forest cover and biodiversity
	4. Reversing environmental degradation
Environmental programme 3 2004–2011	5. *Setting up sustainable funding mechanisms*
	6. Automating environmental actions
	7. Strengthening partnerships

Taken from ONE (2002) and Ministère de l'Environnement (2002)

et al. 2007, 2010). In fact, a long-term funding plan was one of the conditions laid down by donors under the title "Establishment of Sustainable Funding" (see Table 11.1). Similarly, the logical framework of the NEAP's third phase, drawn up in the early 2000s, mentions as a strategic goal the laying of "the foundations for sustainable funding of environmental management actions."

In the early 2000s, various initiatives were undertaken, always under donor leadership, to explore possible ways to finance environmental actions on a lasting basis.[13] A list of instruments for this purpose was drawn up at that time, namely, "(1) national public funds (government budget and public investment programme), (2) additional public funds through the HIPC initiative, (3) trust funds, (4) various levies from tourism, (5) taxes and environmental charges from different business sectors, (6) payments to ensure that forests supply ecological/environmental services, (7) private sector loans and donations, and (8) funds of donors' projects" (CSPF 2001, pp. 4–5).

While the term PES was still not central in the conservation lexicon, PES type mechanisms had been developed. One example is entry fees to protected areas used to support park managers' activities and local community development initiatives. The *Fondation pour les Aires Protégées et la Biodiversité de Madagascar* (FAPBM) was created under the auspices of the Ministry of Environment and financed by international donors.[14] The idea of generating revenue from environmental services

[13] In May 2001 ANGAP, IUCN and WWF organised an international symposium on sustainable funding for protected areas and other environmental programmes.

[14] In early 2001 the environment ministry set up a committee to create a fiduciary fund. The preparatory work for the fund was supported by the World Bank, CI, KfW, USAID and WWF. The commission opted to create a foundation, FAPBM. In late 2004, FAPBM allocated an initial capital of US$5 million from the Madagascan government (debt-for-nature swap with Germany), USAID, CI and WWF. Other donors contributed later, including the World Bank, AFD/FFEM, KfW and GEF/UNDP. US$35 million was collected between 2004 and 2008. The aim was to reach a capital of US$50 million by 2012.

was taking hold. Carbon credits and accrued interest on the FAPBM's invested capital were put forward as potential instruments for continued financing of the management of protected areas.

The first "conservation contracts" were set up by NGOs such as the Durrell Wildlife Conservation Trust (Durbin et al. 2001). The idea of taking water-related benefits into account in planning for sustainable funding to run protected areas was first suggested in a study by Carret and Loyer (2003) for the World Bank. According to this study (op. cit. p. 25), "one hectare of protected area in Madagascar would fetch an average net profit of $10 per hectare per annum, comprising $3 for biodiversity conservation, $4 for ecotourism and $3 for protecting the catchments' hydrologic resources".

11.2.3 From National to Local: Donors as Driving Force

During the 1990s a change in discourse was observed within conservation NGOs and affiliated ecology experts who began studying and advocating the use of water services provided by forest (Townsley et al. 2001). The decentralised authorities in Madagascar have embraced this discourse. Technical and financial collaborations between international donors and Malagasy decentralised authorities were particularly favourable in promoting the intuitive concept that forests "provide water services." Water resources in general have also began moving to the centre stage of international development as seen during the 2005 seminar dealing with the wealth creation from the corridors. "Water speaks to donors" was an admitted motivation of project managers to invest on water issue.

Gradually this institutional dynamic around the corridor, with USAID firmly in the driving seat, began to forefront the idea of environmental services. This rhetoric was used to instil the idea that the forest rendered services that were vital for the communities and development of local people.

Donor influence has not only played a central role in orienting Madagascar's national policy; it is also embedded in specific field projects undertaken by these actors. This is clear when looking at the speed and intensity of application of new national policy components, which vary widely between donors and donor intervention areas. In some cases, innovations in the field have manifested ahead of government policy, as is the case with PES.

One example of donor influence on policy and implementation is seen in the Fianarantsoa region where USAID and, to a lesser extent, conservation NGOs such as WWF and CI, have been present and vocal throughout the regional environmental planning period debuting in the early 1990s. Activities such as the creation of the Ranomafana National Park in 1991 and the WWF debt-for-nature swap in 1997 are examples of activities implemented within this political framework. The USAID financial support for conservation activities can be grouped into three phases: (1) the Landscape Development Initiatives (LDI, 1999–2003), which later became the (2) PTE (*Programme de Transition Eco-régional*) in 2003 and then finally the (3) ERI from 2004 to 2007. Subsequent projects under these phases, such as SAVEM and KEPEM, demonstrate clearly the important role USAID has played in providing institutional support for conservation efforts. In the Fianarantsoa region, American operations were based on the ICDP approach with the aim of combining conservation actions (conservation-based, community-based natural resource management (CBNRM), national parks, etc.) with community-based economic development actions. Actions to provide physical infrastructure such as roads, water points and fish ponds were undertaken to promote alternative livelihood activities to slash-and-burn farming such as beekeeping, fish farming and new rice cropping methods.

Since 1995, a new conservation tool emerged, the "corridors" paradigm, holding that habitat continuity is an essential condition for conserving biodiversity and ecological function (Carrière et al. 2011). Up until this period, environmental services for the benefit of household and community revenue had not yet perceptibly entered the discourse.

A historical analysis of Malagasy environmental policy illustrates how various players, with donors at the forefront, have gradually shifted towards the promotion of PES on both national (policy) and local (implementation) levels. The emergence of PES as an institutional innovation is found to have been incremental in nature rather than a sudden rupture. PES was part of a broader and deeper trend in environmental policy unfolding on the international stage that favoured ecosystem services and market-based instruments (Gómez-Baggethun et al. 2010). To understand more completely this analysis, it is necessary to identify and study the origins of the actual PES arrangements that emerged in Madagascar.

11.3 Payment for Water Services and the Emergence of Local Schemes

A complementary means to analyse PES aside from the path dependency angle is to look at specific processes that have emerged in particular places. For this, this work focuses on four pilot projects (see map): two drinking water PES projects in the towns of Antarambiby (Fianarantsoa) and Sahamazava (Andapa), a hydroelectric PES in Tolongoina and a mangrove wetland PES in Ambondrolava. Table 11.2 outlines the main features of these projects, all of which are still currently under development. The following section details the institutional factors and paths that have led to each individual water PES project being accepted and implemented (Fig. 11.2).

Table 11.2 Madagascar's water PES schemes

Project name	Lead NGO	Other partners	Place	Area (ha)	Number of water users	Number of catchment users	Transaction mode	Amount (€)[a,b]	Project type	Type of ES	Secondary ES
PES for drinking water supply to the town of Fianarantsoa	APMM	Fianarantsoa town council JIRAMA[c] WWF	Antarambiby	3,500	150,000	196 households	Direct contracts between farmers and water and electricity company JIRAMA, which pays compensation in monetary form. Non-monetary compensation in the form of seed	Transaction costs: €15,456; compensation: €77,037	Antarambiby catchment protection	Drinking water	Biodiversity

PES for drinking water supply to the city of Andapa	APMM	Andapa town council JIRAMA WWF	Sahamazava catchment	910	30,000	32 households	Monetary compensation paid by the "consultation platform" with money from users, JIRAMA and ANDAPA city council	Transaction costs: €16,419; compensation: €151,481	Sahamazava catchment protection	Drinking water	Biodiversity
PES for mangrove ecosystems in Toliara	Honko	WWF	Ambondrolava mangroves	600			Monetary compensation from tourist trips and sales of craft and livestock products; non-monetary through financing of school equipment, hospital renovation, sewing machine, etc.	Transaction costs: €8,174	(1) Secure management rights to communal mangroves (2) Training and compensation for mangrove restoration	Biodiversity	

(continued)

Table 11.2 (continued)

Project name	Lead NGO	Other partners	Place	Area (ha)	Number of water users	Number of catchment users	Transaction mode	Amount (€)[a,b]	Project type	Type of ES	Secondary ES
PES for protection of the catchment containing the Tolongoina micro-hydro plant	GRET		Tolongoina	630		35 households (c. 210 people)	Activities for rural development, in adequacy with farmers' needs and water services' requirements	To be defined	Protection of Andasy catchment with compensation for resident households	Water (hydro-electric power)	Forest protection

[a] Compensations are defined here as the actor's initial demand at the commencement of the bargaining process
[b] For transaction costs calculations, see Cahen-Fourot and Méral (2011)
[c] JIRAMA – JIro sy RAno MAlagasy (Madagascan public water and electricity company)

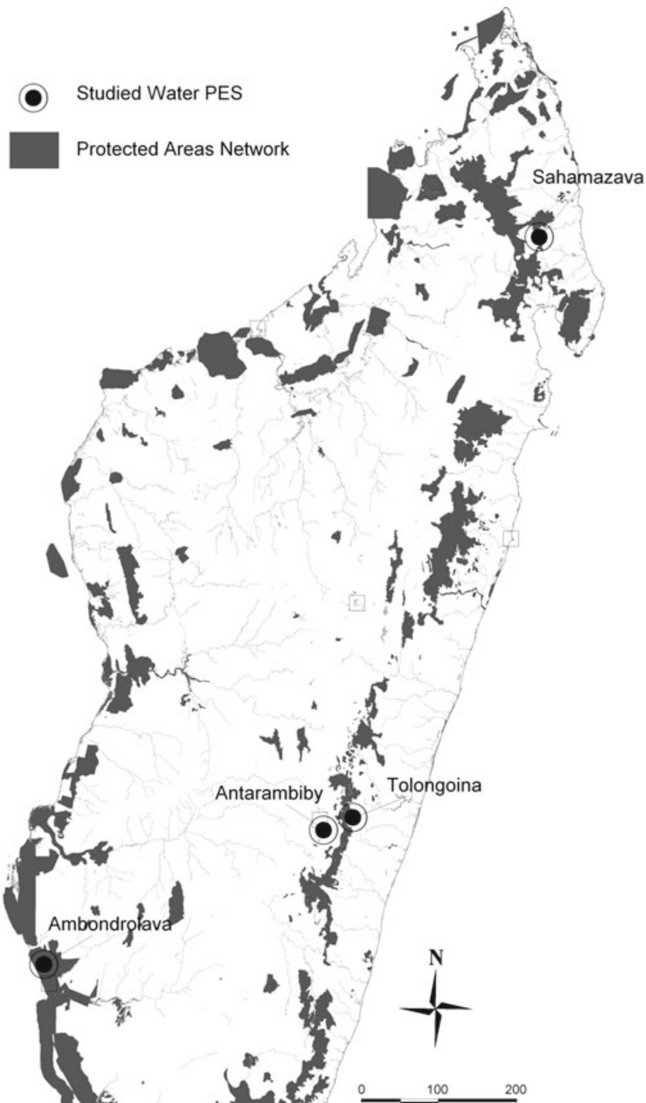

Fig. 11.2 The four water PES projects studied

11.3.1 Sahamazava Drinking Water PES Scheme: Between Reactivation of Past Projects and the Search for Compromise

Andapa is an urban commune with a population of about 30,000 in the north of Madagascar. Its drinking water comes from various springs from a catchment upstream of the town. Since 1975 when a national law was passed transferring water management from urban communes to the state-owned water and electricity company JIRAMA, Andapa's water has been supplied by JIRAMA that has implemented a water purification plan that includes tapping water from upstream springs that pass through the village of Sahamazava and shares the name of the associated catchment.

Andapa lies on flat plane with soils suitable for rice cropping. The community has received various kinds of support for agricultural development including funds from the European Development Fund (EDF) since the 1970s. In the 1990s pressure on natural resources increased due to population growth and irrigation problems downstream of the catchment. This led some farmers to begin off-season cropping and slash-and-burn farming on land further upstream.

Various steps were taken to address the resulting reduction of forest cover in upstream areas including environmental education and transfer of management. In 1993, the WWF initiated awareness-raising campaigns and supported the implementation of a ban on crop farming in sensitive areas. The WWF's initiatives were later taken over by the *Association Nationale des Aires Protégées* (ANGAP).

Since the early 2000s, Andapa's population and town council began to worry about their water supply. Flows from the springs were declining and increasingly irregular. The water quickly became muddy during the rains due to high levels of suspended particles.

When the national strategy for the development of mountain areas was being drawn up in 2002, the chairman of the Andapa local development committee contacted the NGO APMM (French acronym for the WMPA, World Mountain People Association) to help them take action towards sustainable management of the Sahamazava catchment. In May 2003, a discussion workshop organised by APMM brought together local elected officials, ministry representatives, the town council, state technical services, JIRAMA and other local stakeholders. An association was set up to protect the Sahamazava catchment as a result of this workshop. Further, solutions were identified to help protect the catchment such as banning access and reforestation activities. The idea of compensation was also raised, for example, compensating Sahamazava residents from a tax that water users would pay to JIRAMA. APMM was asked to provide advocacy support in the application of a new drinking water supply mechanism in the south of Andapa.

Coordination among stakeholders to implement the new drinking water supply mechanism was fairly straightforward because water sources and consumers are found within the same urban commune. Additionally, local awareness building activities for environmental problems and sustainable water management had already been undertaken by the WWF and ANGAP in the 1990s and then by APMM from 2003 on.

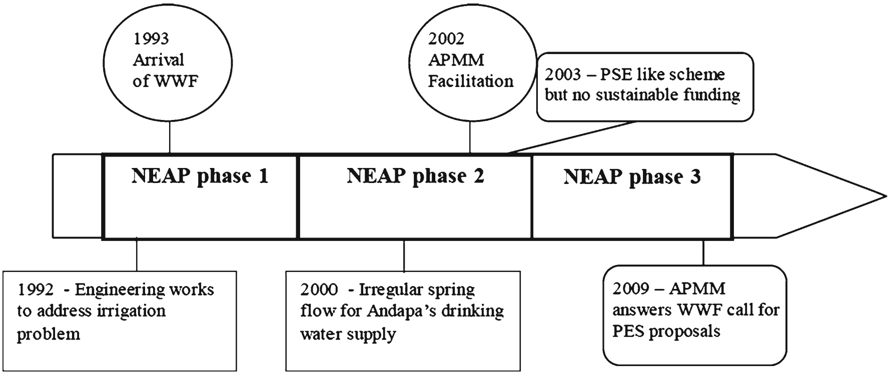

Fig. 11.3 Chronology of events leading to Sahamazava PES project

Some integrated water resource management (IWRM) actions that had already been implemented had much in common with the proposed PES scheme.

In 2009, the WWF published a call for proposals from potential partners to participate in a water PES programme. The APMM seized this opportunity to ensure sustainable funding for a PES scheme to protect the Sahamazava catchment. This PES project design and acceptance was greatly facilitated by a body of existing scientific data as well as the fact that the various stakeholders were aware of and understood environmental degradation issues.

The PES project was derived out of a favourable institutional dynamic, and a manageable number of informed stakeholders with a shared awareness of the environmental problems were associated with irrigation and drinking water. The PES scheme has profited from donor funding to set up the mechanisms that are intended to provide sustainable funding for catchment management. Further, the PES scheme coincides with the previous institutional path rather than requiring new institutions (Fig. 11.3).

11.3.2 Antarambiby Drinking Water PES Scheme: Between Conflict and Complexity

The second water PES scheme studied is that of Antarambiby. The problematic and type of stakeholders in this case are similar to those in the Sahamazava projects discussed previously. The 200-ha Antarambiby catchment has attracted particular attention since the early 2000s. JIRAMA supplies drinking water to the 170,000 residents of Madagascar's second largest town, Fianarantsoa, from a reservoir fed by the Antarambiby catchment. This catchment is suffering high levels of environmental degradation (soil, biodiversity, forest cover), thus requiring a sustainable management plan by stakeholders.

The Antarambiby water PES was preceded by a number of integrated water resource management schemes. In 2000, *Coalition* H_2O, a group of NGOs including WWF, FAMI and *Association Miray pour le Développement*, began an awareness building campaign and mobilisation of stakeholders around the issue of the water-environment link between Antarambiby and Fianarantsoa, not far from Ranomafana-Andrigitra corridor.

In 2001, USAID became interested in the activities of *Coalition* H_2O and proposed an action plan in collaboration with APMM, which had recently become active in the area. In 2002–2003, APMM Tambohitravo Malagasy launched various actions under an IWRM scheme. The aim was to introduce various tools such as a catchment development plan and a reforestation zone plan. In 2003, data acquisition and concerted planning for integrated water resource management were performed with support from French and Swiss universities.

At the same time, an *Organisme Public de Coopération Intercommunale* (OPCI – public body for inter-commune cooperation) was formed. Its purpose was to manage conflicts over water sharing between Fianarantsoa's urban water supply and competing farm irrigation needs. Planning was headed by the APMM with support from foreign universities conducting impact studies and building geographical databases. In 2004, a consultation forum of all pertinent stakeholders was set up. These structures are now directly involved in setting up the PES and had already began working to encourage JIRAMA to undertake reforestation activities and farmers to stop farming lowland areas near urban pumping stations.

The integrated management initiative was supported by various sources. In 2005, the catchment became a pilot catchment under the Hydrology for Environment, Life and Policy (HELP) programme, a UNESCO international hydrology programme whose goal is "networking with other water catchments around the world to improve the link between hydrology and society's water needs." In 2008, the AGIRE programme (a decentralised French-Malagasy cooperation project) took charge of this pilot catchment and the related technical, financial and organisational activities. However, the continuity of the IWRM work soon became endangered by a lack of resources and inadequate funding.

Similarly to the Sahamazava scheme, when the opportunity arose in 2009 to set up PES schemes in Madagascar, the APMM jumped at this opportunity to secure sustainable funding for water resource management in the Antarambiby catchment.

The analysis of the Antarambiby PES project is similar to that of the Sahamazava scheme. In both cases a void in financing was filled by the APMM with the help of international donors willing to set up PES mechanisms. In the Antarambiby scheme, however, a much larger number of less aware stakeholders is involved. The fact that distance between urban populations and upland farming areas is 25 km appears to be hampering the development of the PES (Fig. 11.4).[15]

[15] Unlike the previous project (integrated water resource management) which concerned only the actors upstream to the watershed, this PES project involves new actors being located downstream to the watershed, that is, in 25 km. This situation increases the coordination costs between all these actors.

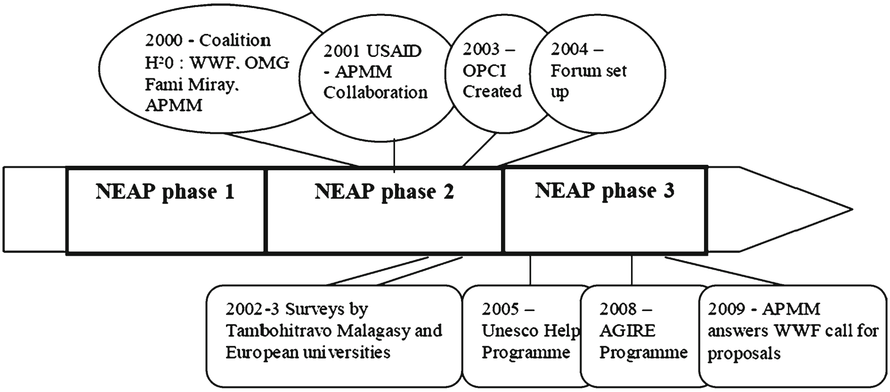

Fig. 11.4 Chronology of events leading to Antarambiby PES project

11.3.3 Tolongoina Water and Hydroelectricity PES Scheme: A Supporting PES Scheme

The third PES project concerns a rural area along Madagascar's eastern coast in the Vatovavy-Fitovinany region. It is located in the commune of Tolongoina (Tanala region) on the edge of the forest corridor between the Ranomafana and Andringitra national parks. The project's primary aim is to develop small autonomous hydroelectric networks in rural areas. The project's promoter is the GRET, a French NGO, financed through an EU programme.

At the PES scheme pilot site, there has been a conservation policy on paper since the late 1990s. Under the USAID ERI programme (described in Sect. 11.2.3), power generation using the water "produced by the forest" had been presented as a way to create value-added from forests. It combined conservationists' arguments with the prospect of practical development, thanks to the rural power supply component. It was from this perspective that GRET initiated a PES project that would be financially, technically and socially sustainable. For technical sustainability, effective steps are needed to ensure that the expected services continue to be supplied. That is, the scheme must provide a certain quantity and quality of water throughout the year to generate the required power. Financial sustainability means keeping costs low enough to maintain the micro-hydro plant's cost-effectiveness, knowing that return on investment is estimated at more than 20 years. With this, GRET believes that sustainable management of the catchment is more cost-effective than regular maintenance or replacement of the hydro plant's dams, channels and turbines. Social sustainability implies that catchment users must not feel unfairly treated by having to change their behaviour regarding water supply services and at the same time not receiving electricity.

The discussions around the PES scheme were driven by the concern to ensure continuing access to hydropower. GRET launched a feasibility study in 2009

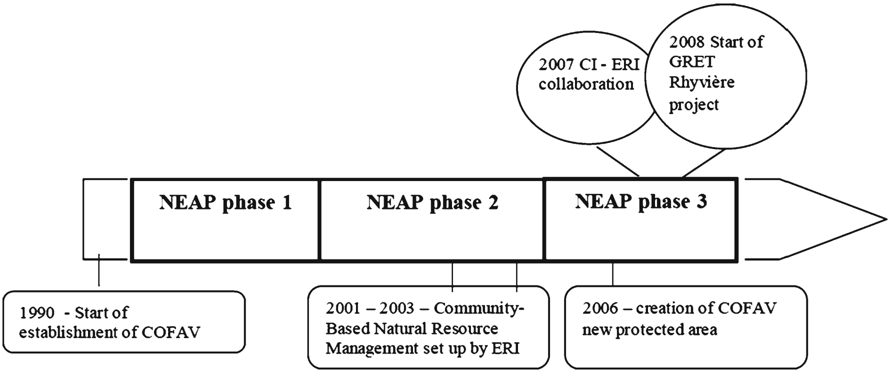

Fig. 11.5 Chronology of events leading to Tolongoina PES project

(Toillier 2009), mainly to test the social acceptance of such a scheme, to check that the water services were real and to lay the groundwork for a PES governance plan within the hydroelectric project.

The Tolongoina PES scheme thus emerged from three prior projects: the WWF-CI conservation corridors, the ERI ICDP project and the GRET rural electrification project. Each of these projects contained an embryo of the idea to create value-added from an environmental service. In the Tolongoina case, however, a rupture in rationale is observed. There is a shift from the rationale of upstream biodiversity conservation funded by international aid such as with the ERI to a local development rationale championed by GRET, involving the private sector and guided by downstream concerns. In fact, the funding requirements of the PES scheme have re-established a balance between the two sources of power with the resulting shift providing more space for discussion. Downstream interests and upstream interests are now more aware of and sympathetic to each other's needs.

The Tolongoina PES scheme arose from a favourable political context. Toillier (2009) has shown strong coherence between the regional context, the conservation antecedents and the aims of GRET. Moreover, Tolongoina seems to be an ideal pilot site due to the small size of the catchment (about 6 km^2), meaning that a change in land use will have a quick and observable impact on hydrologic characteristics (although these have not yet been quantified). Additionally, the existing environmental management system reduces the cost of mounting the project as it can rely on existing institutions and structures (communes and local communities).

One constraint with the Tolongoina PES is its reliance on the micro-hydro plant project component and resulting need for a new operator to be established. In the previously examined PES systems, a state enterprise was present and operating in the area, thus facilitating the incorporation of the PES mechanism. The new hydro plant operator will not be immediately financially sustainable, and the situation implies new relationships to be forged between users, administration and the new operator. For the reasons of financial sustainability and institutional legitimacy, there is a risk that payment for environmental services will be meagre in the early years (Fig. 11.5).

11.3.4 Ambondrolava Mangrove Wetland PES Scheme: Building on the Ruins of the Past

A final water PSE case study concerns a coastal area in southwest Madagascar. The marine and coastal environment is and has been the focus of intense interest for environmental action schemes in Madagascar. The big Toliara reef system and the coastal strip between Manombo and Anakao have attracted particular attention through activities such as marine protected areas, sustainable management of mangrove ecosystems, and value-added from biodiversity through tourism. The second phase of the NEAP (PE2), focusing mainly on decentralised, participatory natural resource management, produced various GELOSE contracts (*Gestion Locale Sécurisée* – secure local management) in the region of study. During the PE2, the coastal villages were targeted by projects for participatory planning, community-based natural resource management and value-added from biodiversity (ecotourism attracted by the lagoon). Protecting the mangrove ecosystems was among the primary preoccupations for such coastal communes as Ankilibe, Mangily and Belalanda where the Ambondrolava site is located. A number of small international NGOs working on scientific monitoring and social management of the lagoon (Blue Venture, Frontier, etc.) also contributed to this dynamic.

The development of the mangroves PES scheme, instigated by the NGO Honko and funded by WWF, blossomed out of this institutional dynamic of international NGOs and GELOSE contracts, which peaked in 2001–2002 at the end of the PE2.[16]

The introduction of the Belalanda GELOSE contract involved setting up a VOI (Vondron'Olona Ifotony or grassroots community), which was theoretically in charge of sustainable management of the mangrove ecosystems. For various reasons (weak local governance, mistrust between members and leaders, hostility among some members, etc.), this management transfer scheme was unable to function appropriately. The VOI did not properly fulfil its role and the mangrove ecosystems continued to deteriorate.

In 2007, UK-based NGO ReefDoctor, which was focusing on the coral reef system of southwest Madagascar, launched actions for marine resource protection and local development in the region. Among other initiatives, ReefDoctor helped establish product chains to improve local people's standards of living. In the case of honey, for example, ReefDoctor support included help with the packaging process, product quality improvement and marketing. Unfortunately, the local population was not able to consolidate these gains after the project closed.

The founders of the association Honko were involved in the ReefDoctor project at that time. They were therefore familiar with the region and its problems well and understood the importance of the mangrove ecosystems. It is under these circumstances, an unfavourable institutional context, after several past failures and

[16] Several GELOSE contracts (or "GELOSE marine" contracts) were launched in this period. Mangrove protection was a core issue for several of these contracts in coastal villages like Manombo (Fitsitiky), Ankilibe, Mangily and Belalanda.

with a problem with local social capital, that Honko began its mangrove protection activities in 2009. The village of Belalanda/Ambondrolava was selected because it is easily accessible and because local people expressed and exhibited strong motivation for action.

The failure of the earlier initiatives complicated the task of setting up the Honko programme. The former leaders of the grassroots groups were insistently keen to take advantage of the new project to entrench their power and control development strategies. Community members knew that one of the main reasons for the earlier failures was due precisely to these corrupt practices of local leaders. How does such a small community, then, identify and empower new leaders?

The local commune, also suspected of corruption during earlier projects, was not brought into the PES project in a major way. Indirectly, it helped to legalise the new grassroots group and grant it land for the project.

The climate of mistrust in the area was strengthened by a weak state authority to enforce laws on environmental matters. In one example, people from an inland community arrived to fish at night using *laro,* a non-selective plant-based poison that kills all aquatic species and is officially banned. The local council intervened to settle this issue but not without problems affecting PES implementation. As another example, before the PES began, a local man cutting mangrove wood threatened to kill the ex-village chief who challenged him on legal grounds.

The institutional path leading to the Honko water and biodiversity PES scheme looks like the least favourable of the four schemes studies here. Before a contract can even be signed, the minimal conditions for establishing a PES scheme do not seem to be fulfilled. Disregard of the law, suspicions of corruption and rent seeking strategies by former local leaders go a long way to explain earlier development intervention failures. Nothing in the current project seems to be generating a break with these old practices. As Mahoney (2000, p. 517) points out, in a configuration where power dynamics constitute the raison d'être that generates and reinforces institutions, there can be no break without a "weakening of elites and strengthening of subordinate groups" (Fig. 11.6).

11.4 Link to Theoretical Aspects

Several authors have demonstrated the significant role institutions play in the establishment of PES (Muradian et al. 2010; Pattanayak et al. 2010). Thelen's institutional change typology theory (2003) shows that there are numerous and diverse approaches to study, understand and validate institutional changes. Applying Thelen's theory specifically to the field of PES, we can distinguish different analytical frameworks. One framework is ahistorical where PES are observed through three different perspectives: (1) through a functionalist or *utilitarian perspective* (PES are chosen for their efficiency in resolving collective-action problems) where promoters of PES propose activities as an alternative to past direct payment practices such as integrated conservation and development projects (ICDP) (Simpson and Sedjo

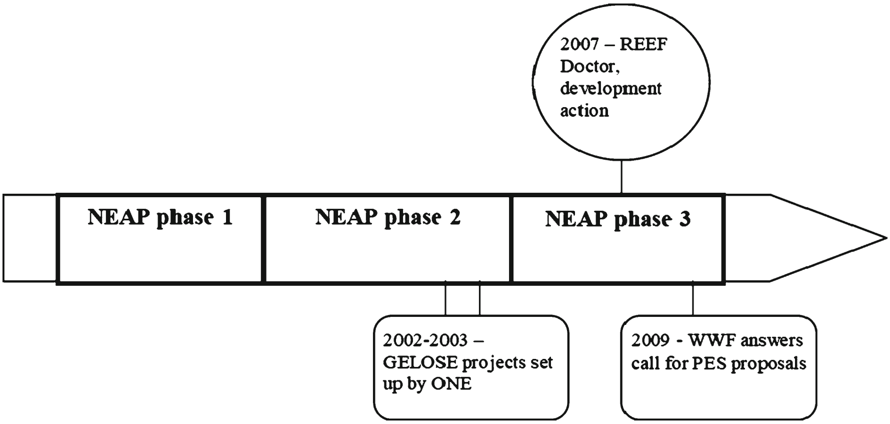

Fig. 11.6 Chronology of events leading to Ambondrolava PES project

1996; Ferraro and Kiss 2002, etc.), (2) through a *culturalist perspective* (PES are chosen due primarily to a shared world/shared responsibility belief). Here authors often link PES with a general framework for ecosystem services (Kumar and Muradian 2009; TEEB 2010; Pesche et al. 2012), and (3) through a *power relations perspective* (PES are chosen by actors who dominate decision-making processes and who thus promote specific interests). Research and subsequent literature specifically on this last perspective is limited, even though many authors have revealed interest and importance in it (Muradian et al. 2010; Vatn 2010).

A second analytical framework for PES is historical and inspired by a path dependency approach. The approach is twofold: (1) to analyse institutional innovations as a direct/indirect result of a series of past events and decisions, and (2) to establish and evaluate future PES trajectories through an *ex ante* perspective to help address effective issues and ultimately inform policy. In spite of numerous references to this path dependency approach, few articles use this framework to perform an in-depth analysis.

The Malagasy experience illustrated in this article advances the use of the path dependency framework. The results indicate that the Mahoney approach, marked by technological innovations, is pertinent at a national level where locking-in mechanisms in conservation policy are clearly observed. Local trajectories within such policies, however, are not irreversible. This is not because the PES presented here are still not in full implementation but instead because they are validated as complement to past actions by continuing institutions. Institutional innovation is therefore incremental and a result of an adaptation process by actors at the local level. This process falls in line with the sedimentation process and institutional conversion theory defended by Thelen.

Our analysis of Malagasy water PES case studies hinges on several key elements within the path dependency framework. In this, we distinguish two types of processes: (1) those steering national conservation policy to *choose* to implement PES instruments, and (2) the processes involved in the actual institutional changes that occur as a result. Concerning the decision to implement PES, we find that historical influences and actor's strategies all play vital roles in this process. Specifically, past decision and situations (historical factors) allow us to define explicitly the processes that have led to selecting and implementing PES. For example, history and experience of collective action and activity implementation at a local level play a predominate role in not only the nature and effectiveness of PES but also the initial decision to undertake such activities.

Concerning institutional changes that occur when incorporating PES into conservation policy (effects of PES implementation on institutions), processes such as lock-in, layering and self-enforcement are identified to help explain the nature and speed of such changes or reactions. Specifically we looked to see if PES implementation contributes to locking effects and if new institutions are created or if existing intuitions are replaced or brought to an end. The analysis also consists of defining the initial conditions of new institutions and demonstrating how these adapt to incorporate PES into their mandates. Globally, this aspect of the analysis can be defined as the evaluation of the incremental or racial nature of institutional change (Table 11.3).

11.5 Conclusion

Institutional analyses of PES, and conservation policies in general, represent an important field of investigation. Analytical frameworks are dynamic and adaptable, thus resulting in several distinguished works by different authors, proposing different types of analyses (Vatn 2010; Daniels et al. 2010; Muradian et al. 2010).

This chapter builds upon previous work by exploring more deeply the path dependency approach, which is underrepresented in literature. Applying the Malagasy experience reinforces the applicability and pertinence of this analytical framework. The historical dimension is proven essentially in understanding the emergence of PES. Rather than simply retracing history, we identify and examine initial conditions and the evolution of institutional processes to judge the probable effectiveness and efficiency of PES.

At the national level in Madagascar, the historical path identified has shaped a largely favourable situation to implement PES schemes. Beginning as early as 2001, the policy framework (i.e. the environmental action plan) has sought sustainable funding methods to ensure conservation actions would last beyond the lifespan of the plan. PES such as REDD-plus mechanisms and the creation of the *Fondation pour la Biodiversité* provide ways to raise funds that supplement official development aid (Méral et al. 2011). Institutional conditions are therefore considered favourable for the emergence of such conservation tools.

Observed institutional conditions have also led to PES schemes connected strictly with protected areas. The Durban declaration of 2003 committed Madagascar to extending its network of protected areas, corroborating the country's chosen path towards sustainable funding of protected areas. Using a term from Mahoney (year),

Table 11.3 Comparison frame of the four case studies and the national policy regarding path dependency variables

		National level	Sahamazava PES	Antarambiby PES	Tolongoina PES	Ambondrolava PES
Initial conditions	Knowledge, problem or service identification	Strong. Since colonial period, Malagasy biological diversity is known by scientists	Strong since the beginning of 2000s	Strong since 2003	Strong since the end of 1990s "Water tower" effect	Strong since the beginning of 2000s through international NGOs (coral reefs)
	Oldness of awareness campaign	Strong. Since 1990 (environmental plan)	Strong. 1993 with WWF and ANGAP, then through APMM	Strong. 2000 *(Coalition H_2O)*, 2003 (USAID) … But recurrent difficulties to link providers and beneficiaries because of the distance	Strong. Programme ERI (USAID), Corridor WWF-Conservation International	Strong. Until the end of the PE2 in 2002. Moderated between 2002 and 2009
	Oldness of actors	Strong. Since 1990. Donors, NGOs …	Strong. Workshop in 2003 with the implication of previous actors. Decision to improve sustainable management of the Sahamalaza watershed	Strong. Same actors since 2000	Moderated	Strong but very conflicting (corruption, failures of former measures, etc.)

(continued)

Table 11.3 (continued)

		National level	Sahamazava PES	Antarambiby PES	Tolongoina PES	Ambondrolava PES
Institutional process	Critical junctures	Durban Vision (2003)	None. Step by step evolution. Decision to implement a PES in 2009 (WWF proposition)		GRET programme in 2009	None
	Inferred trajectories	Power disappearance or reduction of past organisations Apparition of new actors (firms, MBI networks); creation of new protected areas. Lock-in process and self-reinforcement by conservation NGOs – radical innovation	Layering process (integrated management at first, PES now, but carried by the same actors) PES are considered as tool to improve the previous actions Incremental innovation – no lock-in process for the moment		Apparition of a new actor: GRET change from conservation focus to development with a micro electric PES project. Radical Innovation	PES project carried by a new actor (association Honko) but institutional process very limited due to the initial conditions

the Durban declaration of 2003 can be considered a "critical moment." At this moment other choices could have been made, such as sustainable funding of community-based management, which would have potentially changed the institutional environment and resulting adoption and implementation of PES within it.

Analysing the history of this policy and the resulting institutions indicates that the major international conservation NGOs and donors largely determined the direction taken by Madagascar. The choices made at critical moments, like that in Durban, were strongly influenced by political and financial power plays, more so than strictly lock-in effects. Although the idea of sustainable funding for environmental policy was already present at the start of the first phase of NEAP in 1990, the application of actual mechanisms through PES and other market-based mechanisms only surfaced in 2003 due to the favourable international context. International players, donors, conservation NGOs, experts and other stakeholders developed discourse favourable to market-based instruments that was then imparted upon cognitive resource and accepted by the national players in Madagascar. Dependence on the international path, strengthened by the logic of globalisation, has also been stressed in other contexts (Djelic and Quack 2007).

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Chapter 12
The Governance of Costa Rica's Programme of Payments for Environmental Services: A Stakeholder's Perspective

Jean-Francois Le Coq, Géraldine Froger, Thomas Legrand, Denis Pesche, and Fernando Saenz-Segura

12.1 Introduction

Over the past decade, "Payments for Environmental Services" (PES) have received a great deal of attention as a natural resource management approach (Landell-Mills and Porras 2002; Corbera et al. 2007; Engel et al. 2008; Wunder et al. 2008; Pattanayak et al. 2010). Wunder (2005, 2007) defines PES as voluntary transactions where a well-defined Environmental Service (ES) (or a land use likely to secure that service) is "bought by a minimum of one ES buyer from a minimum of one ES provider if and only if the ES provider secures ES provision during a determined time (conditionality)." Pure PES schemes fulfilling all the criteria of Wunder's definition may not always be possible or even preferable (Wunder 2005; Corbera et al. 2007). More recently, scholars have analysed the institutional nature of PES, underlining the importance of the institutional and social context in which it takes place (Muradian et al. 2010; Sommerville et al. 2009; Vatn 2010). They usually

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consider PES as a social construction, reflecting a certain distribution of power among stakeholders, while often emphasizing the need for legitimacy as an important driver of its design and evolution (Corbera et al. 2007; Pascual et al. 2010).

As a pioneer programme using the PES notion, the Costa Rican Payment for Environmental Services Program (PESP) has been analysed as a very promising and innovating instrument for conservation and has been considered a reference for PES schemes. Since its launch in 1997 to address deforestation, the programme has invested more than 200 million cumulative dollars and contracted over 700,000 ha of forest,[1] representing 13% of the national territory. Many scholars have discussed the PESP efficiency (Daniels et al. 2010; Legrand et al. 2011; Sánchez-Azofeifa et al. 2007; Zbinden and Lee 2005) and described it as an innovative market-based instrument for conservation (Pagiola 2008; Rojas and Aylward 2003; Pagiola et al. 2002). Nevertheless, the analyses of the PESP dynamics of evolution and governance have been poorly documented.

The objective of this chapter is to analyse the genesis and evolution of the PESP evidencing the influence of the different stakeholders on its functioning.

To develop this analysis, we relied on an extensive analysis of existing PESP literature, reviewing internal reports, decrees and procedure manuals issued by the National Fund for Forest Financing ("Fondo Nacional de Financiamiento Forestal" – FONAFIFO), which is in charge of PESP implementation. We also conducted semi-structured interviews of more than 50 stakeholders directly involved in the genesis, evolution and management of the PESP as well as stakeholder representatives involved in the management of forest plots. These stakeholders were invited to present their own trajectory and describe the roles they played in the emergence and evolution of the PESP, as well as their perception of the PESP since its beginning. They were asked to explain why and how decisions affecting the PESP arose: who were the main stakeholders involved and how they thought (visions, motivations, …) and acted (strategies, argumentations, resources, …).

In this chapter, after a presentation of theoretical framework, we describe the basic features of PESP and its evolution since 1997 regarding funding, payment and management systems. Then we further analyse the PESP evolution in the light of power balance evolution of stakeholders' groups involved in forest issues.

12.2 PESP: A Theoretical and Analytical Framework

Following Corbera et al. (2009), PES consists of transferring economic resources from ecosystem services providers to consumers so that the former benefit economically while the latter receive the right to use the resources provided by the service in question.

[1] These references correspond to data available on the FONAFIFO website: http://www.fonafifo.go.cr/paginas_espanol/servicios_ambientales/sa_estadisticas.htm

Wunder et al. (2008) classified the Costa Rican PESP as a government-financed programme subject to side objectives, however, did not analyse the diversity of these objectives. Following Muradian et al. (2010), we considered PES as a complex and multi-goal policy instrument, subject to social embeddedness and power relations.

In this chapter, by analysing the genesis and evolution of PESP from a stakeholder's perspective, we understand why and how multi-objectives are combined in the PES instrument.

Assuming a Northian perspective, PESP can be considered as an institution (North 1990), as such its evolution depends on interactions with organizations. Following Corbera et al. (2009), we adopt an institutional framework and concentrate on the institutional design of PESP,[2] setting the following specific questions: Which actors shape the rule-design process and how are their interests represented in the final rules? Why and how design rules change over time?

To understand the conditions of institutional changes, we mobilized sociological policy approaches that consider public policies actors and their interactions (Hassenteufel 2008). Considering public policies as a collective action, policy changes are interpreted as the results of interactions of actors in an evolving context. The actors (in opposition to rational approaches) developed strategies according to their policy action resources (juridical, material, knowledge, political, social) and their cognitive characteristics. We considered the belief system of the actors (Sabatier and Jenkins-Smith 1993) as the cognitive characteristics and how the beliefs of the actors of the same coalition impacted public policy orientation.

Thus, we identified the actors[3] involved in the PESP decision-making process and analysed the PESP implementation rule. We identified the different changes that occurred during implementation of the PESP since its inception and analysed the context in which the changes took place. Through direct interview, we identified each stakeholder's perception of the forest problem and belief system, PESP rules and orientations, as well as their interests, resources and strategies towards PESP orientation.

12.3 PESP Basic Features and Their Evolution

Aiming at addressing the deforestation problem, the PESP was implemented through the 4th Costa Rican forestry law (#7575) adopted in April 1996. The PESP core principle is to provide payment to private forest landowners for the Environmental

[2] The analytical framework proposed by Corbera et al. (2009) also analyses institutional performance and institutional interplay that will not be discussed in this chapter (for an analysis of institutional performance of PESP, see Legrand et al. 2011). We also will not discuss the institutional nature of PESP (Pagiola 2008), nor the efficiency of PES compared to other instruments, nor their scope of PES efficiency (Wunder et al. 2008; Kemkes et al. 2010; Farley and Costanza 2010).

[3] The actors can be individuals or organizations.

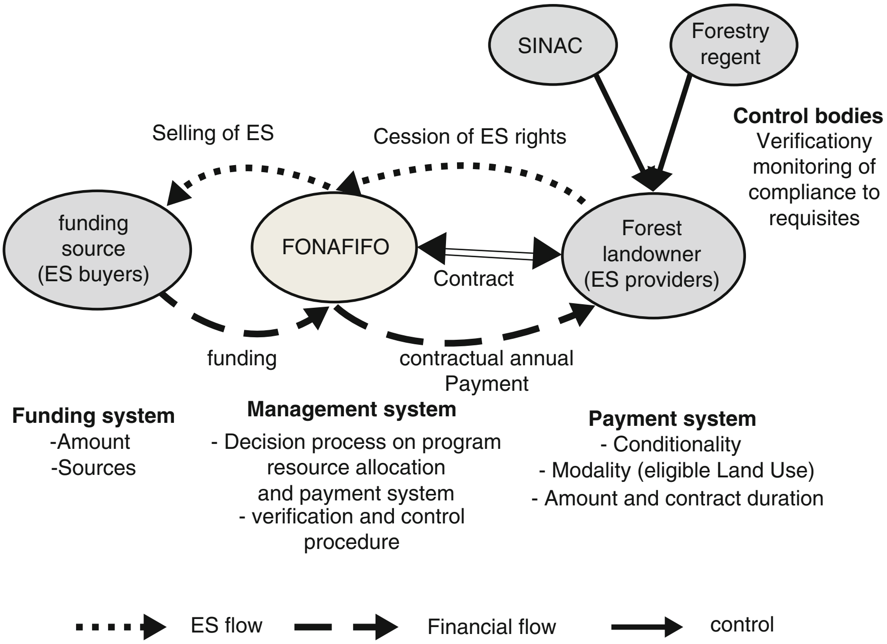

Fig. 12.1 Basic principles of PESP functioning (Source: Authors derived from interviews with FONAFIFO

Services (ES) generated by their forests (Fig. 12.1). Forest landowners cede their rights to their forests' ES to a management entity (FONAFIFO) that sells them to ES users. A formal contract is established between the management entity (FONAFIFO) and forest landowner to set the transaction. This contract is contingent on the existence of a forest management plan with which the forest owners are required to comply with. A forestry regent[4] contracted by the forest owners and who is in charge of monitoring PES contract compliance issues this plan. Forestry regents also act as third party controlling PES contracts between forest landowners and FONAFIFO. The National System of Conservation Areas (SINAC),[5] a public institution, is in charge of landowner compliance with forestry law.

[4] The forestry regent ("regente forestal") is a formal body created by Forestry Law 7575. They are licensed forestry engineers who have the legal power ("fe publica") to authenticate the management plan of private landowners. Forestry regents are accredited by the Board of Agronomy Engineering ("Colegio de Ingenieros Agrónom*os*") that are in charge of monitoring and control of forestry regent activities.

[5] According to the Biodiversity Law (1998), SINAC ("Sistema National de Areas de Conservacion") is an institutional system of decentralized and participative management that integrates the Ministry of the Environment skills in terms of forestry, wildlife and wildlife-protected areas in order to dictate policies, plan and implement processes aimed at sustainability of Costa Rican natural resources management.

Year[a]	Forest protection[b]	in forest protection area	for hydraulic ressources	in "con-servation gap"	Re-forestation[b]	Mana-gement[b]	Established plantation[b]	Agroforestry System[c]	Natural regeneration[b] with productive potencial	Natural regeneration[b] on pasture	Natural regeneration[b] on pasture (kyoto land)
1997	215				517	346					
1998	234				600	366	234				
1999	211				541	331	211				
2000	215				551	-	-				
2001	224				572	349	-				
2002	228				584	356	-				
2003	225				575	-	225	0,82			
2004	224				572	-	-	0,75			
2005	320				816	-	-	1,3			
2006	320				816	-	-	1,3	816	205	-
2007	320				816	-	-	1,3	816	205	-
2008	320				816	-	-	1,3	816	205	-
2009	320	320	400	375	980	-	-	1,3	205	205	320
2010	320	320	400	375	980	250	-	1,3	205	205	320

Fig. 12.2 Evolution of PES modalities and level of contractual payment per modalities (NB: [a] from 1997 to 2004 payment amount defined in Colones, from 2005 to 2010 payment amounts are defined in US$; [b] total amount of payment for the contract duration in US$ Ha^{-1}; [c] total amount of payment for the contract duration in US$ tree^{-1}) (Source: FONAFIFO)

Thus, we can define three main PESP features (Fig. 12.1): (1) its funding system characterized by sources and amounts; (2) its payment system including the type of eligible modalities, the levels of payments and the prioritization of payments; and (3) its management system, which consists of strategic management of the programme (definition of the programme's rules and strategy) and operations (contracting, disbursement, monitoring and control procedures).

12.3.1 The Payment System

The payment system is based on the recognition of law # 7575, the provision of ES by *forest and forest plantations,* and the explicit definition of four ES: greenhouse gas emissions mitigation, water protection, biodiversity protection and scenic beauty.

When the PESP began in 1997, three activities were eligible to receive payments in line with existing forest incentives schemes[6]: conservation of existing forest (PES-Protection), reforestation (PES-Reforestation) and sustainable management of forests (PES-Management) (Fig. 12.2). For each modality, a payment level

[6] Costa Rica has developed economic incentives for forestry since the 1970s. Before PESP, four main economic incentives were in place the Forest Payment Certificate (CAF) created in 1986, the Advanced Forest Payment Certificate (CAFA) created in 1988 to compensate landowner reforestation investments, the Forest Payment Certificate for Management (CAFMA) created in 1993 to encourage sustainable practices of wood extraction and the Forest Protection Certificate (CPB) created in 1995 to encourage protection of existing forests (Daniels et al. 2010; FONAFIFO 2005).

per hectare of land was defined to correspond to the minimum acceptable by the landowner to compensate the cost of reforestation (PES-Reforestation) or sustainable management practices (PES-Management) or to cover the minimal cost of opportunity of forest conservation (PES-Protection).[7]

Since 1997, the payment system has evolved regarding eligible modalities, payment amounts by modality, payment targeting and payment differentiation (Fig. 12.2).

Eligible modalities evolution: Three main changes in the PESP eligible modalities occurred over the last 15 years: (1) the suppression from 2002 to 2009 of PES modality for forest management; (2) the inclusion in 2003 of the PES modality for agroforestry systems, which consists of payment for trees planted in agroforestry systems; and (3) the inclusion in 2006 of a new PES modality for natural regeneration, which consists of paying landowners as an incentive to allow for regeneration of forest on former pastures.

Evolution of importance between modalities: By far the main PES modality, in terms of contracted area, is PES-Protection which accounts for 89% of the total contracted PES area from 1997 to 2010, whereas PES-Reforestation, PES-Management and PES-Natural Regeneration represented, respectively, 6, 4 and 1% of the total contractual area in the same period. Since its beginning, the distribution of contract areas among modalities has evolved. In the early period of PESP implementation (1997–2001), PES-Protection modality accounted for 84% of the total area, whereas the PES-Management and PES-Reforestation accounted for, respectively, 9 and 7% of the total area. Between 2002 and 2005, PES-Protection held a higher percentage, with 94% of the total contracted area. Since 2006, the situation changed once again with 89, 9 and 2% of total contracted area, respectively, for PES-Protection, PES-Reforestation and PES-Regeneration.

Payment levels evolution: The payment evolution has been marked by a substantial increase in the level of payment and the dollarization of the payments, both occurring in 2005 (Fig. 12.2). The levels of payment between 2004 and 2005, respectively, went from 224 to 320 US$ ha^{-1} for the forest protection modality, from 572 to 816 US$ ha^{-1} for the reforestation modality and from 0.75 to 1.3 US$/tree for the regeneration modality. The reforestation modality was made more profitable by raising the level of payment to 980 US$ ha^{-1} in 2009. Between 2008 and 2009, the level of payment remained the same; however, the contract duration was reduced from 10 to 5 years.

Access conditions: During the first years of the programme, the access conditions were similar to those defined in the previous existing instruments. Applicants were asked to present a management plan and to have formal land property rights. Contractual obligations and payments were effective for the PESP beneficiary once

[7] The level of payment for PES-Protection in 1997 was more or less the opportunity cost of extensive cattle raising, which was one of the major alternatives to forestry from the 1960s to the 1990s (Legrand et al. 2010); it was also a mode level of the different evaluations of potential annual costs for the four services and the local market cost of renting a hectare of pasture (Castro et al. 2000).

his land tenure status was confirmed as the legal owner and land title bearer with the National Registry Office. In the case that the land is sold or transferred, contractual conditions will apply to future owners. Since the inception of the programme implementation, certain measures have been taken to ease small landholder participation. In order to lower the transaction costs assumed by small landowners to contract a forestry regent and to fulfil the application forms, a collective contract system was created in 1998 that enabled small landholders to apply for the PESP together. This practice was abandoned in 2002 because of the additional delays created by the heterogeneity of farmers' situations.[8] Nevertheless, in 2010, a quota was attributed to local forestry organizations that facilitated preparation of small landholders' applications. Furthermore, a specific procedure has been put in place in 2002 to enable small landowners without formal land title programme access; however, the specific requirements for landholder without formal land title are often difficult to comply with.

Payment targeting: At the beginning of the programme, the basic principle for application selection was "first in time, first in rights": the PES demand was analysed according to the moment and order of reception at the FONAFIFO office. PESP applications were prioritized to target the most important lands for ES provision (mainly water and biodiversity services). A system of prioritization of demand was put in place in 2002 and progressively strengthened. In 2004, a social criterion was also aggregated. It gave priority to the forest owners located in districts with a low development index. In 2011, a scoring system had been put in practice. It took into account applicant locations, gave priority in case of the PES-Protection, to lands in "conservation blanks," inside national parks and biologic corridors, key water protection areas, low development index districts and indigenous territories. Priority was also given to lands previously under PESP contract or which have already submitted an application and to applications for areas less than 50 ha.

Payment differentiation: At the beginning of the programme, within each modality, the level of payment was equal whatever the location of the land and the ES provision of the land. Since 2009, a differentiation of payment level was initiated for PES-Protection to take into account differences in ES provision (Fig. 12.2). Thus, protection of forests in key areas for hydrological services provision receives an additional payment of 80 US$ ha^{-1} (over 5 years), whereas land located in critical biodiversity zones, outside parks or in insisting ecological corridors receives 55 US$ ha^{-1} additional payment (over 5 years). Furthermore, for the natural regeneration PES modality, the lands eligible for funding through the Kyoto protocol can receive an additional payment of 115 US$ ha^{-1} compared to classic natural regeneration contracts.[9]

[8] As the application was collective, the payment was done only when all the forests owners of the groups were complying with all the requisites. Because some farmers were not complying with some requisites, the other farmers within the collective application were not receiving payments even if they individually complied.

[9] In 2011, this trend towards differentiation of the level of payment was strengthened with the creation of a new PES-Reforestation category, PES-Reforestation with wood species in danger of extinction, for which a higher payment was proposed (1,470 US$ ha^{-1} instead of 980 US$ ha^{-1} for normal PES-Reforestation modality).

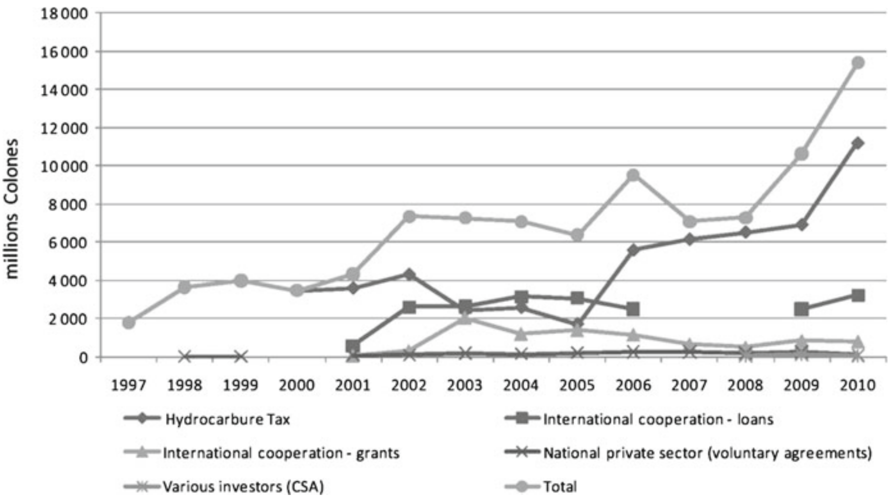

Fig. 12.3 Amount and sources of funding of PESP (in million of colones, 1 euro=684 colones in December 2010) (Source: FONAFIFO)

12.3.2 The Funding System

As a key initial element for PESP, law #7575 defines that a third of the existing hydrocarbon tax will be utilized to pay for private forest owners' ES. This source of funding was justified by the "polluter-pays" principle, i.e. fuel consumers emitting CO_2 compensate for damage by contributing to the payment of carbon sequestration by forests. Thus, in 1997, PESP began operations with the hydrocarbon tax as its only source of funding, with the expectation of raising additional funds from ES users, especially those from carbon sequestration through emerging carbon markets.

As for many programmes, consolidation of funding resources has been a major issue for PESP sustainability. The evolution of programme budgets and funding sources since 1997 illustrates that programme funding sources have been increasing and diversifying overtime (Fig. 12.3).

The fuel tax was the sole funding source of the programme during the programme's first implementation years, and therefore, the first issue at hand was to ensure payment of the fuel tax to the programme. In 1997 and 1998, the Ministry of Finance resisted its obligation to pay FONAFIFO one third of the amount raised by the fuel tax (Government of Costa Rica 1998). In 2001, a tax simplification law created a unique hydrocarbon tax of which 3.5% was clearly earmarked for FONAFIFO to fund PESP, which led to a consolidation of PESP funding. Although the negotiated level had been set at a lower level of the amount stated in the 4th Forest Law, the funds raised from the fuel tax increased when the fuel tax scheme changed. With the increase in fuel prices over the last years, the fuel tax represents the main source of programme funding.

The second funding issue was to increase and diversify the funding source, based on the principle that ES users should pay ES providers. Funding source diversification has been developed and includes payments for additional ES outside of carbon and different funding sources other than the national public tax.

Although it was supposed to become one of the main funding sources of the programme, the global carbon market resulted in disappointing funding until now. After the Norway government signed a two-million-US$ contract pioneer carbon deal in 1997, no other funding was raised through the carbon market because the Kyoto Clean Development Mechanism protocol was quite restrictive and did not include avoided deforestation, a main objective of the PESP. Therefore, national and international fund-raising targeting the private sector has been promoted, which has led to contracts with national hydropower and water companies, and in 1998–1999 contracts with national breweries. Furthermore, in 2001, "ES certificates" were created by FONAFIFO to attract voluntary donations from private investors willing to invest in ES generation (carbon, water or biodiversity). While the number of contracts has increased, funding from private enterprises is still limited (Fig. 12.3).

To maintain and extend PESP funding sources, additional resources were collected from international donors. Since 2001, the international cooperation has contributed significantly to PESP funding. The World Bank and the Global Environment Facility (GEF) contributed a 40-million-US$ loan and a five-million-US$ grant, respectively, through the Ecomarkets project. The loan was a substitute to the government's obligation to channel part of the hydrocarbon tax to the PESP and did not bring additional resources to the programme. In 2008, a new project called the "Mainstreaming Market-Based Instruments for Environmental Management" (commonly called Ecomarkets II) was implemented to extend programme funding sources. It consisted of a 40-million-US$ loan from the World Bank and a ten-million-US$ grant from GEF. A major part of the grant has been channelled through the "Biodiversity Conservation Trust Fund," which was created in 2006 to "serve as the repository of other grants, and of income from sales of conservation certificates in the voluntary market" (Pagiola 2008). Furthermore, in 2003 a ten-million-Euro funding agreement was reached with the German corporation KfW.

In 2006, water use legislation was passed which included a transfer of one fourth of the water tax to PESP. Unfortunately, while the additional funding resource was justified by the contribution of forests to water catchment and infiltration, the first funding transfer was delayed until 2010. However, this funding source has the potential to contribute six million US$ to the project once the water tax collection is fully effective (Madrigal Balestero 2009).

12.3.3 The Management System

The third key element of law #7575 was assigning an organization to manage the PESP. FONAFIFO, a public non-governmental trust fund, was placed in charge of fundraising and management of PESP. Law #7575 sets FONAFIFO's board of directors composition, which is in charge of identifying the main strategic options and validating the financial management. This board is composed of five members: three public sector representatives including one representative of the Ministry of Environment (MINAET), one representative of the Ministry of Agriculture (MAG) and one representative of the national banking system, and two private sector representatives nominated by the National Forestry office ("Oficina Nacional Forestal" – ONF) including one small/medium forestry producers representative and one industrial sector representative.

PESP implementation is regulated by two primary legal instruments that are updated annually: (1) an annual decree signed by the Ministry of Environment, which defines the eligible PES modalities and the total budget allocation for each of them, and (2) a procedure manual that defines the PES access conditions, requisites, priority criteria and administrative rules. These documents are revised annually by FONAFIFO's executive management and are submitted for comment to three main actors: SINAC (the forestry public administration representative), ONF (the forestry private sector representative) and the Board of Agronomy Engineering that supervises the forestry regents activities. After consultation, the decree and procedure manual are approved by FONAFIFO's board and signed by the Minister of Environment.

Since the beginning, PESP management has explored several options regarding (1) operational structure (distribution of responsibility and regulation) and (2) application and control procedures. The PESP operational structure experienced two major changes since 1997. In 2003, PESP administrative operational management was changed from a shared responsibility between SINAC and FONAFIFO to the sole responsibility of FONAFIFO. Prior to 2003, SINAC was in charge of receiving, analysing and checking the compliance of applications in accordance to the procedure manual and in some cases prioritizing applications. FONAFIFO was in charge of the final decision and payment execution. In 2002, the management responsibility between SINAC and FONAFIFO was reformed. FONAFIFO took control of all administrative procedures from the reception of application forms to payment execution, while SINAC focused on defining global prioritization rules and controlling PESP beneficiary compliance to the forestry law. This transfer of responsibility led FONAFIFO to develop its own regional office in 2003[10] in order to be able to receive forms locally. FONAFIFO's operating costs increased with the new responsibilities and included 15 new employees in 1998 and 35 new employees in 2005; however, FONAFIFO did not receive additional financial resources from the state budget to compensate for the increase in costs. In 2008, FONAFIFO was required to change its administrative management from private organization to public organization regulation. Since its creation as a trust fund, FONAFIFO was administrated as a private organization, but in 2008, following a decision issued by the general control

[10] In 2003, FONAFIFO created seven regional offices. Two additional offices were created in 2004 and 2005 by splitting existing offices to facilitate management. Today, FONAFIFO has nine regional offices throughout Costa Rica. To reduce costs, offices are generally located in SINAC regional buildings.

body of the republic ("Contraloria General de la Republica" – CGR), FONAFIFO was mandated to comply with the legal obligations of public sector organizations (especially regarding internal control and employee status). This mandate led to further increases in administrative programme costs and in staff numbers from 52 to 100 employees between 2008 and 2010.

Since its creation, the administrative procedure for application instruction and management and payment control has been simplified and optimized for efficiency. In order to reduce application costs for PES-Protection, the management plan, as well as administrative requisite processes prior to application control, has been simplified. Moreover, contract control has been optimized through a geographic information system in order to facilitate continuous monitoring and to control the effectiveness of the programme on land uses. Finally, the payment delivery procedure to the beneficiary has evolved from a time-consuming certificate (value cheques) system, to a bank cheque system in 2002, to a direct bank transfer to the landowner's bank account in 2005. The bank transfer system has significantly reduced both time and cost to FONAFIFO and landowners.

The analysis of the PESP evolution shows an overall programme consolidation especially regarding funding sources and management practices but also adjustments in eligible modalities, targeting and payment differentiation. In the following section, we will analyse the reasons behind these evolutions from a stakeholder's perspective.

12.4 PESP Governance Under Stakeholders' Influences

12.4.1 The Forest Stakeholders and PESP Decision Process

Numerous actors are involved in PESP governance and can influence the PESP decision process (Fig. 12.4). The first actors are those in charge of PESP management such as FONAFIFO (including civil servants at national and regional offices), SINAC in charge of natural resources (including forest), management and control, the forestry regent represented by the Board of Agronomy Engineering (BAE) and local forestry organizations that promote and facilitate payment access to small forest owners. The second actors are those represented in FONAFIFO's board of directors. Public sector representatives occupy three of the five positions. The Ministry of Environment's representative usually stands as the president of FONAFIFO's board, while the Ministry of Agriculture and the banking sector both maintain a representative on the board. The private forestry sector maintains two representatives on FONAFIFO's board: (1) the large forestry companies (often also a wood seller and manufacturer) representative that is currently represented by the Costa Rican Forestry Chamber (*Camara Costarricense Forestal* – CCF) and (2) the small and medium forest landowners, which are generally members of local forestry organizations and are represented by the National Assembly of Forestry Peasants

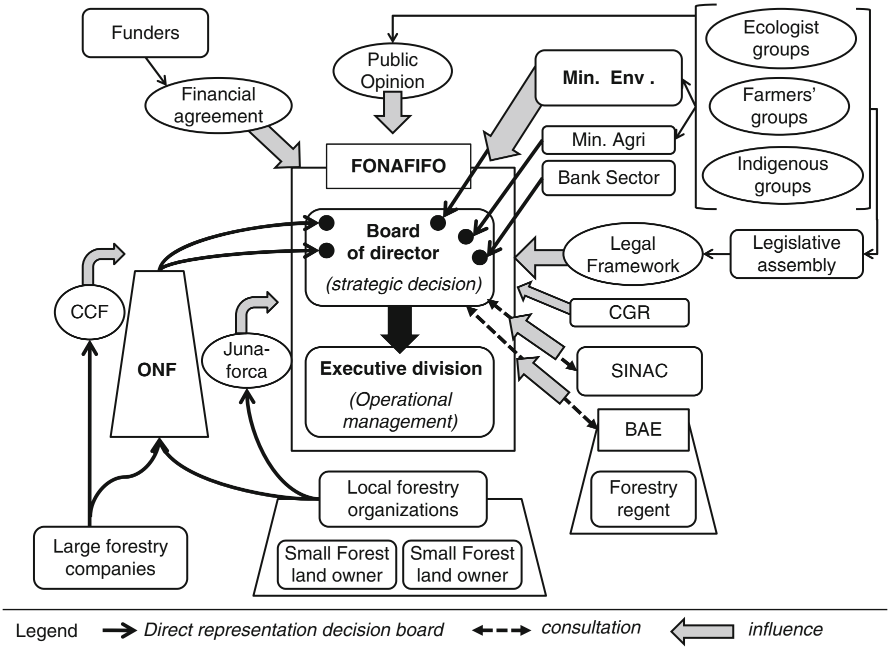

Fig. 12.4 Actors involved in the governance of PESP and decision process (Source: Based on stakeholders' interviews 2008–2011)

(*Junta Nacional Forestal Campesina* – JUNAFORCA). The third actors are those who are not part of the PESP structure but who may influence the evolution of PESP decisions such as (1) representatives of farmers groups, indigenous groups or ecological groups who may have access to lobbying activity on FONAFIFO's board directly or through ministries, deputies or public opinion; (2) funders who can make conditions to their funding agreements; and (3) central state administration and its control bodies (*Contraloria General de la Republica* – CGR) which can evaluate the PESP according to public fund management procedure.

These actors vary in terms of visions, interests and positions regarding forestry problems and policy orientation and thus PESP orientations. Three main stakeholders groups with differences in vision were identified ["as follow"] (Fig. 12.5): (1) agricultural sector representatives, which in the 1980s–1990s considered forests as empty, "unproductive" space; (2) forestry sector representatives that consider forests as "productive" space and a provider of primary material (wood) for the industry; their interests lie in support of wood development production (such as reforestation), and they are prone to be against wood extraction restrictions; and (3) the environmental groups' representatives that consider forests as habitats to be protected to maintain plant and animal biodiversity; thus, they are in favour of incentives for forest protection, ecosystem restoration with native species and the restriction of wood extraction – especially in natural forests (Le Coq et al. 2010c).

sectors	Agricultural		Forestry		Environmental
Stakeholders' groups	Large (business) farmers	Small farmers (peasant)	Small forestry farmers	Large forest landowner and wood manufacturers	Environmentalists and ecologists
Related Public institutions	Ministry of agriculture and animal husbandry		Ministry of Environment		
			FONAFIFO		SINAC
Leading professional representatives	CNAA	UPA National MNC	ONF		FECON
			JUNAFORCA	CCF	
Forest vision	Forest as an empty space "unproductive space"		Forest as a provider of good (and ES – especially Carbon sequestration)		Forest as provider of multiple ES, (and especially support to biodiversity)
Main interests on forest issue	Maintain land use extension for agricultural purpose	Develop agricultural production and agro forestry	Sustain forestry activity (community management for income generation)	Sustain forestry activity (wood production for industry)	Maintain biodiversity and natural ecosystem

Fig. 12.5 Vision and interests of stakeholders' groups (NB: *CNAA* Camara Nacional de Agricultura y Agroindustria, *MNC* MesaNational Campesina, *FECON* Federacion Conservacionista de Costa Rica) (Source: Based on stakeholders' interviews 2008–2011)

12.4.2 1994–1996: The Genesis of the PESP as a Compromise Led by Forestry Stakeholders

In the mid-1990s, the newly elected government of Jose Maria Figueres scheduled the finalization of the forestry law, in discussion since the early 1990s, on the political agenda. The three identified stakeholders groups were in asymmetric positions in terms of involvement and strength. Until the 1990s, the agricultural sector had been a major political force in the country and had blocked former forestry law reforms; however, by the beginning of the 1990s, the agricultural sector began to face numerous difficulties with the implementation of the country's structural adjustment plan. The agricultural sector was facing institutional problems (reduction of civil servants, rapid minister turnover) and strong divisions between representatives of small farmers' movements and large farmers' syndicates. Indeed, whereas agriculture extension was the primary cause of deforestation, between 1994 and 1996, the agricultural sector was poorly represented in the forestry law formulation process (Morilhat 2011). While national environmental consciousness and the number of environmentalist organizations were on the rise in the early 1990s (Steinberg 2001), the environmental sector's representatives, as the newly created Costa Rican conservationist federation (*Federacion conservacionista de Costa Rica* – FECON), were formally poorly involved in the formulation process (Morilhat 2011). In the mid-1990s, the forestry stakeholders group was the predominant actor in mobilizing and empowering the discussion around policy formulation. This group was composed of local forestry organizations (developed in the 1980s throughout Costa Rica) as well as national representative organizations (federated on a national platform in the mid-1990s) such as the Costa Rican Forestry Chamber (*Camara Costarricense Forestal – CCF*), which included within its ranks representatives of large private forestry companies and wood industries as well as representatives of small forestry producers regrouped in the JUNAFORCA and who had the support of international cooperation (Le Coq et al. 2010b). Aside from the private forestry sector organization, the public forestry administration has been strengthened with the integration of the General Direction of Forestry within the Ministry of Environment. Whereas some differences of vision existed inside the forestry stakeholders' group, especially between small forestry groups and large companies' representatives, a common vision emerged around the necessity to maintain forestry support instruments and to consider forest areas for both the products (wood) and the services they can provide to society (especially carbon). Representatives understood these services could provide a new form of justifying support to the forest sector through a PES scheme.

The basic principles of PESP in law #7575 reflect the compromises taken by forestry stakeholders groups. The principle recognition of ES provided by the forest and the principles behind PESP were not thoroughly discussed; however, the law represents a tacit consensus between forestry and environmental stakeholders. The productive forestry stakeholders saw the compromise as a way to justify continuity of support to forestry activities, while stakeholders with environmental sensibility saw it as a way to introduce ecosystemic concerns into forestry policies. Land use change prohibition was the main point of contention for some productive forestry sector stakeholders since it was considered an infringement on the rights to private property use. Nevertheless, it became acceptable for them because (1) they were conscious of the wood shortage risk for the wood industry if the forest resources continued to decrease, (2) they were facing increasing pressures from environmental groups to adopt more sustainable practices, and (3) the law included an article that reinforced their property rights against squatters and a clear financial compensation principle for the restriction of their land use rights through the PES mechanism.

Finally, the key PESP principles reflect the productive forestry stakeholders' interests. First, the forest definition includes regenerated forests or plantations, and the initial PESP modalities include reforestation and also forest management (including wood extraction) that was questioned by environmentalist stakeholders. Second, the law assigns PESP management to a forestry institution (FONAFIFO) in which private forests stakeholders are well represented on the governing board.[11] Third, as a condition of the PES contract, the control of the management plan execution was given to forestry regents that are private forestry engineers.

[11] In 1997, law #7575 created the ONF as a non-state public organization. Conformed by 45 forestry organizations, the ONF is the representative organization for the private forestry sector in regard to the definition of national forestry policies.

12.4.3 1997–2001: The Consolidation of PESP Led by Forestry Stakeholders

From 1997 to 2001, the main focuses of the forestry public administration were to initiate operation of the new programme and to secure programme funding. During this period, the productive forestry sector groups were still considered a strong force and an important resource and were leading the PESP implementation agenda. The CCF was maintaining and strengthening its power, increasing its memberships to 152 affiliates in 1999 and developing services to their members. During this period, the PESP remained within the existing forestry incentive instruments. Taking advantage of its leadership in the governance of the PESP, an additional modality was created in 1998 and 1999 in line with the forest productive groups interests: a PES for established plantations to aid landowners with wood plantation maintenance costs.

Although few stakeholders knew about the ES concept and PES mechanism when law #7575 was formulated and adopted, stakeholders' knowledge increased with the implementation of the PESP. In 1998, a newly elected president, Miguel Angel Rodriquez, following the advice of the vice minister of environment, Carlos Manuel Rodriguez, organized a national consultation on PESP to raise awareness and inform rural stakeholders on the programme. This extensive consultation[12] led to many questions regarding PESP such as the inclusion of all activities that provide ES outside of the forest ecosystem within the scope of the PES. A law project was developed to increase the PESP spectrum to include new ecosystems. The creation of an environmental bank, where all ES provider payments could be concentrated, was proposed. Nevertheless, the law project was too ambitious and difficult to put into practice because, on one hand, all sectors were requesting funding as ES providers (including banana, coffee sector, etc.) and, on the other hand, the ES users (such as the public water distributor and energy producers) were not ready to pay as ES users. Therefore, in this context, the project was abandoned, reaffirming the forestry orientation of the PESP. However, this process led to the broadcast of information and an increase in the appropriation of the meaning of the PES concept among the agricultural and environmental stakeholders. It also brings attention to the necessity to secure a PESP funding source and to diversify outside of the fuel tax. In line with this necessity, new contracts were signed with private enterprises to fund the PESP.

With PESP implementation, environmentalist stakeholders began to pay more attention to the PESP's effects. In 1998, a multidisciplinary group of scholars with an environmental vision analysed the Osa region's forest management plan, a hot

[12] In 1998–1999, a national workshop and three regional workshops were organized. Workshop participation was large and included representatives of various ministries, the private sector, environmentalists' groups, universities and public enterprises for water distribution and energy production.

spot of biodiversity in Costa Rica. This study showed evidence of mismanagement of forestry management plan.[13] Based on this study, environmental groups developed a mass media campaign against these practices, and in 2000, under the pressure of these groups, the Ministry of Environment declared an administrative ban to stop "forest management" and "established plantation" PES modalities.

In spite of the risk of dilution of the PES concept through inclusion of other sectors and the pressure to ban the "forest management" and "established plantation" modalities by environmentalists, the PESP gained its political legitimacy because (1) the final beneficiary demand was important, especially for the protection modality; (2) the FONAFIFO management was effective; and (3) PESP began to be recognized by international forums and communities and was considered as a flagship programme for Costa Rica. In this context, in spite of the opposition of the Ministry of Economy, a strong mobilization of the forestry sector stakeholders (ONF, JUNAFORCA, CCF and MINAET leaders and administration) managed to secure and better channel the hydrocarbon tax fund to FONAFIFO with the reform of the tax system (*Ley de Simplificación y Eficiencia Tributarias*, Nº 8114 of 2001–2000).[14]

12.4.4 2002–2005: Strengthening of Ecological and Social Orientations Under Environmentalist Stakeholders and International Influence

The 2002–2003 year marked changes in the PESP towards a greater focus on environmental and social objectives, more in line with the interests of small farmers, forest landholders and environmentalist stakeholders. These changes reflect a shift in the balance of power between the different stakeholder groups.

In the early 2000s, the interests' groups supportive of a productive forest vision experienced a reduction of their power due to three factors. Firstly, the CCF that had been the primary organized representative force of the private forestry sector began to fade. In 1999, with the change of lead CCF representatives, dialogue between the different groups represented in the CCF (large forestry enterprises, wood industry sector and small and medium forestry producers) began to decline. In early 2000, CCF experienced a rapid disaffiliation and reduction of its means with small and

[13] The study shows that in Osa, the practice of wood extraction that was supposed to be applied to forests under PES-Management was in fact not well applied. Furthermore, adoption of sustainable management practices was shown to be ineffective in relatively small forest plots to maintain biodiversity.

[14] During the negotiation of this law, the Ministry of Economy proposed a fixed amount, but the forest stakeholders managed to obtain 3.5% of the hydrocarbon tax, which has enable them to raise additional funds since the increase of fuel price during early 2000.

medium forestry representatives (JUNAFORCA) splitting from CCF. This, in effect, left only the representatives of large forestry enterprises and industrial sectors. Secondly, the ONF that was supposed to represent the forestry private sector faced financial difficulties and was not able to counterbalance the CCF reduction in strength. Finally, the local forestry organizations began to suffer from the reduction of the direct support they had in the 1990s (Barrantes 2009).

At the same time, stakeholders oriented towards more environmental/conservation or social visions gained forces and took leadership of the PESP agenda setting. Three factors enabled them to gain forces: (1) the increased influence of the international donors in the programme, (2) the change of the Ministry of Environment and (3) the development of new knowledge on ES measurements and PES results. In the early 2000s, facing difficulties in obtaining the funds dedicated to PESP from the Ministry of Economy and without obtaining the expected funds from the carbon markets, negotiations between FONAFIFO, the World Bank (WB) and the Global Environmental Fund (GEF) began. According to their agenda, the WB and the GEF pushed to include higher concerns towards poverty reduction and environmental efficiency in the PESP. In 2002, a new president, Abel Pacheco, nominated a new Ministry of Environment, Carlos Manuel Rodriguez. This new ministry was more sensitive to environmentalists' positions and favourable to the inclusion of other activities that provide ES to the PES, as well as better payment targeting. During this time, the minister assigned a biologist to the FONAFIFO, as the representative of the Ministry of Environment, to better support ecologists' orientation of PES. Thirdly, the concept of PES in general and the Costa Rican experience in particular began to demand more attention in the international and national academic forum. As a pioneer with successful experience, the PESP became the subject of many studies that analysed the effects of PESP on poverty and debated its efficiency, especially in terms of additionality. Moreover, the evaluations of ES provided by diverse ecosystems (such as agroforestry systems or silvopastoral systems) were developed and began to provide evidences of ES provided by non-forest ecosystems. Moreover, other studies such as Gruas 1 identified areas of higher biodiversity interests, yielding tools to better define targeting according to biodiversity protection objectives.

This new configuration of the balance of power and resources between the different interests' groups stakeholders led to an inflexion of the PESP towards a stronger focus on environmental and social objectives to the detriment of a more productive-oriented forest support vision. The unpopular PES modality for "forest management" and "established plantation" was abolished in 2003, and the "Agroforestry System" modality was introduced in 2003 after a campaign led by small forestry (JUNAFORCA) and some SINAC administration representatives, with the support of the Minister of Environment. Moreover, in line with the environmental vision and supported by SINAC civil servants, GEF and ecologist groups, a prioritization system was put in place to better target ES payment towards important biodiversity areas and areas with a lower development index.

12.4.5 Since 2006: A Multidimensional Evolution Reflecting the Complex Balance of Stakeholders' Influence

The evolution of PESP since 2006 illustrates a multidirectional orientation driven by multiple stakeholders who performed a complex equilibrium of power and learning interactions upon ES and PES mechanisms, within national and international forums.

The forestry stakeholders oriented themselves towards a more productive vision and have experienced a modest recovery in strength in the PESP decision process. Since the mid-2000s, the forest issue has dramatically changed from those of the mid-1990s and currently supports a conservation strategy. This is evidenced by the reduction in the deforestation rate and the increase in forest cover. In the 1990s, the deforestation rate was high and forest cover was low (less than 40%), but by 2005, deforestation rate was low, and the forest cover had risen to more than 50% of the country. However, the restriction on forest exploitation in Costa Rica has resulted in the import of wood for industrial purposes. Moreover, in the framework of carbon neutrality by 2021, implemented by President Oscar Arias in 2007, a more intensive use of wood as material is seen as a way to substitute for higher carbon footprint materials (such as cement or metals) arguing for more wood production. These three justifications contributed to a re-evaluation of the payment level for PES-Reforestation to increase incentives for wood production, and the forest management PES modality was reintroduced in 2010 to support wood sustainable extraction.

On the other hand, environmental influence on PESP seems to be fading as the support from international NGOs is decreasing following the financial crisis of 2007 and as other issues have been gaining more importance in the agenda of environmental organizations (i.e. the campaign towards the interdiction of mining of Cruzitas in 2009–2010). Nevertheless, the national environmental mood is still gaining force in the Costa Rican population following the education campaign of the last decade, resulting in forest conservation as an important PES factor and more than 80% of PESP budget being dedicated to PES-Protection modality.

Although initially the agriculture sector representatives were reluctant to accept environmental issues, the environmental issues awareness of some agricultural sector groups has been increased in the last few years (Le Coq et al. 2010a). Although the agricultural sector has not been proactive in PESP governance since the 1990s, some farmers groups recently integrated the ES concept, PES mechanism and PESP to support their activities. As an example, coffee producers, with the help of researchers from CATIE, developed a lobbying process towards the FONAFIFO board in 2008, arguing for the recognition of a new specific PES modality for coffee agroforestry ecosystems.

The latest PESP evolution of PESP appears to be a result of stakeholders' learning process, academic research and international influences, in the context of a steady effort to increase available funds for the programme to enable to pay to more beneficiaries.[15] Hence, with the Ecomarkets II project, GEF grants tend to reinforce ecological orientation of PESP, setting as grant conditions the differentiation of PES payment for conservation in areas of high biodiversity interest that are not included in other existing protection schemes. With the new loan from the World Bank, the orientation towards research of higher programme efficiency through targeting and payment differentiation is promoted. With the expectation of raising funds from carbon market, the PES modality for natural regeneration of pastures has been created and appears to be more eligible to carbon market. Finally, differentiated payment for forest protection in areas of hydraulic interests has been made possible with the development of the water tax funds.

12.5 Conclusion

The genesis and evolution of PESP reflect an evolution of balance of power between different stakeholders' groups. The PESP appears initially as a genuine original construction led by well-organized forestry stakeholders groups, including both small and large forestry enterprises. Its evolution has been influenced by a change in the balance of power between stakeholders, characterized by a reduction of power of the forestry stakeholders defending the productive vision, and a strengthening of the influence of stakeholders oriented towards more environmental and social purpose. As the balance of power between stakeholders appears to be an important explicative factor of the evolution of PESP, the search for funding sources to sustain and enlarge the PESP has been one of the driving forces of the latest PESP evolution. The other driving force has been the learning process: (1) the learning process of the management institution (FONAFIFO) that developed the capacity to adapt the instrument to new constraints and opportunities arising from national situations and international opportunities and (2) the learning process of the stakeholders involved in forest issues and rural development and have developed the capacity to manage the PES concept to support their vision and interests.

The PESP acts as market-based instrument responding to complex governance, where orientation and management depends on the dynamic equilibrium of power and influence among the multiple stakeholders involved. The evolution of the balance of power depends on the capacity of these stakeholders to take advantage of the national and international contexts and to mobilize policy action resources. Beyond the consensual central objective to maintain and develop Costa Rican forest cover, the PESP acts as a multi-objective instrument where the respective importance given to environmental, social or economic dimensions depends on the balance of power between the stakeholders. The durability of PESP relies on the capacity of the PESP, and especially the intermediary institution (FONAFIFO) in charge of its implementation, to maintain the technical management legitimacy and social legitimacy in terms of balance of interests between stakeholders.

[15] In spite of the increase of available funds for the programme, according to FONAFIFO executive officers, only 30–50% of the PES demands are currently covered due to lack of funds.

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Chapter 13
Governance Across Multiple Levels of Agri-environmental Measures in France

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The notion of ecosystem services appeared late in France, which has been reluctant to adopt this idea, choosing instead to defend the concept of the multifunctionality of agriculture. The French position is analysed considering the emergence and then the removal of multifunctionality in the international agenda for agricultural negotiations, followed by the rise of ecosystem services (services provided by ecosystems to society) and environmental services (produced by actors). These trends are reflected by the French agri-environmental measures: a sense of acknowledging and valuing the multifunctionality of agriculture for the management, at the margin, of environmental issues in agricultural policy.

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A comprehensive and institutional approach to payments for environmental services (PES) will be used to examine the case of France. After explaining why agri-environmental measures (AEMs) can be regarded as PES, we will look at their implementation, considering AEMs as public policy instruments (Lascoumes and Le Galès 2005) and looking at how they were put in place in the French and European institutional contexts. The history of AEMs in France can be better understood by incorporating an analysis of the emergence and the removal of the notion of multifunctionality in the design of these agri-environmental mechanisms. It also brings to light the difficulties the successive agricultural policies have had in integrating the different environmental concerns translated into European regulations.

This chapter will focus on the governance issues arising from the introduction of AEMs. These issues, which are specific to France, will be examined at different territorial levels and in several territorial contexts: a region in metropolitan France, Auvergne, and two overseas regions of France, Guadeloupe and Réunion. The analysis of the national governance of AEMs highlights the poor communication between the different administrations responsible for agriculture and environment. The design of the mechanisms associated with the AEMs was led by a highly centralised administration, in cooperation with the majority farmers' union, promoting a mass mechanism in favour of farmers. At the regional level, comparison with the governance of TAEM mechanisms shows that agricultural stakeholders have mixed feelings about them. They are not yet convinced of the effectiveness of the measures they have undertaken and seek above all to maintain their income. The three case studies underline the importance of intermediate actors in the implementation of AEMs at the local level. We identify two types of implementation. First, intermediate actors from the environmental sector integrate the environmental objectives of AEMs and the economic objectives of farmers (e.g. Auvergne).

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Second, intermediate actors from the agricultural sector try to protect the economic interests of farmers. This tends to reduce innovation in the field of environmental protection (e.g. Guadeloupe, Réunion).

13.1 Can Agri-environmental Measures Be Regarded as Payments for Environmental Services?

Environmental integration in French agricultural policies continues to be characterised by a regulatory approach (environmental conditionality introduced by the reform of the common agricultural policy, CAP, in 2003) and the compensation of additional costs resulting from more environmentally sound practices. The notion of environmental services, which emerged in the late 1980s and gradually spread throughout the international political arena (Millennium Ecosystem Assessment 2005; The Economics of Ecosystems and Biodiversity report, TEEB 2009), is emerging as a new paradigm, being mobilised in France to renew the design of agricultural policy instruments in order to respond more effectively to the challenges of environmental integration (French Ministry of Agriculture, Food and Fisheries 2009). It reflects a desire to go beyond the compensation of additional costs that has been favoured so far and whose limitations, especially in terms of incentives, have become clear, to reason in terms of payments for an environmental service provided (PES). Moreover, the agri-environmental policies implemented in the European Union over the last 30 years are generally analysed in the literature as examples of PES (Baylis et al. 2008; FAO Food and Agriculture Organization 2007). These policies are implemented using voluntary and contractual incentive instruments. They include the following devices:

- Local agri-environmental schemes (OLAE), introduced by Council Regulation EEC n°2078/92 of 30 June 1992
- Premiums for maintaining extensive livestock-farming systems (PMSEE)
- Agri-environmental grassland premiums (PHAE)
- Territorial farming contracts (CTE)
- Sustainable agriculture contracts (CAD)
- Territorial agri-environmental measures (TAEM)

The aim of agri-environmental contracts is to encourage farmers to maintain or adopt more environmentally friendly farming practices, while fostering economic development and maintaining the rural fabric. They take the form of a contract by which farmers voluntarily undertake to maintain or adopt these practices, in exchange for payment by the state. This mechanism may appear comparable to a PES, understood as being a financial incentive to produce this type of service. Care must nevertheless be taken regarding the definition of PES considered in this case.

The most common definition of PES is the one proposed by Wunder (2005): a payment for environmental services is "a voluntary transaction in which a well-defined environmental service […] is bought by at least one ES buyer from a minimum of one ES provider if and only if the provider continues to supply that service." Although these characteristics are very similar to those used to define agri-environmental contracts, these contracts cannot be considered as "pure" PES according to Wunder's definition. First, assessments of agri-environmental contracts[1] show that the environmental services associated with them are not always clearly identifiable, making it difficult to measure their environmental impact. Second, the payments made are not conditional upon effective production: they are paid yearly, and the implementation of the contract is monitored by the competent authorities on a random basis.

Although the rationale behind agri-environmental contracts is payment for the provision of environmental services,[2] they cannot be qualified as PES according to Wunder's definition (2005). However, this definition was recently challenged by Muradian et al. (2010), who see PES as the "transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources." This approach provides a broader understanding of PES and makes the concept more appropriate for describing agri-environmental contracts. However, in the case of AEMs, the incentive approach is more a matter of offsetting the costs generated than "paying" for the provision of an environmental service, as the aim of AEMs is to encourage agricultural practices that are compatible with environmental protection through financial compensation for the additional costs and the foregone income resulting from practice changes.

13.2 Governance and AEMs

According to Vatn (2010), who regards the definition of PES provided by Wunder (2005) as an essentially theoretical reference, PES can be analysed as governance structures. We will consider AEMs from this perspective, detailing the governance issues resulting from the implementation of this instrument in the French context. As instruments, AEMs play a part in the regulation and governance of the system to which they belong; we will therefore examine the power relations generated by the instruments in question. These power relations are consubstantial with the concept of public policy instruments defined as a technical and social mechanism that organises specific social relations between the public authorities and their recipients according to the representations and meanings it carries (Lascoumes and Le Galès 2005, p. 13).

[1]Here, we are talking about completed agri-environmental contracts, in other words, those that preceded the TAEMs.

[2]The aim of AEMs is to "encourage farmers to protect and enhance the environment on their farmland by paying them for the provision of environmental services" http://ec.europa.eu/agriculture/envir/measures/index_en.htm.

The term governance, on the other hand, refers to approaches that consider the articulation of modes of regulation and raises questions about changes in political, economic and social regulation.

Four main elements can be used to characterise governance (Boussaguet and Jacquot 2009):

- Institutional complexity (there is no single forum for power and decision-making, and the articulation between these different "forums" is therefore an important issue)
- An increasingly blurred public/private boundary (this is important for PES)
- The procedural dimension of public action: forms and instruments are sometimes favoured over substance (Lascoumes and Le Galès 2005)
- A different relationship with authority (more horizontal, more flexible) and the development of less binding public policy instruments (including AEMs based on contractual approaches)

Some authors are also introducing the issue of the articulation of decision-making levels into approaches in terms of multilevel governance. These stress not only the growing vertical interdependence between stakeholders operating at different territorial levels (hence, the term multilevel) but also the growing interdependence between governmental and non-governmental actors (to which the term governance refers) (Bache and Flinders 2004, p. 96). The repositioning of the state level, the polycentrism, the negotiations and the pluralism at work in public policy are addressed by the proponents of this approach (Kohler-Koch and Eising 1999; Marks and Hooghe 2001). The concept of multilevel governance also helps to identify the reconstruction of areas and levels and the new interdependencies that reveal new problems as well as decision-making forums for tackling these problems, which is important for the analysis of AEMs in France. AEMs will therefore be addressed in this chapter from the viewpoint of these theoretical references, considering first their origin, then their effect on national public policies and finally their impact on regional governance in three comparative case studies.

13.3 The Origin of AEMs in France: A Succession of Mechanisms

In a context of trade liberalisation, and with agricultural aid being called into question, the challenge is to position agri-environmental measures and payments for environmental services in the World Trade Organization (WTO) "green box." Indeed, in the agricultural sector, we are witnessing a global evolution towards less public intervention and greater use of the market as a means of regulation. The history of the French agri-environmental system must be integrated into this global process.

13.3.1 A Brief History of AEMs

The directive on less-favoured areas of 1975 marked the beginning of environmental integration in agricultural policies, acknowledging the role agriculture plays in maintaining the natural environment. At the European level, in 1985 Article 19 of EEC Regulation 797/85 provided for aid for environmental protection initiatives in environmentally sensitive areas. In France, due to the reluctance of professionals, the instrument was implemented later, in 1989, along with the collective land planning operations (OGAF), one objective of which was to reduce agricultural pollution with the construction of the first AEMs, fostering a contractual approach.

In fact, the environmental issue was truly integrated into the framework of the European Union's Common Agricultural Policy during the 1992 reform. EEC Regulation 2078/92 of 30 June 1992 provides for aid,[3] the AEMs, aimed at encouraging environmentally friendly farming practices. France, as member state, has thus been developing agri-environmental programmes since the early 1990s. This "greening" of French agriculture took shape in the emergence of the concept of multifunctionality in the political agenda via the agricultural framework law (LOA) of 1999. Under this law, "agricultural policy shall take into account the economic, environmental and social functions of agriculture and participate in regional planning with a view to sustainable development." This is a fundamental change of direction for the agricultural model set up by the agricultural framework laws of 1960 and 1962.

The key intervention instrument for multifunctionality, the territorial farming contract (CTE) established by the LOA of 1999, is a contractual framework associating the state and the farmer that provides both support for individual productive activities and payment for the provision of public services (corresponding to a social demand previously expressed at the regional level). The CTE system was abandoned in August 2002, several weeks after elections marking a change in the political majority. The sustainable agriculture contracts (CAD), which replaced the CTE, were themselves replaced in 2007 by the territorial agri-environmental measures (TAEMs) that are still in place. The most fundamental changes introduced by the TAEMs in relation to the CTE and the CAD are of two types. First, they concern the withdrawal of farms as a unit for the application of state aid in favour of the region; now only land belonging to farms in predetermined areas is the object of economic compensation. They also concern the refocusing of compensation on the environmental aspect to the detriment of the social and economic dimensions of agricultural activity. Alongside the TAEMs, eight AEMs with specifications drawn up at the national level cover the whole of the national territory, in fields such as the protection of endangered breeds or plant resources, the conversion to organic farming or the modification of technical practices and crop rotation.

[3] Aid part financed – up to 50% for the most part – by the European Agricultural Guidance and Guarantee Fund (EAGGF – Guarantee section).

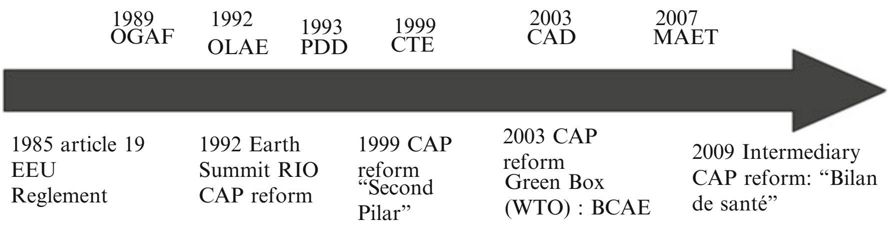

Fig. 13.1 French AEMs: a succession of mechanisms

TAEMs are a mechanism aimed at translating the external, non-market values of the environment into real financial incentives for the local actors who produce such services (Engel et al. 2008).

TAEMs are designed to be applied to targeted parts of priority action areas. These sensitive zones are defined in relation to three types of objectives drawn up at the European level and adapted to the French context: (1) the implementation of the Natura 2000 network (biodiversity conservation), (2) the preservation or restoration of water quality (Water Framework Directive) and (3) other regional environmental issues. But in fact, the agri-environmental measures integrated in different mechanisms (OLAE, EAM, CTE, CAD and TAEM) are all based on the same concept: compensation for the additional environmental costs resulting from the adoption or maintenance of environmentally friendly practices (Fig. 13.1).

The successive changes to the system of agri-environmental measures are marked by the variability of the approaches adopted (sector, region, plot, farm, etc.). They reveal a desire to improve the efficiency of measures, with the successive mechanisms nevertheless constantly favouring a contractual approach (voluntary commitment for a 5-year period) and obligations of conduct rather than of result.

The history of AEMs in France provides a fairly clear illustration of the pressure exerted by the European level regarding the introduction of the environmental dimension into the CAP, with the French State now obliged to transpose this requirement into a national context that is not necessarily a favourable one, especially because of the reluctance of the agricultural sector to integrate the different European environmental directives. In fact, it is through conditionality that they are supposed to be applicable to farmers (Bonnieux 2009). Future European prospects look set to further increase this momentum.

13.3.2 From AEMs to PES? European Pressure via the CAP

In 2008, the CAP health check resulted in 18 % of aid being redirected towards environmental objectives and support for sustainable development in agriculture. Specifically, this movement meant funds were transferred from the first to the second pillar. This period (2008) corresponds to the opening of a opportunity window that intensified in France from 2009 to 2010 during the first discussions on the reform of the CAP after 2013. Debates focused on the announced funding cuts that could be offset for farmers by recognition of the environmental services they provide. The three scenarios currently envisaged by the European Commission opt for varying degrees of "greening" of the CAP. A first option consists in conserving the two current pillars and introducing progressive changes focusing on the environment. A second option is to establish compulsory additional support for the first pillar (compulsory, annual, comprehensive, non-contractual). Finally, a third option would lead to abandoning income support measures and market measures and concentrating all aid on environmental objectives. In this case, direct aid would be axed and replaced by environmental aid. Irrespective of the scenario eventually chosen, it seems clear that by making the CAP "a competitive European policy in both environmental and economic terms," policymakers are establishing a basis for payments for environmental services provided by farmers, even though the term service does not appear explicitly.

In France, the implementation of the AEM system affected the way public agricultural policy is conducted at the national level.

13.4 Implementation of AEMs: Issues of Governance at the National Level

In France, the implementation of AEMs has had an impact at the national level (especially on the way the links between agriculture and environment are approached) that questions the methods of policymaking on this issue. As public policy instruments, AEMs are also ways of looking at the world and especially in agriculture and its relationship with the environment: "one could say that the instruments mobilised to address agricultural issues express the vision that, at a given time, will become the reference used as the basis for tackling the agricultural problem" (Muller 2010, p. 340). In other words, although the procedures (and instruments) do not work exclusively towards the resolution of problems and above all create specific frameworks for interaction to "construct" problems and interpret the action undertaken (Lascoumes 1993, p. 104), AEMs have transformed the way in which the link between agriculture and the environment is built.

13.4.1 AEMs Question the Sectoral Basis of Public Policy

In a country with a tradition of centralisation and a highly sectoral, top-down organisation of public policy (Jobert and Muller 1987), the existence of agri-environmental measures is a development, imposed by the European Union, that questions the distribution of roles and power in French government departments between the agricultural sector and the environmental sector. Agri-environmental policies are by definition compromise policies – between government sectors with different approaches, between opposing rationalities (e.g. producing at lowest cost vs. protecting the environment with costly measures) and between actors (numerous and heterogeneous) – that have different approaches to action (Lascoumes 1993, p. 18). More than others, these policies are the result of mutual adjustments between different actors, approaches and rationalities (Lascoumes and Le Bourhis 1997).

Thus, at the level of the central state administration, the Ministries of Agriculture and Ecology[4] are both taking an interest in the environmental services provided by agriculture. Their activities take place both internally (in commissions dedicated to the agriculture/environment interface in each of these ministries) and at the inter-ministerial level.

Ten interviews[5] conducted in 2010 with Ministry of Agriculture officials reveal that environmental services are being given greater consideration within the ministry. This has been particularly noticeable over the last 5 years (corresponding to the introduction of cross-compliance into the CAP), with an acceleration in 2009 and 2010. But this ministry's position on environmental services remains somewhat detached: the primary function of the ministry is, according to its officials, geared towards agricultural production and farm income, with the environment seen as an important but secondary concern. This led in particular to the integration of AEMs in an individual contract (CTE then CAD) at farm level, combining economic and environmental measures. Finally, the implementation of AEMs lacked any articulation with other agri-environmental mechanisms linked to the Nitrates Directive or to the pesticide plan, for example. While the Ministry of Agriculture's position on environmental services favours a "sector-based" approach (agriculture), Ministry of Ecology officials approach this issue differently. The six people interviewed at the Ministry of Ecology stressed the importance of the CAP for the development of discussions on the issue of environmental services in France, discussions to which the officials of this ministry contribute not only in the commissions dedicated to the agriculture/environment/biodiversity interface but also within joint commissions.

In late 2010, the Ministry of Ecology thus issued a memorandum presenting its position on the reform of the CAP: the concept of environmental services provided by agriculture was used extensively in this memorandum. The ministry thus expressed its support for an architecture of the CAP on two levels: the first guaranteeing "a base of farm income and practices" and the second "paying for environmental services provided." According to the document, this second level was explicitly intended to "foster methods and systems of production corresponding to

[4] French Ministry of Ecology, Energy, Sustainable Development and the Sea (MEEDDM).

[5] Interviews conducted as part of the SERENA research programme, see Aznar O., Valette E., Amon G., Augusseau X., Bonin M., Bonnin M., Brétière G., Caron A., Daré W. s., Déméné C., Déprès C., Décamps M., Gomes M., Hrabanski M., Jeanneaux P., Maury C., Queste J., 2010, *Emergence de la notion de Service Environnemental en France*, SERENA Programme, Working document n°2010-02, 66 p.

practices recognised for their environmental services. The aim is not to compensate for any foregone income, but beyond this, to actually pay for environmental services provided."[6]

This stance was heavily criticised by the professional agricultural organisations and the Ministry of Agriculture and was rapidly withdrawn from the Ministry of Ecology's website. This was a reminder that in political arbitration, AEMs are first the responsibility of the Ministry of Agriculture and its farming partners. Indeed, the sudden emergence of the Ministry of Ecology in these matters seems to disrupt the traditional channels of negotiation and co-management set up since the 1950s at all territorial levels between the state and farming representatives. At the interface between agriculture and environment, AEMs are introducing new dimensions into this partnership.

13.4.2 Do (T)AEMs Reveal Divides?

With AEMs, the whole structure of state governance is brought into play through its devolution and coordination mechanisms at the different levels of intervention, ranging from the European Union to the territory (with environmental issues). Consequently, the state controls the whole standard production process and organises the interface with socio-professional and environmental stakeholders. The demands of service producers, farmers, foresters and water users are reinterpreted within joint management structures. Some of the grievances of the most powerful pressure groups are dealt with by differentiating the instruments (multiplication of standard measures) or by adapting the conditions of use.

The implementation in France of the first AEMs in 1992, further to the renegotiation of the CAP, thus followed the traditional channels of the French co-management system, associating farmers unions and especially the majority union, the *Fédération Nationale des Syndicats d'Exploitants Agricoles* (FNSEA – French national federation of farmer's unions), with any decisions or changes concerning agriculture. The specifications for the first AEMS were therefore negotiated at the national level through the traditional channels, and during their implementation, arbitration was conducted at the departmental level, within the *Commissions Départementales d'Orientation de l'Agriculture* (CDOA – Departmental Agricultural Management Commissions), which are largely dominated by farmers' representatives (e.g. chambers of agriculture, *Centre National pour l'Aménagement des Structures des Exploitations Agricoles* – CNASEA, National Centre for the Development of

[6] "Pour une politique agricole durable en 2013. Principes, architecture et éléments financiers," French Ministry of Ecology, Energy, Sustainable Development and the Sea (MEEDDM) document, 2010, p. 5.

Farming Structures).[7] However, for TAEMs, the negotiations took place at smaller regional levels and introduced new stakeholders and new scope for action. The majority union's position on these measures is therefore far less favourable than the one taken at the national level for AEMs, partly because its representatives feel they are not "in control" of the mechanisms. In the case of the Auvergne region, for example, the Chamber of Agriculture acted more as an obstacle than an ally to this issue. In union discourse (especially *Jeunes Agriculteurs*, the FNSEA[8] and *Coordination Rurale*) at the national level regarding AEMs, and more broadly PES, an ideological argument that is fairly widely shared places at the heart of agriculture, and thus of aid for agriculture, its productive function, which also partly explains this relative detachment. An employee of *Jeunes Agriculteurs* thus indicated that for his union, "the primary function of farming and of farmers is the production of food and certainly not to produce ES, even if these are paid for."[9]

These union actors tend to systematically link the issue of PES with the CAP and especially its renegotiation for 2013: European CAP funding cuts are expected, and these actors see PES as a way to offset the cuts, while indicating that this function of agriculture should not take precedence.

For the *Confédération Paysanne*, the discourse is far more favourable to the integration of the environmental services provided by agriculture, especially within the framework of AEMs and TAEMs: "The *Confédération Paysanne* has progressively evolved; the concept of the environment entered the discourse in the 1980s with the issue of charges for environmental damage. The issue of PES is a classic within the *Confédération Paysanne* […] I prefer to talk about payment for services rather than aid."[10]

These different elements support the assumption of a two-tier agriculture proposed by P. Muller (2010, p. 348), which would partly explain these opposing positions on AEMs and TAEMs. These two tiers can be summarised as follows:

- An agriculture centred on mass food production for which compliance with environmental standards is a constraint imposed according to external global standards (and for which organised interests will wield their influence in negotiations with the state). This is the kind of agriculture that the FNSEA tends to supports, for example, firmly negotiating to ensure aid under the first pillar of the CAP is maintained at a sufficient level and fearing that the environmental measures that are currently eligible for payment via the AEMs will become compulsory (and without financial compensation) in the near future.

[7] See Rapport d'Evaluation à mi-parcours portant sur l'application en France du règlement CE n°1257/1999 du Conseil, concernant le soutien au développement rural, Chapter VI: "Soutien à l'agroenvironnement," January 2004, CNASEA.

[8] The FNSEA is the majority agricultural union in France.

[9] Interview conducted in Paris in December 2009 as part of the SERENA research programme

[10] Interview conducted in 2010 with the president of the GMO seed commission for the *Confédération Paysanne* as part of the SERENA research programme.

- Territorial agriculture centred on local economic activities (of which the provision of environmental services may be one of a number of components), for which the environmental constraint is a resource linked to global standards that are internalised or at least territorialised. This is the type of agriculture generally supported by farmers who adopt contractual TAEMs. The contact established between these farmers and environmental stakeholders is a decisive element in this renewed understanding of the environmental constraint and its integration into the agricultural sector.

The implementation of TAEMs is in fact based on stakeholders responsible for environmental management, whose objective is to preserve the quality of ecosystems and who will seek environmental service providers to this end. Although farmers are a key part of this mechanism, they are, according to environmental stakeholders, paid for technical action that is beneficial to the environment and not compensated for the additional costs resulting from practice changes – what the agricultural profession wants. In this sense, the emergence of new operators could foreshadow the appropriation of the PES referential at the local level, despite the highly variable degree of agricultural sector involvement in the implementation of the TAEM mechanism.

The implementation of agri-environmental measures therefore also has an impact on local governance.

13.5 AEMs and Territorial Governance

The territorialisation of the AEM mechanism results in changes in the governance of agri-environmental issues in different areas. But the instrument itself varies according to local interpretations. By comparing three local adaptations of the mechanism, we will show that the territorialisation of AEMs differs according to several variables. We identify two main variables:

– The articulation of AEMs with other existing (or past) mechanisms and instruments
– The targeting of the mechanism in environmentally sensitive areas

13.5.1 Three Contexts

Réunion and Guadeloupe are French departments that are marked by their insularity and their distance from metropolitan France (as OMRs), their tropical situation in the Indian Ocean and the Carribean Sea and their agricultural history inherited from the colonial period. Agriculture on the islands is traditionally dominated by a sugarcane and livestock-farming sector in Réunion and by banana cultivation and sugarcane in Guadeloupe. It is supported by an agricultural policy geared towards the consolidation of the different sectors, with an emphasis on high productivity. Nevertheless, the islands face considerable environmental challenges due to close connections between urban and agricultural areas (a density of 600 ha/km² in the useful part of the island in Réunion), to the existence of biodiversity characterised by a high level of endemism that has earned Réunion UNESCO World Heritage status and French National Parks status and by pesticide pollution in Guadeloupe.

Auvergne is a region in metropolitan France where grass-fed cattle farming is predominant. It nevertheless has a wide range of farming systems: the Limagne plain has cereal farms, while the mountainous region has a high concentration of suckler cow and dairy farms, which are mainly geared towards cheese production (the region holds five protected designations of origin). Auvergne has some interesting biodiversity and a good quality environment, except for pesticides in the Limagne plain and nitrogen residue in dairy farming areas. In the Allier valley, intensive maize production leads to problems regarding water pollution and the sharing of water resources.

13.5.2 Agricultural Stakeholders Cautious About the Territorial Agri-environmental Mechanism

The "territorialised" dimension of the TAEM mechanism was already found in the OGAF and OLAE. In TAEMs, it implies a certain number of singularities in terms of its implementation. Thus, like the previous measures, TAEMs are still coordinated by the deconcentrated departments of the Ministry of Agriculture – with priority to the regional level with the *Direction Régionale de l'Alimentation de l'Agriculture et de la Forêt* (DRAAF – Regional Directorate for Food, Agriculture and Forestry) over the departmental level with the *Commissions Régionales Agro-Environnementales* (CRAE – Regional Agri-environmental Commissions).[11] However, TAEMs must be drawn up by a local project leader for every sensitive area in order to ensure their adaptation to the specific context and challenges of this area. A limited number of measures (unit commitments) must be selected for each area in order to make actions clearer and more coherent. The TAEM mechanism is intended to foster the emergence of territorial project leaders or operators. These appear as the preferred contacts for farmers whose farms are located in sensitive areas. The agri-environmental operator may be nominated further to a spontaneous application, an active encouragement or a response to a call for tenders by the state departments.[12]

[11] The CRAE is mainly made up of representatives of the DRAAF, the DREAL and the *Agence de l'Eau*. It also includes members of the DDTs, departmental councillors, the ASP, all the AEOs concerned and the ADASEAs.

[12] Circular DGPAAT/SDEA/C2010-3059.

13.5.2.1 Farmers' Motivations for Adopting Contractual TAEMs

In all of the regions studied, many farmers stress that the measures they choose to formalise by contract are primarily those that enable them to receive aid without any practice changes. Contrary to these financial considerations, there was little reference to environmental concern as a reason for committing to contracts. In Auvergne, farmers have mixed feelings about the environmental efficiency of TAEMs and stress their historical role in the preservation of the Auvergne environment that they would like to see recognised. In Réunion, the review of motivations, conducted across two water protection areas, shows that for both farmers and technicians, water is by no means a priority in the choice of AEMs (Herrou 2010).

Farmers have become aware of the gradual reduction of their scope for action in the choice of articulated mechanisms: for example, in Réunion, to subscribe to a MCAE (AEM for sugarcane), cross-compliance principles must be followed (Queste et al. 2011).

The succession of mechanisms increased farmers' doubts and fears concerning the hidden objectives of this new agri-environmental policy (further decoupling of aid and the introduction of cross-compliance; concerns about the emergence of more stringent environmental regulations).

In Guadeloupe, farmers' motivations for signing contracts with the state were fairly similar: the search for higher income combined with a commitment to marginal change or even maintenance of their technical practices. Despite some major changes in principles and objectives from the TEC to the SAC mechanisms and then TAEMs, there has been considerable continuity in the measures and their main beneficiaries. The measures have evolved little,[13] and the main beneficiaries of the programme remain banana growers, who receive most of the available budget.

This cautious positioning of farmers on TAEMs must be weighed against the limited involvement of the chambers of agriculture and farmers' unions associated with the implementation of the mechanism.

13.5.2.2 The Limited Involvement of Departmental Chambers of Agriculture in TAEMs

The varied positioning and levels of involvement of the departmental chambers of agriculture (CAs) in the implementation of TAEMs are worth noting. Whereas they played a decisive role in the previous contractual agri-environmental mechanisms (from the adaptation of Article 19 to the CTE/CAD), only one CA positioned itself in Auvergne as an agri-environmental operator (AEO) for TAEMs on biodiversity issues. None of them did so for water issues. The involvement of this CA, which was an exception to the rule, was the logical continuation of previous coordination

[13] Apart from the replacement of AEMs for perennial high-altitude banana plantations by support for fallow practices.

action for the implementation of Natura 2000 (Noulin 2010). The other CAs delegated the implementation of TAEMs to protect their image among farmers and instead criticised the fact that the procedure completely dissociates the environmental element from the economic element (contrary to the CTE and CAD). In Réunion, the CA was actively involved in the formalisation of CAD with farmers, especially sugarcane planters. The CAD was seen as a highly innovative mechanism to support farmers and renew farm advisory services (Chia et al 2008). By contrast, the TAEMs have not been truly appropriated by the consular institution. Priority was given to technicians to invest in the creation of global farming projects (PGE), a procedure imposed by the commission, which conditions access to investment measures. Finally, the CA technicians had limited resources and information for publicising and encouraging the formalisation of TAEMs. In Réunion, this is also explained by the influence of the dominant sectors (sugarcane and livestock rearing) over local agricultural policy and consequently over the design of AEMs (Daré et al. 2011).

In Guadeloupe, the Chamber of Agriculture has started to record CTE at the end of the device, but the commitment of the Chamber of Agriculture has changed with the replacement of the CTE by the CAD. The deconcentrated services of the Ministry of Agriculture seem to keep the management and mastery of MAE, from the MAE incorporated in CTE in 2000 to MAET and MAE until today. In Guadeloupe, the influence of the dominant sectors (banana, sugar cane) has not led to a thorough renovation of the technical models incorporating environmental issues. The CTE and CAD were thus mobilised as complementary mechanisms aimed at strengthening the structure of sectors and reinforcing existing farms (Dulcire et al. 2006). Since the joint introduction of cross-compliance and TAEMs, agri-environmental mechanisms now play a very different role. The influence of the dominant sectors over the TAEM mechanism, in Réunion and Guadeloupe, is also explained by the fact that the mechanism has not been targeted at "environmentally sensitive" areas. Comparison with the case of Auvergne is very instructive in this respect.

13.5.3 TAEMs or the Territorialisation of an Environmental Issue: Contrasting Situations

Comparing the three case studies shows how the territorialisation of the mechanism, adapted in the case of the TAEMs according to environmental issues in metropolitan France, affects the governance of the mechanism.

13.5.3.1 Sectors Versus Environmentally Sensitive Areas

The CTE and CAD mechanisms, which preceded the TAEMs, had a limited territorial approach; most collective projects were disconnected from the territory in favour of the sectors (Gassiat et al. 2010). In the TAEM mechanism, territorialisation is a key element but mainly concerns the territorialisation of environmental issues. Achieving better environmental efficiency implies establishing coherent territories from an environmental viewpoint, and this was the basis for the territorial adaptation of the mechanism, which favoured areas identified as "sensitive," essentially corresponding to Natura 2000 and the Water Framework Directive (WFD) in metropolitan France.

In this process, farmers located in sensitive areas – the target areas – can sign contracts while those not in such areas are not eligible for TAEMs. This differs considerably from the previous mechanisms, in which contracts were a commitment by farmers, with no reference to the territory. In the case of TAEMs, there may be a disconnect between what is appropriate in terms of the environmental project for a territory and what is appropriate for farmers.

This potential disconnect further underlines the importance of the role of intermediate actors "operating" TAEMs, who ensure coordination and negotiations with farmers in order to attempt to close the above-mentioned gaps as far as possible. This also explains the reluctance of traditional operators, who came only from the agricultural sector (e.g. CA) and who are unable to relate to these mechanisms that are territorialised from an environmental rather than an agricultural viewpoint.

Comparing the case of Auvergne with that of Guadeloupe and Réunion provides a number of insights into the importance of the territorialisation of the environmental issue for the appropriation of the TAEM mechanism. Indeed, in the case of metropolitan France, the TAEMs have been adapted to predetermined sensitive areas by the transposition of European law: the Natura 2000 areas and the priority areas under the WFD, for which agri-environmental operators readily declared their support as these are generally the structures in charge of the management and coordination of these areas. On the other hand, in the case of the overseas departments, since the Natura 2000 and WFD zoning is not yet completed, the search for agri-environmental operators has proved problematic, leaving the sectors free reign to take over the mechanism.

In the context of Réunion, it is also necessary to add the weight of the "administrative" inertia of these mechanisms and the handout approach that tends to favour measures that are easy to manage and target the highest number of farms already identified in administrative databases (Daré and Queste 2011). The two main sectors, sugarcane and livestock rearing, have largely benefited from this approach, which directs the mechanisms towards farm support. Thus, agri-environmental measures for sugarcane (MCAE) and agri-environmental grassland premiums (PHAE) account for the greater part of commitments. For livestock rearing, the flagship measure concerning pasture management overshadows other measures that could contribute to improving the environmental record of livestock farms.

13.5.3.2 The Influence of Intermediate Actors and of Their Absence

The research conducted in Auvergne shows that the territorialisation of agri-environmental policy sought through the TAEM mechanism works through a type of state delegation of public services to intermediate organisations (project management structures from environmental protection associations, joint unions, regional authorities, the *Office National des Forêts*, consular organisations, etc.). These project designers and leaders play a key, varied role in the implementation of the formalisation mechanism. In Auvergne, they are relatively specialised in each of the issues – whether water or biodiversity protection – identified as priorities at the national level. At the interface between government departments and farmers, these agri-environmental operators guarantee better coherence between the definition of measures and local challenges and also ensure greater involvement by the farmers concerned (a 70% contract rate in the areas in question).

The arrival of TAEMs in Réunion did not result in the emergence of new intermediate actors capable of making them operational. Let us consider the TAEMs linked to the protection of water resources and the failure to implement them. This failure is chiefly explained by the delay in the local application of the WFD. Indeed, it is faced with governance difficulties for the implementation of a water development and management plan (SDAGE), which reflect the fragmentation of responsibilities between the authorities and the government departments and a lack of consultation. Consequently, the development of territorial diagnosis that are relatively detailed and mobilise different partners (agriculture and environment, but from which the *Office de l'Eau* is absent) did not ensure optimal management, especially in terms of the effective targeting of AEMs in the areas concerned. There is no institution in a position to encourage and formalise TAEMs for water with farmers.

On the contrary, in the case of Auvergne, intervention by the *Water Agency Loire Bretagne* has led to greater attention being given to the environmental efficiency of contract-based measures, especially concerning stricter monitoring of coherence between the measures chosen and the recommendations made within the framework of the diagnosis for plots considered. In Auvergne, the intervention of the *Water Agency* thus results in a better integration of the environmental efficiency objective – or the principle of cross-compliance – in the implementation of TAEMs for WFD issues.

In Guadeloupe, with SACs, then TAEMs, the agri-environmental systems were gradually recentred on environmental challenges and partly lost their strategic interests for operators in the main agricultural supply chains. The banana sector, which was closely involved in the TEC debate, has nonetheless remained the main beneficiary of AEMs, notably through a specific "banana cover: bare fallow" AEM, which has involved most of the application files accepted and the payments made. When TECs arrived in Guadeloupe at the beginning of 2000, the environment was not a priority concern of the banana supply chain. With the "chlordecone crisis"[14] of the 2000s, agricultural stakeholders in Guadeloupe changed their views of the environment issue, having previously been somewhat unreceptive to it. Consequently,

[14] Linked to the discovery of water, soil and plant pollution by a very persistent molecule used until 1993 to control the banana weevil.

the banana supply chain in Guadeloupe has truly converted to the environmental cause, seeking to restore the image of the sector and to take part in defending its economic interests. “Sustainable banana” is used both to distinguish the products on the increasingly competitive European market and to continue benefiting from the public aid granted by the supply chain (Cathelin 2010). In this context, AEMs appear to be the appropriate instrument for defending an agricultural production sector by increasing its green credentials.

Even if the territorialisation of the TAEM mechanism according to environmentally sensitive areas results in segregation between farmers and between areas that may have adverse effects, this is in fact a rather positive point in the case of Auvergne. First, because the mechanism as it stands enables the emergence of intermediate actors (the agri-environmental operators) who bridge the gap between agriculture, environment and territory. To do so, they mobilise different resources resulting from their presence in the territory, especially local coordination. However, in the Auvergne region, the introduction of the agri-environmental operator (AEO) had a beneficial effect as an intermediation structure. The introduction of this territorialised negotiation process has changed the nature of interrelations between players and widened the range of possible choices, which were previously limited to accepting, or not, the imposed specifications. From now on, farmers can make suggestions for drawing up measures specific to the zone they are involved in.

The absence of this intermediation activity in the case of Réunion and Guadeloupe, due to the delay in the overseas departments in the establishment of Natura 2000 and WFD areas, changes the face of the mechanism. Thus, although comparing the three cases shows that farmers’ motivations for signing contracts and the position of chambers of agriculture are similar, the territorialisation of the TAEM mechanism according to environmental concerns changes the contract coverage rates and gives farmers some leeway in negotiations, especially because of the emergence of intermediate actors. In the absence of this environmental territorialisation, the overseas cases show that the lack of intermediate actors leaves the dominant sectors free reign to regain control of the mechanism.

13.5.3.3 Conclusion: 2014 Prospects Under Debate

AEMs and TAEMs are instruments similar to PES that are strongly marked, in the French case, by institutional path dependencies expressed in different ways.

At the macro level, these path dependencies are seen in the defiant attitude of the professional agricultural organisations towards an instrument that marks a certain distancing from the agricultural activity and the protection of farmers’ economic interests. The professional organisations’ misgivings are also linked to the involvement of the Ministry of Ecology calling into question the recognition of agricultural sector specificity and its regulation, since the 1950s, by co-management between the Ministry of Agriculture and the farming profession (agricultural lobby). A third cause of reluctance is the fact that AEMs, which are drawn up at

European Union level, reveal the European level's control over the national level in terms of agricultural management.

At the local-territorial level, the path dependencies are seen in the permanence and the adaptation of many institutional actors who took part in the implementation and operation of previous instruments (OGAF, OLAE, CTE/CAD), responsible for mediation between the authorities that define the regulatory frameworks, situated at the European and national levels, and the farmers applying the instruments. The aim of these intermediate structures is to adapt measures in view of the environmental issues of the territory without disregarding the socio-economic conditions of production. In some cases (Auvergne), these intermediate structures have evolved, mobilising actors from the environmental sector, and have succeeded in innovating in the identification of the measures to be implemented. In other situations (Guadeloupe, Réunion), their concerns are marked by the desire to protect the economic interests of the agricultural sector, which tends to reduce innovation in the field of environmental protection.

As part of the preparations for the reform of the PAC, initial projections support the maintenance and consolidation of AEMs and TAEMs. Discussions focus on several points. First, they concern the terms of payment for farmers, a subject of disagreement between the proponents of subsidies and of service provision. They also focus on the nature of the AEMs that should be encouraged, given that the environmental performance of the most widespread AEM (the grassland premium) is debatable, while AEMs with limited application (such as the conversion to organic farming) appear to have a positive environmental impact. Finally, they concern the governance of the mechanism to find the best balance between the efficiency of measures and their administrative management costs.

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Part III
The Social Embedding of PES

Chapter 14
Indigenous Protected Areas in Australia: The Importance of Geophysical and Institutional Scale in Assessing Their Effectiveness for Biodiversity Conservation

Nanni Concu and Katherine May

14.1 Introduction

Indigenous Protected Areas (IPAs) are regarded as a successful instrument for environmental conservation and indigenous development in Australia (see, for instance, Gilligan 2006; SEWPaC 2010a; Smyth et al. 2004). In this chapter, we critically engage with the unique institutional arrangement of IPAs to highlight some of its complexities. We draw attention to instances of convergence and tensions between indigenous and non-indigenous interests, goals and values by unpacking the IPA framework through the concept of scale.

IPAs are established on indigenous-owned land or sea to promote biodiversity and cultural resource conservation. They are a form of Indigenous and Community Conserved Area (see CENESTA 2009). IPAs are voluntarily declared by indigenous landowners who manage them according to International Union for Conservation of Nature (IUCN) guidelines. There is no federal, state or territory legislation regulating IPAs; they are managed by 'legal and other effective means' consistent with the IUCN guidelines by indigenous landowners as part of Australia's National Reserve System (NRS). The federal government provides financial resources for planning and management through the IPA programme and several other funding schemes. Many IPAs also receive financial and in-kind support from state and territory government agencies, non-government conservation organisations, research institutions

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and commercial enterprises. The first IPA was declared in 1998. At the time of writing, there are 40 declared IPAs across Australia, covering over 23 million ha. Consultation projects are under way for another 40 (SEWPaC 2010a). IPAs currently represent around 23% of the NRS, and they include some of the most biodiverse and highly valued of all NRS properties (Gilligan 2006).

IPAs in Australia occupy a unique intercultural space by incorporating and integrating non-indigenous institutional elements within indigenous landownership, culture and management systems. In Brenner's phraseology, IPAs are fluid and contested *scalar constructs* (Brenner 2001); they are an assemblage of cultural, historical and institutional arrangements and processes interacting in a geographical space, defined according to non-indigenous spatial representation. IPAs are then the product of scalar politics or relations between actors with different powers, capacities, opportunities, constraints, and access to, and rights over, resources.

Scalar politics intersect the spatial and temporal span of ecological processes (Barber 2005). The flows and stocks of environmental goods and services (e.g. timber, water quality, biodiversity) are generated at scales that transcend ephemeral administrative and political regions (Brunckhorst and Reeve 2006). Often overlapping jurisdictions and competing agencies with different values hinder effective management and lead to conflict rather than cooperation (Albrecht et al. 2009). Thus, environmental management that relies solely on political/administrative boundaries is unlikely to be the most effective approach to conservation. It is increasingly recognised that managing environmental public goods necessitates a multilevel approach based on the vertical and horizontal integration of institutions and actors that overcomes disciplinary biases or power imbalances (Berkes 2007; Adger et al. 2005). This recognition matches indigenous conceptualisations of 'country' that transcend administrative distinctions between sea and land, taking a holistic management approach (Baker et al. 2001; Barber 2005; Dhimurru 2006; Rose 1996; Yunupingu 1997). Therefore, effective environmental management requires fostering vertical and horizontal integration within IPAs and analysing its results in terms of convergence of, and tensions between, indigenous and non-indigenous values, interests and knowledge.

In this chapter, we examine the extent of this convergence and the associated tensions within the IPA framework and hence identify some issues that should be addressed to minimise conflicts, foster cooperation and promote effective environmental conservation. Following Zulu (2009), we adopt a 'scale perspective' to analyse the different aspects of IPAs. Scale analysis identifies the spatial extent of agents' powers, the nature of their rights, opportunities and capacities, and their relations. Such cross-scale analysis of resource management is currently under-researched (Adger et al. 2005). Existing literature on IPAs in Australia centres largely on the historical context of the framework (Szabo and Smyth 2003), comparisons with other ways indigenous Australians are involved in natural resource management (Bauman and Smyth 2007; Orchard et al. 2003; Smyth et al. 2004) and on studies of particular IPAs (Muller 2003). Langton et al. (2005) provide an assessment of the IPA framework in Australia but limit their analysis to the discussion of instances of convergence of indigenous and non-indigenous interests. Smyth (2008) discusses specific instances of conflicts and tensions and the emergence of alternative organisational IPA models, specifically addressing an indigenous perspective of scale (see also Muller 2008b, c). This chapter, whilst drawing on the existing literature, provides a broad analysis of the different aspects of IPAs across spatial scales, in different bioregions, and across different land tenure systems and colonial histories. We first introduce the concept of scale and some methodological issues. Then we examine the scalar construction of IPAs, identify actors, institutions and their relations. We specifically highlight instances where scalar relations generate convergence and tensions between indigenous and non-indigenous interests. Next, the IPA case studies are introduced and collectively examined. We then discuss the major issues that emerge from the scale analysis and some of the implications.

14.2 Scale Analysis and Methodological Issues

Over the past decades, a robust body of literature has emerged on the theory of social construction of scale (Brenner 2001; Cox 1998; Delaney and Leitner 1997; Howitt 1998, 2002, 2003; Smith 1992; Taylor 1993; Zulu 2009). Central to this literature is the idea that scale arrangements are not an ontological given but are produced by the motivations and strategies of social actors. Through policies and practices that alter and produce scale, social actors can change decision-making processes, access to resources, institutional arrangements and the physical environment (Brown and Purcell 2005; Zulu 2009). Swyngedouw (1997) sees scale as the arena through which empowerment and disempowerment are produced and where power relations are contested and compromises are negotiated and regulated. Here we adopt Howitt's (1998, 2003) notion of scale as having at least three elements: size (or spatial extent), level (position on the scale) and relation (the interaction between levels). Howitt (2003) suggests that the social construction of scale is a 'vehicle for participation, recognition and change', emphasising the importance of 'links within and across scales for providing opportunities for transformation of existing power relations'. In our analysis of IPAs, we use 'level' as a means of organising the different actors, institutions and processes. We investigate what factors are most likely to contribute to the definition of the spatial extent—or size—of IPAs. We also analyse relations between actors, institutions and processes to highlight instances of conflict and convergence.

We used several methodological approaches for primary and secondary data collection and analysis. A review of the literature—including government documents and reports—and informal discussions with experts on IPAs provided the information to understand the institutional context of IPAs. Publicly available GIS datasets were used to analyse the relationship between the spatial boundaries of several ecological management units, including IPAs. The spatial data was then used to compute the statistical relationship between the size of IPAs, environmental health indicators for bioregions and river basins, land uses and institutional factors such as land rights.

Using regression analysis, we aimed to understand what factors at the national level affect the establishment and size of IPAs.

Complementary primary data were collected through fieldwork visits to some of the IPAs used as case studies. Other primary data were collected through a questionnaire to the managers of all the IPA case studies, selected to represent a range of IPAs established in Australia. The questionnaire design is based on the World Commission for Protected Areas (WCPA) framework (see Hockings et al. 2006). Primary data collection sought to identify strengths and weaknesses of, barriers to and means for effective management of IPAs.

Important methodological caveats are in order. Firstly, we acknowledge that scalar analysis is not all-encompassing and scale centrism has been widely critiqued as such (Brenner 2001; Leitner and Miller 2007). Secondly, scale analysis is here conducted without fully accounting for the indigenous conceptualisations and meanings of space. IPA boundaries are indeed the attempt to make manifest indigenous ownership according to non-indigenous spatial representations (Morphy and Morphy 2009). We are aware that scale analysis is in itself a culture-specific framework, with limited ability to capture every conceptualisation of human/nature interactions.

14.3 The Scalar Construction of IPAs

Institutions, actors, motivations and drivers at play in the scalar construction of IPAs define size, boundaries and the extent of power, authority, rights and responsibilities over the physical environment. Figure 14.1 illustrates the main elements involved in IPAs at different levels. Actors and institutions are shown in the top half, and the elements usually associated with representations of the physical or natural environment are shown in the bottom half. Figure 14.1 sketches the complexity of the IPA framework.

14.3.1 Actors and Institutions Shaping IPAs

Key institutional actors and processes shaping IPAs include international environmental institutions and frameworks; Australian political bodies—federal, state and territory—and their policies; and the indigenous people pursuing their environmental, cultural and economic interests. The IUCN sets the global framework encompassing principles for supporting indigenous people's involvement in biodiversity conservation (CNPPA/WCMC 1994; Beltrán 2000; Borrini-Feyerabend et al. 2004; Dudley 2008). These principles revolve around the definition of protected areas as *land and/or sea* managed for the conservation of biological diversity and cultural resources *through legal or other effective means,* such as indigenous governance systems (CNPPA/WCMC 1994). Similarly, the global Convention on Biological Diversity (CBD) recognises indigenous peoples' knowledge and use of biodiversity

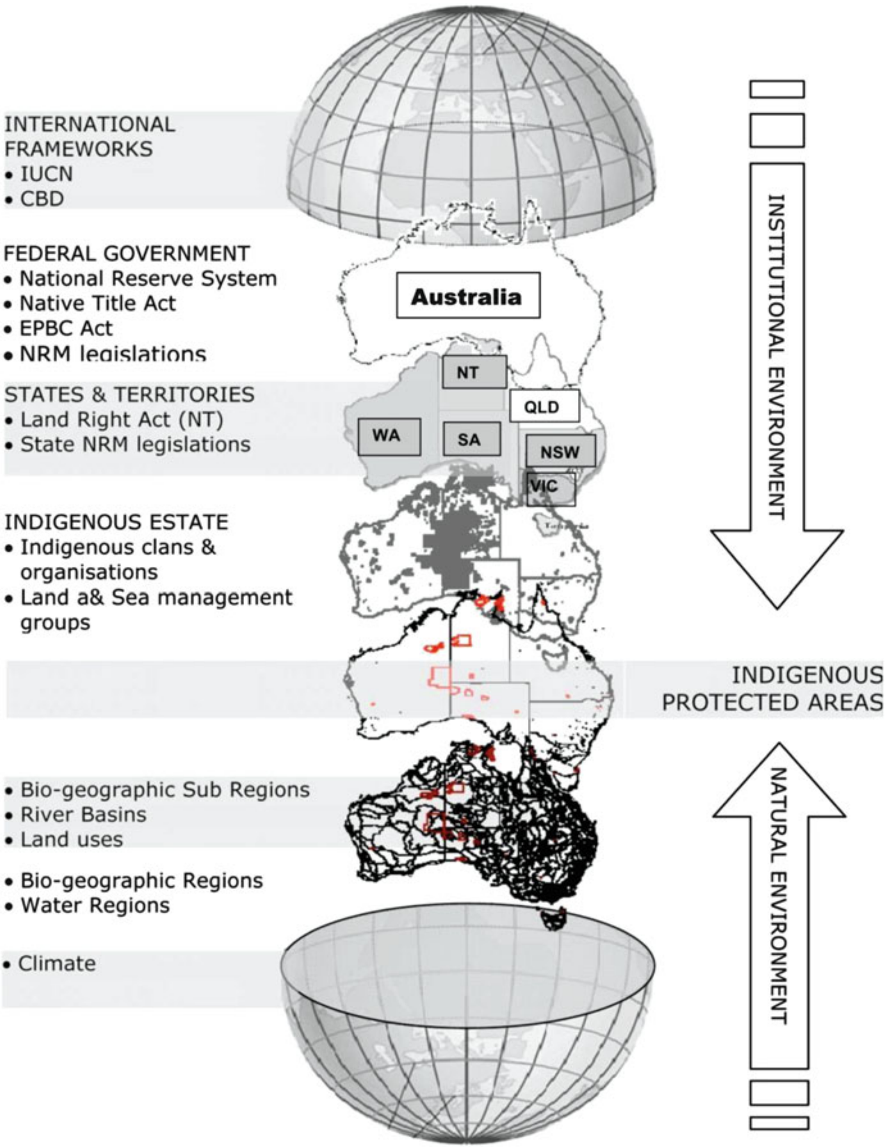

Fig. 14.1 Institutional and ecological scale of Australia's IPAs

(Secretariat of the Convention on Biological Diversity 2011). Article 8(j) specifically commits signatories to the convention to respect, preserve and maintain indigenous knowledge, innovations and practices, as well as to enhance indigenous involvement in the management and protection of biodiversity (Langton et al. 2005).

The Australian federal government is a signatory to the CBD and committed to establishing a comprehensive, adequate and representative system of protected areas according to IUCN guidelines (NRMMC 2005). These protected areas constitute the National Reserve System (NRS). As more than 20% of the Australian landmass is indigenous owned (Altman et al. 2007), Australia's obligations under the CBD required incorporating indigenous-owned land and involving indigenous people in the management and protection of biodiversity. This is the main foundation of the federal government's IPA programme, established in 1997 and refunded in 2008 with AU$50 million to 2013. This programme sits under the national natural resource management framework Caring for Our Country (see Commonwealth of Australia 2010). The IPA programme has allowed for the expansion of the NRS with minimal public expenditure, as the federal government does not need to purchase or lease the indigenous land, as is necessary for *gazetting* national parks.

What we refer to here as indigenous-owned land encompasses a range of tenure systems, rights and interests.[1] At one end of this spectrum, the *Aboriginal Land Rights (Northern Territory) Act 1976* (Cth) (ALRA) grants exclusive possession of or freehold title to land. At the other weaker end, the federal *Native Title Act 1993* (NTA) grants non-exclusive native title rights that do not amount to title over the land (Altman et al. 2007). With the exception of the marine areas in Dhimurru IPA, all the current IPAs have been established on some form of exclusive indigenous title. In recent years, however, the IPA programme has been funding planning for IPAs over multiple tenures, including land not owned by indigenous people as well as sea country.

For indigenous people, IPAs offer a flexible approach to formalise their land and sea management activities. Unlike national parks, IPAs have no legal status, allowing indigenous landowners to retain sole control over their land. Indeed, co-management agreements have sometimes resulted in conflicts over management goals and priorities between government agencies and indigenous landowners (Muller 2003; Woenne-Green et al. 1994). Indigenous landowners set their own plan of management for IPAs, and specific customary law and governance structures guide their land and sea management activities. Their motivation differs from a mere desire for biodiversity conservation. What they term 'caring for country' is motivated by the desire to live on their ancestral lands and seas, safeguard food security and exercise local economic, cultural and political autonomy (Langton et al. 2005). Furthermore, indigenous landowners emphasise the importance of intergenerational knowledge transfer and kinship relations in managing their land and sea country (May and Kerins 2010). Hence, through IPAs, indigenous landowners are pursuing a range of goals: (a) seek recognition of their role as managers of important environmental and cultural resources, (b) access financial resources for their management efforts, (c) increase their employment opportunities, (d) gain recognition of their skills and ecological and cultural knowledge, and (e) support indigenous development aspirations to create culturally and economically sustainable livelihoods

[1] According to Altman et al. (2007), around 30 separate pieces of legislation have been enacted by Commonwealth and state governments over the past 40 years, leading to the recognition, grant, transfer or acquisition of title to land by or for indigenous Australians.

(BAC 2009; Dhimurru 2006, 2008). In summary, it can be said that the international framework has created a platform where the interests of different actors and socio-political processes have converged.

However, some tensions also occur. Some of these tensions revolve around the concepts and practices of environmental conservation and development. Feral buffaloes, for instance, are a key conservation concern in several IPAs (Concu 2011). Such concern is based on the impacts buffaloes have on the ecological integrity of ecosystems, their potential to transmit diseases to domestic animals and the risk to human safety (Albrecht et al. 2009). Culling is considered the most effective way of minimising these impacts. Whilst some indigenous people support buffalo culling, for others, the cultural significance of buffalo, their direct use as a reliable source of fresh food and their potential for economic returns mean that extensive culling is not an option. Tourism development is another example of this mismatch. As many IPAs have outstanding environmental and cultural values, there is potential for ecologically sustainable tourism development. Some indigenous landowners have embraced tourism as an economic development strategy (Muller 2003). However, others are wary of increased numbers of non-indigenous recreational users of their resources and their impacts on cultural and sacred sites (Concu 2011). These examples highlight the diverse views both within indigenous groups and between the indigenous and non-indigenous domains about what form conservation and/or development should take.

Differing perceptions, visions and interests are also present in relation to marine resources (see Yunupingu and Muller 2009). Coastal indigenous people's notions and scalar perceptions of 'country' do not make a distinction between land and sea; for them, they are indivisible (Smyth 2008). Coastal IPA managers continue to assert their 'sea country' rights and their demands for recognition of marine IPAs (BAC 2009). To date, only one IPA officially includes marine areas. Indigenous tenure over the sea is not exclusive, and other actors are involved in the management and use of marine areas.

Recent developments indicate that new arrangements are however emerging. In the Northern Territory, the 2008 High Court ruling on the Blue Mud Bay case (Morphy and Morphy 2009; Northern Territory of Australia v Arnhem Land Aboriginal Land Trust 2008) enables indigenous landowners to exercise the same level of management control over intertidal land, water and marine resources as they currently exercise over the terrestrial components of IPAs. The High Court ruling applies to some 85% of the Northern Territory coastline. At present, the legal, economic and management implications of this decision are still being debated, and indigenous agencies (on behalf of indigenous coastal landowners) and the Northern Territory government are negotiating a long-term arrangement.

Furthermore, funding for sea country management plans from the IPA programme indicates the government's support for indigenous involvement in the conservation of marine resources. The government clearly states that existing laws, regulations and responsibilities would continue to apply in any sea country IPA—including existing bag limits and fisheries management arrangements (SEWPaC 2011). However, sea country IPAs would allow the interests and priorities of IPA managers to be included in the management of marine areas and partnership agreements with stakeholders to be established (SEWPaC 2011). There is also potential future support for marine IPAs at the international level with the development of guidelines for applying IUCN protected area categories to marine protected areas (see IUCN 2010).

The extent of IPA managers' enforcement powers to effectively manage IPAs is also a contentious issue. On one hand, the lack of federal, state or territory legislation regulating IPAs enables indigenous landowners to retain control over their land. On the other, IPAs are private land—even if held by a land trust or corporation on behalf of indigenous landowners—and hence, environmental management is regulated by legislation that applies to any landowner, in addition to indigenous customary laws. Unlike national park managers, who manage land on behalf of the public, IPA managers do not have powers that stem from pursuing the public interest. Hence, they have no capacity to control activities undertaken outside the IPAs even when they have adverse impacts on their protected area.

Three recent policy changes at the national level are also expected to impact on the ability of indigenous landowners to live on and manage their land. The ongoing reform of the Community Development Employment Projects (CDEP) programme[2] has removed the option of accessing new CDEP wages for indigenous managers (or rangers) (Macklin and O'Connor 2008). Whilst the impact on IPAs has been mitigated by the establishment of the federal Working on Country (WoC) programme which funds indigenous rangers' wages, not all ranger jobs have been replaced. Also, the federal government's National Indigenous Reform Agreement (NIRA) is promoting labour migration by withdrawing federal support for service provision in remote locations on indigenous-owned land (COAG 2009). In the Northern Territory, this is compounded by the *Working Future* policy framework (NTG 2010) which focuses service provision on only 20 selected communities or 'growth towns'. These centralising policies could fracture indigenous communities and hence reduce their involvement in the management of their IPAs.

14.3.2 Determinants of IPA Size

Actors and institutions with different powers and capacities play a role in defining the size (or spatial extent) of IPAs. The declared purposes of IPAs, as part of Australia's NRS, include environmental conservation as defined by non-indigenous interests, science and ontologies. Australia's basic units for environmental management and protection are biogeographic regions as defined by the Interim Biogeographic Regionalisation for Australia (IBRA) 2001. IBRA employs a landscape approach to classify land surface in 85 regions and 403 subregions (DEH 2004). Australia's NRS

[2] The federal government set up the CDEP programme as an alternative to social security payments by providing grants to community organisations to employ members in development projects, including land and sea management activities (Altman and Hunter 1996).

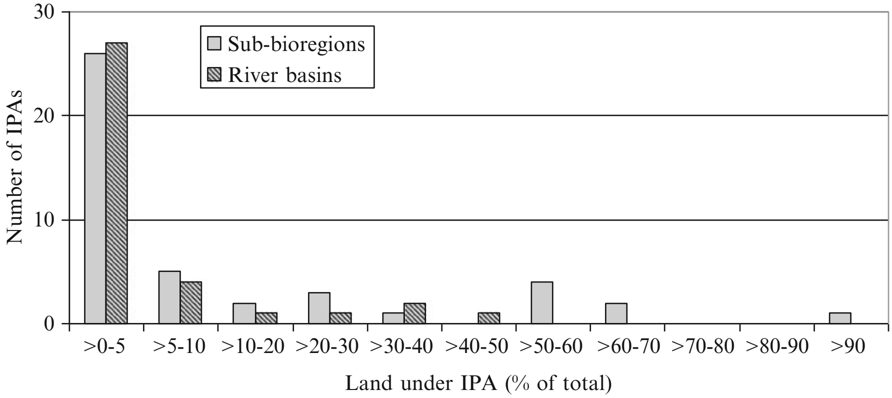

Fig. 14.2 IPAs and ecological boundaries

uses IBRA as the fundamental framework to identify conservation priorities at the national scale. According to Beeton et al. (2006), major environmental issues for the Australian landscapes include fragmentation of ecological communities, mostly driven by human activities such as extensive farming, and degradation of coastal environments due to commercial and recreational fishing, population growth and urbanisation, pollution, mining, tourism and climate change.

Australia's river basins are another important spatial unit for environmental conservation. Australia's surface water resources are hierarchically organised in 12 drainage divisions, 77 water regions and 246 river basins (Geoscience Australia 2004). A river basin includes the total area of a catchment draining into a river mouth. According to the National Land and Water Resource Audit, catchments and basins in the poorest condition occur in areas with high-intensive land use. River basins include several biogeographic subregions. The latter are the appropriate scale for assessing the status of terrestrial ecosystems; the former are used to assess the status of natural resources that affect the conditions of water resources.

Currently IPAs contribute to protect 44 sub-biogeographic regions, and the majority of IPAs protect no more than 5% of these regions (Fig. 14.2). Only the Arnhem Coast Groote sub-biogeographic region is almost entirely under IPA management. Only 36 river basins have land managed through one or more IPA. Again, the majority of IPAs only cover up to 5% of a given basin.

In Fig. 14.3, we show the land allocation in selected sub-biogeographic regions for the states of South Australia, Western Australia and the Northern Territory. Land uses include IPAs, conservation areas (including indigenous-owned land used for customary purposes that is not part of the NRS), grazing in natural vegetation, dryland agriculture, irrigated agriculture and intensive uses—residential, industrial, mining, etc. For this sample of sub-biogeographic regions, it appears that the areas of IPAs and grazing in natural vegetation are negatively correlated. The larger the size of the latter, the smaller the IPA. Productive land uses within each sub-biogeographic region affect the degree of environmental protection. Diverting land from productive

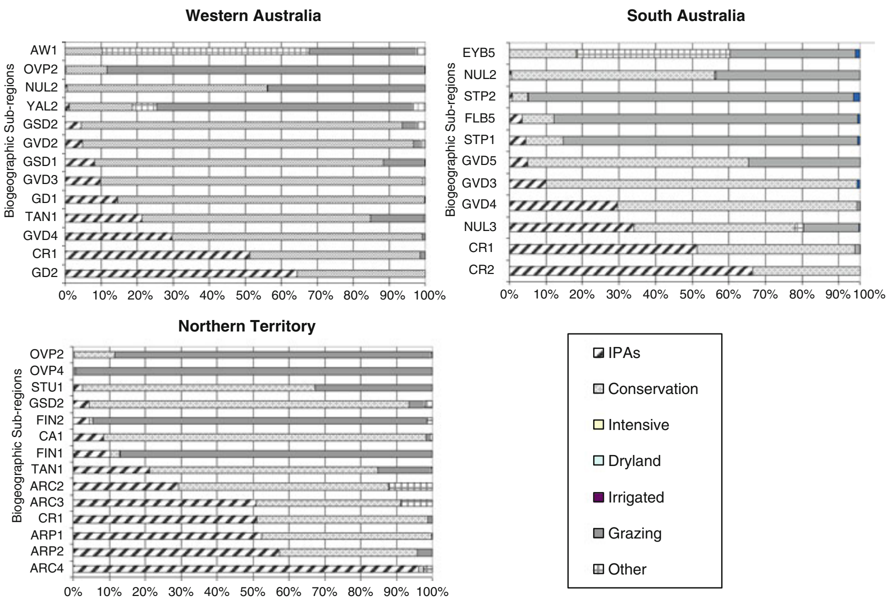

Fig. 14.3 Land uses in selected biogeographic regions and states

uses to environmental conservation has opportunity costs. Therefore, economic factors also influence the role that IPAs play in protecting Australia's biodiversity.

Similar indications emerge from assessing the statistical correlation between ecological factors; land uses—as hectares of land under different management and exploitation regimes—and system of land rights; and the percentage of land under IPA management in each sub-biogeographic region and river basin using regression analysis. Land uses and systems of land rights are proxies of economic, social and institutional factors. One should not expect these proxies to capture the complexity of the institutional arrangement. For subregions, ecological factors include environmental health indicators such as *weed stress, feral animal stress* and *threatened species* (see NHT 2001). For river basins, we use *catchment disturbance index*, *habitat condition*, *hydrological condition* and *water quality* indicators (see NHT 2002). The system of land tenure is represented by a dummy variable indicating if an IPA is regulated by indigenous land rights legislation or not. Several linear regressions were estimated, testing for different specifications of the dependent and independent variables. A simple linear regression provided the best statistical fit. Its results are summarised in Table 14.1.

According to the coefficient estimates in Table 14.2, for the subregion regression model, only three explanatory variables are statistically significant.[3] No environmental

[3] Correlation between variables required us to drop some environmental health indicators and the dummy variables for states and territories.

Table 14.1 Linear regression model for IPA areas in subregions and river basins

Dependent variable				
Area of subregion/river basin under IPA (%)				
	Subregions		*River basins*	
Independent variables	Coefficient	P[T>t]	Coefficient	P[T>t]
Constant	0.02026327***	0.0050	−0.00074264	0.8620
Environmental health indicators				
Stress from weeds = medium	−0.01138802	0.3556		
Stress from weeds = high	−0.01252941	0.8864		
Threatened species (#) = medium	−0.0072363	0.6289		
Threatened species (#) = high	−0.00628226	0.9434		
Catchment disturbance index (= moderately modified)			0.000611757	0.9649
Catchment disturbance index (= severely modified)			0.00625803	0.3973
Catchment disturbance index (= no data)			0.0220416**	0.0191
Land uses (in ha)				
Grazing in natural vegetation	−0.00000071***	0.0091	0.000000642	0.4270
Conservation areas	0.000000049**	0.0324	0.000000127	0.8557
Intensive land uses	−0.00000029	0.9087		
Dryland agriculture	−0.000000061	0.6525		
Irrigation agriculture	−0.00000014	0.9039		
Other uses[a]			−.000001108	0.8641
Security of land tenure				
Land rights legislation = yes	0.05024530 ***	0.0001	0.0292173***	0.0004
Number of observations	403			242
R-squared	0.0860			0.102
Adjusted R-squared	0.0627			0.076
F[10,392](prob)	0.0001		F[4,237] (prob)	0.0006

***Significant at 1%
**Significant at 5%
[a] This class includes intensive uses and dryland and irrigated agriculture

health indicator has a significant impact on the share of IPAs in sub-biogeographic regions. Among the land use variables, grazing in natural vegetation and conservation areas are statistically significant. No other land use seems to affect the share of IPAs in these environmental management units. The area of grazing land is negatively correlated with the percentage of subregional areas under IPA management. This confirms what the graphs in Fig. 14.3 suggest: conservation is in direct competition with pastoralism. The coefficient for conservation areas is also significant but positive. As the land use class ‘Conservation’ includes land managed for customary indigenous use—and this is not considered part of the NRS—it is likely that this land is the one primarily declared an IPA. The positive and statistically significant

Table 14.2 Details of IPA case studies

IPA	Location	Size	Year of establishment	Bioregion	IUCN category[a]	Management organisation	Current annual budget (09/10)	Annual IPA programme funding (09/10)	No. of staff
Dhimurru	North-east Arnhem Land, NT	920 km^2 (terrestrial) 90 km^2 (marine)	2000	Arnhem Coast	V	Dhimurru Aboriginal Corporation	AU$1.8m	A$250,000 A$310,000 (WoC)	22 (14 rangers; 10M, 4F)
Djelk	Central Arnhem Land, NT	6,600 km^2	2009	Arnhem Coast and Arnhem Plateau	VI	Bawinanga Aboriginal Corporation	AU$2m	A$315,000 AU$800,000 (WoC)	36 (34 rangers; 26M, 8F)
Wattleridge	Northern Tablelands, NSW	6.5 km^2	2001	New England Tablelands	VI	Banbai Business Enterprises Inc	AU$190,000	A$165,000	15 (10 rangers; 9M, 1F)
Tarriwa Kurrukun	Northern Tablelands, NSW	9.3 km^2	2009	New England Tablelands	VI	Banbai Business Enterprises Inc	AU$170,000	A$150,000	
Framlingham Forest	South-west Victoria	9 km^2	2009	South-east Coastal Plain	V	Framlingham Aboriginal Trust	AU$80,000	A$80,000	3 (2 P/T rangers)
Deen Maar	South-west Victoria	4.5 km^2	1999	South-east Coastal Plain	VI	Framlingham Aboriginal Trust	AU$110,000	A$110,000	
Tyrendarra	South-west Victoria	2.4 km^2	2003	Victorian Volcanic Plain	VI	Winda-Mara Aboriginal Corporation	AU$130,000	A$130,000	5 rangers (4M, 1F)
Nantawarrina	Central eastern SA	580 km^2	1998	Flinders Lofty Block	II, IV,V, VI	Nipapanha Community Inc.	AU$640,000	A$170,000 A$470,000 (WoC)	11 (9 rangers; 6M, 3F)

[a]For a full description of IUCN protected area categories, see http://www.unep-wcmc.org/protected_areas/categories/index.html

coefficient for the dummy variable representing subregions under indigenous land rights legislation supports this interpretation. It indicates that where indigenous people have more secure and extensive title over land, the share of IPAs in sub-bioregions is larger. In other words, the analysis suggests that IPAs are built over existing indigenous land management institutions and secure property rights, and they are extending the NRS with few opportunity costs. Similar results are obtained in the river basin regression model. Land rights have a strong positive effect on the share of IPAs in river basins, whilst no coefficient for environmental health indicators is statistically significant. The environmental conditions of a river basin have no influence on the area of the basin that is managed as an IPA. In addition, land uses are not statistically significant.

Both regression models have a low explanatory power (R^2 have value of less than 0.1). This is not surprising given the use of a small number of proxy variables capturing institutional elements. It is plausible, for instance, that declaring an IPA is strongly influenced by shifting government priorities, such as the reform of CDEP, as IPAs become instruments to attract scarcer public money for indigenous land and sea management activities. But it is extremely difficult to approximate the shifting nature of the political process through a quantitative variable. Overall, it appears that IPAs are more common where indigenous people have more effective rights to their land. There seems to be no link between the area under IPA management (as percentage of the total area) and the environmental health of sub-biogeographic regions and river basins. Arguably, ongoing tensions among productive uses, indigenous tenure and conservation are more important factors in explaining the size and establishment of IPAs.

14.4 IPA Case Studies

Moving from the national level, we focus on data collected from eight IPAs located in different regions of Australia. This sample was chosen to provide an indication of the diversity of IPAs in terms of location, ecosystem, size, amount of IPA programme funding received and number of staff. Table 14.2 provides a summary of the main characteristics of each IPA.

Djelk and Dhimurru IPAs are the largest areas in our sample. They are located in Arnhem Land in the Northern Territory on Aboriginal land held by the Aboriginal Land Trust under the ALRA. Both areas are managed by Aboriginal corporations, and are among the most well-resourced IPAs. Wattleridge and Tarriwa Kurrukun IPAs in New South Wales are both managed by Banbai Business Enterprise, an Aboriginal organisation, on behalf of the Guyra Local Aboriginal Land Council. The Land Council holds the title of the land under the *NSW Aboriginal Land Rights Act 1983*. Framlingham Forest IPA and Deen Maar IPA are located in south-west Victoria and are both managed by the Framlingham Aboriginal Trust. These IPAs are around 50 km apart and share management and staff. Tyrendarra IPA, also located in south-west Victoria, is owned and managed by the Winda-Mara Aboriginal

Corporation on behalf of the Gunditjmara people. It is the smallest IPA in our sample and covers around 2.5 km^2. Nantawarrina IPA is located in the central eastern region of South Australia. It is jointly owned and managed by the Aboriginal Lands Trust (South Australia) and the Nipapanha Community Inc. under the Aboriginal Lands Trust Act of 1966 (Muller 2003).

There are several common themes emerging from the case studies that illustrate the scalar politics of IPAs. IPAs converge with indigenous aspirations to manage their own cultural and natural resources as well as provide them with the means of pursuing a variety of goals. Whilst these are rooted in the protection of the environmental and cultural values of their land, indigenous landowners specifically state objectives related to empowerment, control and respect for indigenous values (Dhimurru 2006); reoccupation and reinvigoration of 'empty country' (BAC 2009); training and strengthening of the local indigenous population (Hunt 2010); and tourism enterprises as a means of economic development, such as in Nantawarrina IPA (Muller 2003) and Deen Maar IPA (Krishnapillai 2000).

The reasons why, and when, indigenous landowners declare an IPA highlight some conflicting interests and unequal power relations. For example, in the case of Nantawarrina IPA—the first IPA to be declared in Australia in 1998—the Nipapanha Community Inc. decided to declare an IPA after significant pressure from the Upper Flinders Ranges Soil Board to address land management issues in the area (Muller 2003). Previous pastoralism had caused extensive land degradation, exacerbated by exotic weeds and feral animals. This was impacting on the environmental health of the bordering Flinders Ranges National Park. In the absence of other long-term appropriate funding and resources available for land management on indigenous-owned land, the IPA programme offered the only viable option.

In other cases, the declaration of an IPA was driven by a need to mitigate the deleterious impacts of land use practices performed prior to land rights legislation. For example, the area where Deen Maar is located was used for primary production from the late 1800s; wetlands were drained and vegetation removed. Hence, one of the primary foci of the IPA plan of management now is the restoration of the wetland system and revegetation of native species (SEWPaC 2010d). In other IPAs, historical legacies continue to have an impact. Dhimurru IPA surrounds a bauxite mine and processing plant, which was established prior to the ALRA. Dhimurru has no management control over the land within the mining lease area which poses a continuous threat to the IPA. Indeed, the main reason for declaring the Dhimurru IPA was to enable the indigenous landowners to access IPA programme funding to mitigate the threats associated with the mine, including the impact of an increasing non-indigenous population to the area. Indigenous landowners had the option of gazetting their land as a national park, but they decided to declare an IPA instead as it enabled them to manage their land according to indigenous customs and practices. Similarly, Framingham Forest IPA, which is a much smaller 9 km^2, is surrounded by high-intensity dairy farming. Various forms of water extraction and fertiliser use in surrounding properties, as well as encroachment by cattle, have serious impacts on the IPA.

Many IPAs use western scientific tools and techniques such as GIS and surveying or sampling alongside indigenous ecological knowledge systems and techniques

(Ens 2012). Dhimurru rangers work in partnership with the Northern Territory Parks and Wildlife Service (NTPWS) under their own IPA plan of management. Djelk IPA and the contiguous Warddeken IPA share a full-time ecologist employed by the Northern Territory Department of Natural Resources, Environment, The Arts and Sports (NRETAS) Biodiversity Unit, to implement biodiversity monitoring surveys in the IPAs (May et al. 2010). This 'two-way' approach to indigenous land and sea management is being adopted in IPAs throughout Australia.

Financial support from the IPA programme provides core funding for planning, management and monitoring. It enables IPA managers to plan management activities in the medium term and build the governance and institutional capacity of the managing organisation. In our sample, core funding ranges from AU$80,000–315,000 per year, for up to 5 years to 2013 (SEWPaC 2010c). This funding has been particularly important for smaller, less well-established indigenous land and sea management organisations and helps to attract and leverage other funding sources. At Nantawarrina IPA, this funding is facilitating the 'binding' of the community (Simon Duke, former manager, Nantawarrina IPA Personal communication, 4 May 2010).

Nevertheless, IPA programme funding is rarely sufficient to cover all IPA management costs. Hence, many IPAs are supplemented with funding from other government and non-government sources. The majority of IPAs receive funding for rangers' wages through the WoC programme (see Table 14.2). The WoC programme is extremely popular with remote living indigenous people (May 2010), and almost all the 660 positions funded to 2013 have already been allocated (Stalenberg 2010). However, WoC funding does not meet the demand for jobs in some communities. This lack of jobs and funding is preventing some people from moving back onto their ancestral land (Simon Duke, former manager, Nantawarrina IPA, Personal communication, 4 May 2010), and IPA plans of management cannot be implemented. Indeed, Deen Maar and Framlingham Forest IPAs, which cover a combined area of 13.5 km^2, share two part-time WoC-funded rangers; they cannot be expected to achieve significant environmental conservation outcomes.

Some IPAs are also heavily reliant on short-term activity-specific grants from other government programmes or from non-government conservation organisations (May et al. 2010; Dhimurru 2008). Other IPAs have entered into payment for environmental services (PES) schemes. For example, Djelk IPA receives funding through the West Arnhem Land Fire Abatement (WALFA) project (see May et al. 2010), and Dhimurru IPA has a contract with the Australian Quarantine and Inspection Service (AQIS) to sample and monitor mosquitoes, ants and exotic weeds (Concu 2012). Whilst managing a diversity of funding sources is a demanding task, such diversity is seen as offering a degree of resilience in the event of funding loss from one or two sources (Smyth 2008). However, evidence indicates that many of these PES schemes and contracts are short term and still subject to changing policy priorities or economic volatility (see Concu 2012).

The provision of public funding can be a source of tension between short-term and long-term goals. Day-to-day management activities in IPAs are often driven by the need to be accountable to funding bodies, rather than to indigenous stakeholders (see also Muller 2008a). In addition, management of funding usually requires skills acquired through a non-indigenous education. Hence, most IPAs rely on non-indigenous staff for administrative tasks and financial management. On one hand, public funding is necessary for IPAs; on the other, it skews management priorities towards the short-term needs of the funding bodies and away from the long-term aspirations of the indigenous landowners. This limits the importance of indigenous skills and knowledge in the running of the IPA organisations. Furthermore, funding is often targeted to specific environmental conservation activities, whilst many indigenous landowners equally emphasise the importance of cultural practices and the protection of cultural heritage sites. However, it is generally harder to convince funding bodies of the environmental significance of cultural maintenance and protection and vice versa, but some bureaucratic circles recognise the link (Dhimurru, Personal communication, 30 June 2010).

Effective enforcement powers are also an issue. Only Dhimurru IPA and Nantawarrina IPA have permit systems to limit and monitor use and access within the IPAs as prescribed under land rights legislation pre-existing the establishment of the IPAs. Whilst this system provides the IPAs with some revenue, enforcement powers are limited. IPA managers can only report trespassers like any other private landowner. Similarly, Djelk has a contract with Australian customs to carry out coastal surveillance, and whilst their efforts have resulted in a number of convictions for illegal fishing, their powers are limited to surveillance and reporting the incidents (May et al. 2010).

Dhimurru IPA is the only one in the country that has marine areas formally declared part of the IPA (Dhimurru 2008). This encompasses 9,000 ha of marine area, which contains a number of marine sacred sites registered under the Northern Territory *Aboriginal Sacred Sites Act 1989*. At the time the Dhimurru IPA was declared (2000), the cultural significance of the marine areas was considered a sufficient reason to be included in the IPA, even though the Sacred Sites Act provides less authority than the ALRA (Smyth 2007). The plan of management for the Djelk IPA, declared in 2009, remains a draft because it includes management of marine areas. In spite of this, marine management activities such as marine surveillance, biodiversity monitoring and marine debris control continue to take place, supported by funding from government agencies and non-government conservation organisations (see May et al. 2010). Managers of Dhimurru IPA also continue to demand for all the marine areas (not just those containing sacred sites) adjacent to their terrestrial IPA to be included in the IPA. In 2006, they published their Yolŋuwu Moṉuk Gapu Wäŋa Sea Country Plan (Dhimurru 2006) to explain their cultural rights and responsibilities to their marine resources and how they should be valued in the wider context of marine management. Indeed, Dhimurru asserts that ‘This [marine management] plan is an opportunity for us to speak for our sea country in our own way and to do this at a scale that is culturally and geographically appropriate’ (Dhimurru 2006). Dhimurru is also in the process of undertaking systematic conservation planning of Dhimurru Sea Country, combining cultural knowledge and western science with a view to developing a multi-use zoning model to bolster support for their marine IPA declaration and to make indigenous rights and interests in marine resources in this region more visible (Dhimurru 2011).

The locations and sizes of these case study IPAs indicate some disparity with the main priorities for expansion of the NRS (see SEWPAC 2010b). For instance, the three IPAs located in south-west Victoria are some of the smallest of all IPAs across Australia. They are characteristically isolated ecosystems surrounded by grazing land. This conflicts with NRS priorities to focus protection on large areas surrounded by relatively intact ecosystems. It also supports the results of our analysis that in areas of agricultural land use, IPAs are smaller. The locations of Dhimurru IPA and Djelk IPA support the NRS priority of being surrounded by relatively intact ecosystems by respectively bordering the Laynhapuy IPA and Warddeken IPA. However, as such these IBRA regions are already represented in the NRS. Further, not all IPAs include 'largely intact' areas. In the case for Framingham Forest IPA, prior to being declared an IPA, this isolated pocket of forest was almost totally devastated by wildfires. Government's priorities for expanding the NRS may be less significant in determining the establishment of IPAs than other factors outside the initial goals and aspirations of the indigenous people to declare an IPA.

Mining is a potential threat in some IPAs. In Australia, all minerals are owned by the Crown (in right of the State or Territory), and IPA declaration offers no safeguard against mineral extraction. Mining has yet to take place in an IPA in Australia, but it is unlikely to remain that way. In the Djelk IPA, there are mineral deposits such as uranium and bauxite. Exploration licences are constantly being negotiated between the relevant land trust and mining companies. It remains to be seen how the interests of IPA managers, mining companies and the government's conservation agenda will be reconciled on this issue.

14.5 Discussion

The major issues emerging from our analysis show the substance of the scalar politics of IPAs. The first issue is the different conceptualisations of conservation. The interests of indigenous landowners and the government have somewhat converged; the IPA programme supports indigenous aspirations to manage their 'country'. At the same time, Australia has substantially increased the overall size of the NRS with minimal cost and has gained international recognition of its support for indigenous interests. However, tensions arise between the conservation and sociocultural aims of IPAs as well as over long- and short-term objectives: indigenous goals include social, economic and cultural benefits that may not fit non-indigenous conservation priorities. Funding is usually tied to environmental outcomes, whilst the wealth of cultural and social activities carried out by IPA staff remains under-recognised and under-reported. Furthermore, this funding does not seem adequate for undertaking longer-term, robust biodiversity conservation activities or large-scale biodiversity surveys. Indeed, there is limited biological and ecological baseline data on IPAs which hinders effective monitoring of performance against environmental targets (Altman et al. 2001). Our analysis at the national scale that revealed no environmental health indicator has a significant impact on the location and size of IPAs also suggests uncertainty about the environmental outcomes of IPAs.

Second, different notions of, and aspirations for, development are compounded in IPAs. The government's IPA programme is an instrument to assist indigenous landowners to pursue their aspirations to live on their country, their livelihoods and practice and preserve their culture. Yet, such objectives conflict with the goals of other government policies linked to the NIRA and the Working Futures policy. Underlying these policies is the assumption that tangible economic outcomes are a fundamental element of development and that mainstreaming indigenous Australians is the best way of approaching indigenous development. As the case studies suggest, intangible, noneconomic goals such as cultural maintenance are also clearly integral to the development aspirations of many indigenous landowners managing IPAs.

Third, different understandings of the environment underpin the tensions regarding control and access to resources. This generates tensions between indigenous interests in secure land and sea rights and a holistic approach to management and non-traditional uses of the land and sea, such as pastoralism, fishing and mining. As we have shown, nationally IPAs are mostly located in economically marginal areas, and NRS priority areas are not necessarily a significant determining factor. More significant determining factors include indigenous title to the land and the organisational capacity of indigenous groups to manage and protect the area in the long term. If a more strategic, systematic approach to identifying priority areas for inclusion in the NRS is to occur in the future, then this will require additional capacity building for relevant indigenous groups.

Linking these three elements is the constant tension between private and public interests. On one hand, indigenous landowners with declared IPAs retain private title over their land and pursue goals in their interests. On the other, it is the government's responsibility to support IPAs, provided they are in the public interest, which is defined by a suite of goals that include economic, social, cultural and environmental objectives. The government assesses the environmental and cultural aspirations of indigenous landowners through these objectives and decides on the merit of financial and institutional support. Such decisions are undoubtedly compounded by historical, political, cultural, ideological and demographic factors evident in the intercultural space of IPAs. In a number of aspects of IPAs, the public interest largely outweighs the interests of indigenous landowners, such as the objectives, size and location, management strategies, accountability issues, enforcement powers, and funding and resources.

As the number of IPAs in Australia continues to increase each year, indigenous landowners with IPAs need to find a means of continuing to assert their rights and interests so that they are not overshadowed by the public's interest. Indeed, to solely adopt the strategy of demonstrating the public benefits of their land and sea, management practices may be counterproductive. The limited biological and ecological baseline data leaves indigenous landowners vulnerable to suggestions that the environmental benefits of public investments in IPAs cannot be demonstrated (Concu 2011).

An alternative is for IPA managers to coordinate their efforts to influence the concept of public interest and bring it closer to indigenous notions of conservation and development. This would require a better understanding of the values underpinning indigenous and non-indigenous ontologies and an attempt to emphasise common values rather than differences. The idea of establishing a national IPA organisation that could coordinate both these strategies, run by, and lobbying for the rights and interests of IPA managers has been discussed at annual national IPA managers meetings[4] (Dermot Smyth, Personal communication, 21 May 2011), but to date, no such organisation has been established. Alternatively, regional IPA networks could be created to work in alliance with local and regional Aboriginal land councils to support specific interests as IPA managers, such as lobbying for the recognition of sea country IPAs or for increased enforcement powers within IPAs. This is likely to be particularly relevant in areas such as north-east Arnhem Land in the Northern Territory where a number of contiguous IPAs have been established. Regional networks could also be a means of sharing expertise between existing IPAs and advising and supporting new or fledgling IPAs.

In an era of government budget constraints and increasing interest by indigenous people in establishing IPAs, it is entirely possible that in the future IPAs in Australia will be planned, declared and managed by indigenous landowners without the support of the IPA programme, obtaining funding from other sources such as philanthropic organisations or through fee-for-service contracts (Dermot Smyth, Personal communication, 21 May 2011). By asserting their political and financial autonomy and reducing their reliance on public money, indigenous landowners have a means of moderating the public interest as the criteria for assessing their needs, interests and visions in relation to their IPAs.

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[4] These meetings are organised annually by the Australian government's IPA programme.

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Chapter 15
Governing Forests for Provisioning Services: The Example of Honey Production in Southwest Ethiopia

K. Freerk Wiersum and Tefera Belay Endalamaw

15.1 Introduction

Within the field of environmental sciences, usually four categories of environmental services are distinguished, i.e. provisioning services, regulatory services, carrier services and cultural services (Millennium Ecosystem Assessment 2005). In developing new governance systems for effectively maintaining these services, at present much attention is given to innovative programmes for payments of regulatory services such as climate regulation (REDD programmes) and hydrological regulation (water payments). These programmes incorporate the notion of 'making markets work for forest communities' (Scherr et al. 2003) and the idea that environmental payments should contribute to poverty alleviation and local community development (Wunder 2008; Milder et al. 2010). Traditionally, several local practices for payments for environmental services exist; they mainly concern provisioning services (Vedeld et al. 2007). Forests and other nature areas have since long provided people living in or near them a variety of wood and non-wood forest products which were used in their livelihoods. The local use of natural resources does not only include products for subsistence use but also products that are sold as a means to gain a household income. The sale of non-timber forest products forms a good example of a traditional payment system for environmental provisioning services. These payment systems may contribute significantly to the local livelihoods

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(Belcher et al. 2005; Ros-Tonen and Wiersum 2005; Kusters et al. 2006). During the last decade, much attention has been given to stimulate production of non-timber forest products as a means to both forest conservation and poverty alleviation. Within this context, recently specific attention has been given to the nature of NTFP governance systems (Laird et al. 2010). The concept of NTFP governance basically refers to the system of rules that shape the actions of social actors involved in the production and marketing of governance of NTFPs (Laird et al. 2010). It involves the process of standard setting and organisation of measures to regulate access to the resources and to the market (Ros-Tonen and Kusters 2011). This process includes both decision-making and implementation of measures for sustainable exploitation and management of NTFPs as well as decision-making and implementation of regulations and institutional arrangements for access to markets. The NTFP governance systems are complex with multiple centres of authority both at the local community level, where the NTFPs are actually produced, and at the national and international level where NTFP policies and regulations are enacted (Laird et al. 2010; Ros-Tonen and Kusters 2011). For effective NTFP governance, it is important that the national or international governance structures are well related to the local arrangements for NTFP governance. Hence, in developing new arrangements for conservation payments, care should be taken that they complement rather than compete with existing local institutions.

This chapter describes the characteristics of NTFP governance systems at local level and how they are related to governance processes at 'higher' levels. The local governance system primarily involves a set of social and technical practices for organising and controlling the access to the resources and their level of production. The level of production depends not only on the ecological production potential but also on human agency in stimulating production through technical practices. In several cases, local people have stimulated the provision of NTFPs by actively managing and sometimes even enriching valuable forest resources (Wiersum 1997). The different management systems may be subject to different forms of access to the resources, and hence, a local NTFP governance complex may exist. In case that NTFPs do not just serve subsistence purposes but become involved in payment systems, the complexity of NTFP governance arrangements is further increased. In such cases, the governance system does not only concern access to resources but also access to markets. Consequently, the local governance arrangements are becoming impacted by external governance arrangements related to market transactions. Moreover, with the increased interest in better regulation of both forest use and conservation and in effective market organisation, access to both resources and markets is increasingly impacted by government policies and regulations.

The complex governance system for NTFPs is demonstrated in this chapter by means of a case study on honey production in the mountain forests of Southwest Ethiopia. First, it will describe the overall importance of the SW Ethiopian forests for providing a variety of environmental services contributing to local livelihood conditions. Next, it will specify the provisioning role of forest in relation to honey production. Then it will identify the local governance arrangements for honey production as well as the growing interaction with external governance arrangements.

This information will be used to draw conclusions in respect to critical issues in developing new governance programmes for payments of environmental services.

The data presented in this chapter are based on the experiences of an NTFP development programme (Bognetteau et al. 2007) and a series of studies that were carried out within the framework of the programme. These studies included participatory baseline studies and more specialised studies on the role of NTFPs in local livelihood systems (Chilalo and Wiersum 2011) and on the nature of honey production (Endalamaw and Wiersum 2009). The first study consisted of a survey amongst 150 randomly selected households, and the second study a survey amongst 64 randomly selected households and additional focus group interviews. The data of these studies are supplemented by additional literature data.

15.2 Forests and Livelihood Conditions in Southwest Ethiopia

The mountain region in Southwest Ethiopia harbours the largest of the two remaining continuous blocks of relatively undisturbed Afromontane forest vegetation in the country. The highlands cover an altitudinal range from 900 to 2,700 masl and form the upper catchments of several important rivers, such as the Baro and Akobo (tributaries of the Nile) and the Omo. The forests in this region do not only play a major role in water regulation of these rivers but are also of significance for conserving biodiversity. The forests are floristically distinct (Friis 1992; Tamirat 1994) and contain over 107 woody species belonging to 84 genera and 41 families (Kumelachew and Taye 2003). The region is recognised as a biodiversity hotspot of global interest with *Coffea arabica* as a flagship species (Gole et al. 2000). It is reputed as the area of origin for this species, and there is a long history of forest coffee providing an environmental income to local people (Schmitt 2006; Wiersum 2010). Another ancient form of making local use of environmental provisioning services consists of honey production. In contrast to coffee exploitation, which involves one specific forest species only, the traditional beekeeping practices involve the use of a diverse flora (Fichtl and Admassu 1994).

Due to its relatively isolated location, the local communities in the mountain forest region of Southwest Ethiopia have historically been highly dependent on the forest resources for their livelihoods. These communities consisted of people of different ethnic groups. The major indigenous groups are the Sheka (Sheka and Manjo tribes), Majingir, Sheko and, to some extent, Meinit and Bench. Some of these groups (e.g. the Majingir) still adhere to a hunting/gathering lifestyle, but the most populous groups (Sheka) are engaged in mixed farming, including not only agricultural cultivation and animal keeping but also forest exploitation.

During the last decades, the area has become gradually more accessible as a result of improved infrastructure. This has resulted in both immigration and gradual extension of cultivated lands. These dynamics were more prevalent in the mid hills than the uplands, and consequently, a gradual diversification in forest landscape took place. Two main types of forested landscape can be distinguished

Table 15.1 Main characteristics of the two land-use types in the southwest Ethiopian mountain region

	Upland zone (Masha)	Mid-hill zone (Sheko)
Elevation	1,800–2,600 masl	900–1,800 masl
Natural vegetation	Mixed deciduous forest and bamboo forests	Mixed deciduous forests with coffee as a characteristic understorey species
Forest cover	About 50–60%	About 15%
Main ethnic groups	Sheka honey producers and Menjo forest dwellers	Sheko and Bench agriculturalists, Menet and Mejengir hunter/gatherers, immigrant settlers mainly Amhara and Tigre
Land use	Forest use and small-scale subsistence-oriented agriculture	Various types of wild coffee exploitation and coffee cultivation, small-scale agriculture, with some locally marketable products
Average size cropland/household		
Rich households	3.1 ha	9 ha, mainly coffee land
Medium rich households	2.2 ha	4.2 ha, mainly coffee land
Poor households	0.8 ha	0.7 ha, mainly croplands
Average household cash income (US$/year) (n = 150)		
Total	115	209
From NTFPs	48 (41%)	110 (52%)

Source: Bognetteau et al. (2007) and Chilalo and Wiersum (2011)

(Bognetteau et al. 2007), characterised by different forest and land-use types as well as different degrees of forest cover (Table 15.1). In the mid-hill area, mixed deciduous forests occur with coffee as a characteristic understorey crop. The collection of forest coffee has gradually been supplemented by coffee cultivation. The need for smallholder agricultural lands for food production and the establishment of commercial coffee plantations by external investors has resulted in fragmentation of the forest cover. At higher altitudes, coffee cultivation is less common, and deciduous forests are complemented by bamboo forests. In these uplands, the forest cover is much higher than in the mid hills, and most agriculture is subsistence oriented (Fig. 15.1).

Both in the upland zone and the mid hills, local communities are still highly dependent on the forests for their livelihoods. Besides wood for construction and fuel, several non-wood forest products are collected; they are used for a variety of purposes such as food and condiments, fodder, binding materials and medicine (Bognetteau et al. 2007; Chilalo and Wiersum 2011). While a large number of the NTFPs are used for subsistence purposes, a more limited range is also traded in order to provide income. In the uplands, besides honey and some forest coffee, also bamboo and spices (*Korerima* – Ethiopian cardamom; *Timiz* – long pepper) are commercially exploited. In the mid hills, forest coffee is the most important NTFP, followed by fruits and spices. The NTFPs contribute significantly to the household

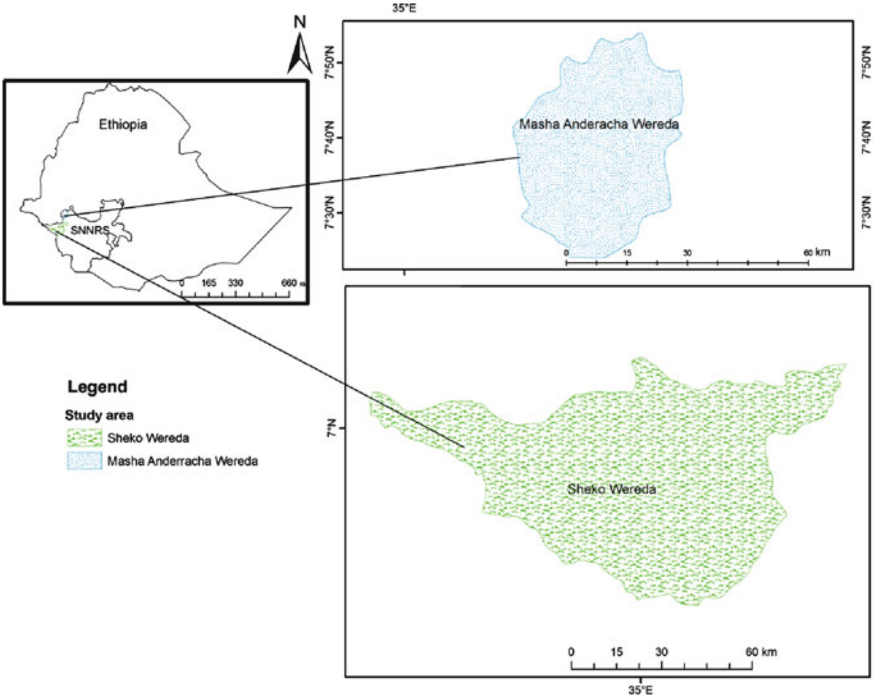

Fig. 15.1 Map showing the location of the mid-hill (Sheko) and upland (Masha) study regions

incomes. In the uplands, NTFPs provide 24% of the mean total household income or 41% of the mean household cash income and, in the mid hills, even 30 and 52%, respectively (Bognetteau et al. 2007; Chilalo and Wiersum 2011). In the mid hills, not only the relative contribution of NTFPs to household income is higher than in the uplands but also the absolute amount (US$110/year vs. 48/year) (Chilalo and Wiersum 2011).

The greater importance of NTFPs in the mid hills is related to the great value of forest coffee. Whereas in the mid hill 69% of the households are engaged in forest coffee production and 24% in honey production, in the uplands this concerns 13 and 73% of the households, respectively. The average household income of the actual forest coffee and honey producers is US$139 versus US$45 in the mid hills and US$72 versus 52 in the uplands (Chilalo and Wiersum 2011). The importance of forest coffee in the mid hills is not only related to the optimal environmental conditions for forest coffee in this region but also to relatively well-developed marketing system (Wiersum et al. 2008). In contrast, the production, processing and marketing conditions for honey are still poorly developed. Due to the traditional production and processing techniques, the quality of the honey is low, and this limits marketing beyond the local markets (Bognetteau et al. 2007). Consequently, this product is of most importance in the uplands where few alternative commercial NTFPs can be exploited. Whereas in the mid hills, rich and poor households own

Table 15.2 Main traditional governance arrangements for managing forest resources in southwest Ethiopian mountain forests

	Upland zone	Mid-hill zone
Natural forest systems	Maintenance of religious forests as common property resource delineation of *Kobo* forest blocks for individual honey production	Community-controlled wild coffee forest privately owned semi-natural coffee forests
Converted forest systems	Individual *Kobo* rights on tree on communal lands for hanging bee hives	Privately owned mixed coffee gardens

Source: Bognetteau et al. (2007)

23 and 2 beehives, respectively, in the uplands they own an average of about 100 beehives and 50 beehives, respectively (Bognetteau et al. 2007).

The local people do not only value the forests for their environmental provisioning services but also for providing several regulatory and carrier services, such as the regulation of hydrological processes and water supply and the provision of good microclimate conditions (provision of shade and reduction of heat) and the maintenance of soil fertility. The forests also provide cultural services: Some forests have religious and spiritual values, many people enjoy the aesthetics of forests and trees and forests may also be appreciated as an important resource ensuring the future of the children (Bognetteau et al. 2007).

15.2.1 Traditional Forest Governance Arrangements

The important role of different forest environmental services in local livelihoods, notably those related to the provisioning of non-timber forest products, has resulted in the development of a variety of traditional governance arrangements for conserving and managing the forests (Table 15.2). Some forests are conserved in their natural conditions as sacred forests or for the production of either wild coffee or honey. Other forests may be gradually modified in order to increase their provisioning services. This process of gradual adaptation of forests in order to increase their provisioning service is well expressed in respect of coffee production. Although forest coffee is still collected in some natural forests, other forests have been structurally adapted into semi-natural coffee forests through the stimulation of coffee production (e.g. by slashing competing vegetation and stimulating increased sprouting of coffee plants) or transformed into mixed coffee gardens in which coffee is actively propagated together with other useful species (Schmitt 2006; Wiersum 2010). Consequently, the natural forests have gradually been transformed into a forested landscape consisting of a mosaic of natural and anthropogenic forest types. This development reflects the local agency in both conserving forests for environmental services and stimulating provisioning services in respect to NTFPs. The change from natural forests to adapted forests did not only involve a change in the local norms regarding forest exploitation and management intensity but also a change in the rules regarding resource access. Whereas the extraction of products from natural forests may be based on common property arrangements, the more intensive management in adapted forest types is based on private forest access or even land-use rights (Bognetteau et al. 2007; Wiersum 2010).

15.3 The Role of Forests in Honey Production

Due to the importance of the Southwest Ethiopian forests as the area of origin of coffee, several studies have highlighted the local importance of forest coffee as a non-timber forest product (Gole et al. 2000; Schmitt 2006; Wiersum 2010) and the related governance arrangements (Gatzweiler 2005, 2006; Wiersum et al. 2008). Much less attention has yet been given to the other NTFPs, including honey as the second most important NTFP. In the following, first the role of forests in honey production will be described. In the next sections, the local governance arrangements in respect to its production and the growing importance of external governance arrangements will be identified.

The production of honey involves an intricate set of forest-bee interactions with trees having multiple roles in honey production (Crane 1990; Svensson 1991; Hill and Webster 1995). They are a main source of bee forage and provided traditionally a nesting place for bee colonies. The role of forests was further diversified when beekeeping hives were introduced. In such cases, trees do not only provide raw materials for hive production but also provide space for hanging hives. Moreover, vegetative material is also used for smoking and fumigation of hives. In addition, trees also provide shelter and protect bees from adverse climatic conditions, e.g. by moderation of temperature extremes through shading, and reduce susceptibility to pests and vermin. The different services provided by trees for honey production are well recognised by the local people. This is demonstrated by their preferences for different kinds of trees for different beekeeping purposes (Table 15.3). Traditional people define a tree as a preferred bee forage tree by its attractive and melliferous flowers. The quality criteria for hanging hives include presence of multiple branches of dependable strength for carrying hives higher up the tree crown and for providing good standing space for beekeepers while fastening hives or harvesting honey. As good hanging trees are relatively sparse, they are higher valued than forage trees. For making the traditional hollow-log beehives, both the ease of woodworking and durability are factors influencing the choice of trees used. In the past, also barks and twigs were used for hive construction, but at present, only roundwoods are used. For hanging the beehives, climbers/lianas are used; the flowers of these plant species may also provide bee fodder. Beehives are preferably located near good bee forage resources. Such baiting and good hive quality are considered as important factors determining whether a hive will attract bees and become colonised. In order to

Table 15.3 Tree species preferred for different beekeeping purposes (in order of preference)

Pollen and nectar source	Hive placement	Hive fumigation	Hive construction
Schefflera abyssinica	*Aningeria adolfi-friederici*	*Ekebergia capensis*	*Euphorbia abyssinica*
Ficus thonningii	*Ficus sur*	*Piper capense*	*Ficus spp.*
Apodytes dimidiata	*Prunus africana*	*Clausena anisata*	*Aningeria spp.*
Manilkara butugi	*Polyscias fulva*	*Olea spp.*	*Euphorbia abyssinica*
Ekebergia capensis	*Ficus thonningii*	*Cyathea manniana*	*Cordia africana*
Celtis africana	*Croton macrostachyus*	*Vernonia spp.*	*Croton macrostachyus*
Vernonia spp.	*Ekebergia capensis*	*Trichilia dregeana*	*Arundinaria alpina*
Croton macrostachyus	*Manilkara butugi*	*Ekebergia capensis*	*Polyscias fulva*
Cordia africana	*Albizia spp.*	*Eucalyptus spp.*	*Celtis africana*
Aningeria spp.	*Aningeria spp.*	*Maesa lanceolata*	
Allophylus abyssinicus	*Milicia excelsa*		
Albizia schimperiana	*Bridelia micrantha*		
Maesa lanceolata	*Celtis africana*		
Olea welwitschii	*Olea welwitschii*		

collect the honey from the hives, they are often fumigated; also for this activity, local knowledge is used in selecting species to be used in fumigation.

The forest-bee interactions do not only include the services provided by different plant species for honey production but also involve the pollination of flowers by bees. One major species profiting from such pollination is coffee. Hence, from an ecological point of view, forest coffee production and honey production are synergetic.

15.4 Local Governance Arrangements for Honey Production

As demonstrated by the use of beehives, honey production does not only involve the collection of wild forest products but also include several specific management practices. Three main categories of local practices for honey production can be distinguished: social controls for regulating access to honey production resources, technical practices for managing bee colonies and technical practices for managing the forest/tree vegetation.

15.4.1 Social Controls for Regulating Access to Honey Production Resources

Originally, honey was collected as an open access resource from wild bee colonies nesting in trees in the natural forests. Traditionally, such honey gathering was mainly done by hunting/gatherer groups belonging to the Majenger and Menet ethnic groups (Stauder 1971). Gradually, also the other ethnic groups practising agriculture became attracted to honey production. Rather than extracting honey from wild nests, they use beehives. At present, most honey is produced in traditional beehives that are hung in forests trees. This form of honey production is controlled by the presence of local regulations regarding access and use rights (locally called *kobo* rights) to either forest plots or specific trees for hanging of the beehives (Wakjira and Gole 2007; Endalamaw and Wiersum 2009). These regulations started in the late nineteenth century when landlords awarded forest blocks of some 40 ha of the communal forests to local inhabitants in order to regulate access to trees for hanging beehives. Although this *kobo* system was initially not officially recognised by the former government, it has in many places survived, even when the forests became nationalised. At present, *kobo* blocks have been integrated in the official national forest estate (Bognetteau et al. 2007; Wakjira and Gole 2007). Gradually, the *kobo* system of allotment of individual forest plots for hanging beehives has been extended to individual trees. Such *kobo* trees may grow on state-owned forest lands, communally owned lands, private agricultural plots or privately owned tree gardens. *Kobo* rights on trees are more widespread than *kobo* rights on forests. In a local survey, it was found that 34% of all farmers still actively preserve *kobo* forests and that 79% actively protect *kobo* trees (Endalamaw and Wiersum 2009). The protection of the *kobo* forests includes not only adherence to the local rights on forest and tree use but also measures for controlling bush fires, especially during beehive fumigation in connection with honey harvesting. The *kobo* arrangements also include arrangements for local solving of beekeeping-related conflicts. Although the *kobo* use rights have at present been recognised by the government, they are not supported legally or by formal institutions (Wakjira and Gole 2007). Consequently, local disputes regarding honey tree or forest ownership are still put before a clan leader or a group of elders, who facilitate dialogue and contribute to conflict resolution on the basis of traditional system for adjudication of local rules.

As illustrated by the gradual recognition of the government of the *kobo* system, this traditional governance arrangement for forest resource is gradually supplemented by formal forest governance arrangements. In the survey on honey production, two-thirds of the respondents reacted positively with 'yes' to the question of whether the traditional local arrangements for access to honey production resources will be continued by the next generation. Other respondents mentioned several reasons why the arrangements might be discontinued. The major reasons were the allocation of the land for private entrepreneurs by the government, increasing periods of abandonment of *kobo* trees or forest lands without using them for honey production, and absence of male heir.

15.4.2 Technical Measures for Managing Bees

As demonstrated by the change from wild honey collection to beekeeping in hives, beekeeping involves several technical measures for managing bees. In the survey about honey production, 33% of all respondents reported that hives were placed in natural forests, 48% in *kobo* forests and 19% in home gardens and agricultural lands. In another survey amongst 120 households, it was found that 12.5% of the honey producers were involved in wild honey collection and/or forest beekeeping, 27.5% in forest and farmyard beekeeping and 60% in farmyard beekeeping (Solomon 2009). Recently new techniques for honey production using modern beehives and improved quality control were introduced as a means to increase honey production. These innovations were introduced by external development organisations and demonstrate the increasing importance of external organisations in stimulating honey production. In initial on-farm trials, these technical innovations increased honey production by 150% (Bognetteau et al. 2007). But these new techniques are still little used; in the honey production survey, they were found to be used by only 6% of the respondents. The modern techniques require good oversight, and the modern beehives are therefore mainly located in the farmyards near the owner's house. Thus, the introduction of these techniques strengthens the trend of gradual transfer of beekeeping from the forests to other land-use zones. However, this does not mean that forests lose their role for honey production; they are still considered as important sources for new bee colonies and bee forage.

15.4.3 Technical Practices for Managing the Forest/Tree Vegetation

As demonstrated by the forest and tree conservation practices under the *kobo* arrangements, farmers undertake conscious practices for managing the forest and tree resources of importance to honey production. In a recent survey amongst 64 randomly selected households, only 3% of the respondents mentioned that they do not undertake vegetation management practices for stimulating honey production. These practices do not only involve the protection measures mentioned earlier but also measures to tend trees and stimulate their regeneration. In the survey, 50% of the respondents mentioned that they consciously retained and tended saplings and seedlings, and 20% mentioned planting seedlings of desired trees. Such planting often regarded the planting of indigenous tree species such as *Prunus*, *Polyscias* and *Olea* as a means of ensuring the future availability of good hive-hanging trees. Other reasons for tree planting, but of lesser importance, were to ensure future wood supply for hive making or the provision of forage species. These tree planting practices are a response to both loss of forests due to agricultural extension and the gradual transfer of beekeeping from the forests to anthropogenic land-use types near settlements. But notwithstanding these trends, 74% of the respondents believed that beekeeping is dependent upon forests and that it contributes to forest conservation (Table 15.4).

Table 15.4 Local opinions ($n=64$) about why beekeeping assists in forest conservation

Reasons	% of times mentioned
Forest conservation is essential to sustain beekeeping practice	45.3
Beekeeping is dependent upon forest trees and flowers	20.3
Beekeepers protect their forest plots from tree felling by other people	20.3
Beekeepers refrain from tree felling	6.3
Beekeeping contributes towards the detection and control of forest fires	4.7
Forests are a source for providing bees	3.1
Total	100

15.5 Growing Interaction with External Governance Arrangements

As illustrated by the descriptions of the nature and dynamics of the local arrangements for NTFP governance, during the last decade, the traditional local structures for managing honey production have been the subject of external influences. Three different types of such external governance arrangements can be distinguished: government regulations, market arrangements and development standards of non-governmental organisations. They concern either new forms of access to the resources, production standards or access to the market.

Regarding government regulations, two main governance arrangements influence honey production and trade. As discussed earlier, the local tenure structure are gradually becoming supplemented by formal systems for forest ownership. The government policies on forest ownership have been quite dynamic. Whereas the initial formal forest tenure focused on gazettment of state forest reserves, gradually more attention is given towards participatory forms of forest management and formulation of community-based regulations on local forest use (Zewdie 2005; Bognetteau et al. 2007; Gobeze et al. 2009; Solomon 2009). In SW Ethiopia, these new policies mostly focused on the management of coffee forests (Gole et al. 2000; Gatzweiler 2005), but gradually the relevance of the traditional *kobo* rights is becoming formally recognised. As discussed above, at the local level, these rights are still adhered too. A second major government regulation shaping the governance of honey concerns the policy of stimulating local cooperatives for improving the product quality management and marketing of agricultural produce. Such governance arrangements for stimulating access of small producers to markets are implemented independently of participatory forest management (Wiersum et al. 2008). The lack of coordination between different government innovations for involving local communities in forest management and in marketing of forest products indicates a lack of an integrated approach towards developing external arrangements for NTFP governance.

As illustrated by the role of external development organisations in introducing modern honey production practices, some non-governmental development organisations are trying to stimulate the honey production and marketing. These external development initiatives concern the introduction of modern standards for honey production and assistance with creating new, higher-value markets (Bognetteau et al. 2007). As a result of the traditional production and local processing techniques, the productivity of traditional beekeeping and the quality of the honey are low. This limits the interest of national trading companies in buying the local honey and limits honey sales to the less financially rewarding informal market. The marketing of honey is further hindered by the lack of organisation of producers. This restricts access to market information and makes farmers highly dependent on price setting by local traders, giving them marginal economic returns for their products. As discussed above, several efforts have been undertaken by NGOs to stimulate product quality; these development efforts also included training in better dealing with principles of market governance. Locally they resulted in producer prices for crude honey to be increased by 200–250% (Bognetteau et al. 2007).

Recently, a further initiative in creating novel standards for marketing of honey was undertaken by developing a partnership between a socially responsible commercial enterprise and the local producers and to market the honey as a regional specialty with an area-of-origin guarantee (Bognetteau et al. 2007). This governance innovation resembles recent efforts at stimulating better forest governance through certification systems. This option for novel arrangements for NTFP governance has been pioneered in Ethiopia in connection with forest coffee (Gatzweiler 2005, 2006; Wiersum et al. 2008). Several questions have arisen in respect of how best to organise such a certification system. An important question that emerged was whether the certification standards should be focused on forest coffee or on ecologically sustainably produced coffee (thus including garden coffee cultivation) (Wiersum et al. 2008). These different options on certification standards are reflected by the differences in approach of forest coffee certification and the initiatives for honey labelling. These two processes proceeded rather independently and involved organisations stimulating either sustainable forest management or socially and ecologically sustainable ‘agricultural’ production, respectively. Consequently, the question arose of whether the present diverse approach to certification of specific NTFPs could not be replaced by an area-based approach. Such an approach could be focused on the sustainable management of forested landscapes involving both natural forests and modified (agro)forestry systems (Wiersum et al. 2008). Such an approach offers a good opportunity to develop a specific standard for multiple forest products and services. Such a landscape labelling approach (Ghazoul et al. 2009) would not only solve the problems of having to develop separate certification systems for different NTFPs (such as in our case coffee, honey and possibly also spices) but would also allow to incorporate the new programmes for payment of environmental regulation services.

15.6 Discussion and Conclusion

One of the oldest forms of payments for environmental services concerns the provisioning services. In many local communities, forests contribute significantly to local livelihoods through the provision of not only subsistence but also commercial products. As demonstrated by the example of honey production, this has resulted in the development of complex system of local governance arrangements. The presence of such location-specific and dynamic systems of local management of environmental services deserves explicit attention when developing novel arrangements for environmental governance at national and international level. Two issues are of major importance: (a) The local governance complex are primarily livelihood rather environmental conservation-oriented and concern forested landscapes with a mix of natural and adapted forest types; (b) new governance arrangements from external organisations should complement and build up on local governance arrangements rather than compete with them.

Traditionally, many communities have actively governed their forests in order to optimise the contribution of forests to their local livelihoods. As demonstrated by the example of honey production, the local governance structures concerned a complex system on rules on access to resources on both communal and private lands. They included management practices for conserving forests and enhancing valued forest resources and enable a gradual intensification from wild honey collection to conscious management of both bees and trees. As demonstrated not only by our data from SW Ethiopia but also by data from other regions of Ethiopia (Solomon 2009) and East Africa (Fischer 1993), the intensification of honey production did not only regard bee management but also forest conservation and vegetation management. The first phase of intensification involves the formulation of social controls on access to the forest resources which support honey production. Subsequently, measures may be taken to manage the required tree and bee resources. Such intensification often requires better control over the bee resources. Consequently, the honey production process is gradually shifting from natural forests to adapted (agro)forestry systems in the neighbourhood of human settlements. In the mountain forest area of Southwest Ethiopia, a similar trend in domestication and gradual intensification in management occurred in respect to forest coffee (Schmitt 2006; Wiersum 2010) and yams (*Dioscorea cayenensis*) (Hildebrand 2003). All these examples illustrate the dynamic nature of local processes for governing the most valuable provisioning services of forests. The traditional governance arrangements for ecological provisioning services resulted in a co-evolution in forest ecological conditions and socio-economic conditions and the development of a mosaic of natural forests and adapted forest-analogue vegetation types. This forested landscape mosaic provides optimal conditions for local people to profit from the multiple environmental services offered by forests and for an optimal combination of forest conservation and rural livelihood conditions. Novel arrangements for governing environmental services should carefully consider the merits of such a landscape approach in comparison to a more restricted ecosystem approach (e.g. Ghazoul et al. 2009).

The introduction of the new management and marketing arrangements by external organisations indicates that the traditional local system governing the provision of environmental services is gradually becoming connected to external governance arrangements. These do not only concern the conservation and management of the environmental services but also their marketing. The introduction of the new governance arrangements by the government and external development organisations should support rather than compete with the traditional local governance. Several efforts to stimulate community-based forest management and local institutions for the governance of forests are undertaken in Southwest Ethiopia (Bognetteau et al. 2007; Gobeze et al. 2009). However, unfortunately, many policies and development measures disregard the traditional settings for providing provisioning services to local people. Examples are the gazettment of state forest reserves without considering how to administer local *kobo* rights and the provision of land-use rights to investors for establishing commercial coffee plantations without considering the development potentials of the endogenously evolved coffee production systems. These examples demonstrate how a mismatch between external and traditional forms of governance will create conflicts with regards to both rules and regulations and governance authority. This will have counterproductive effects on the conservation of the forest and the role of forest services for local livelihoods.

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Chapter 16
Investing in Sustainable Use of Biodiversity for Social Benefit in Brazil

Peter H. May and Valéria da Vinha

16.1 Brazil's Role in Biodiversity Conservation: "Use It or Lose It"

Brazil is a "megadiverse" country, sheltering within its borders from 15 to 20% of all known species on the planet (Conservation International 2005). Megadiversity harbours great wealth and opportunity, but also major responsibilities. This chapter concerns itself with the importance of biodiversity and related ecosystem services to the long-term productivity of Brazil's agricultural and natural resource industries, as well as the potential benefits that biodiversity-based enterprise may provide as part of national development strategies.

Biodiversity and related ecosystem services (as defined in the Convention on Biodiversity (CBD)) embody intra- and interspecies genetic variability, the very stuff of life and evolution. Biological diversity and native ecosystems such as wetlands cushion the impact of extreme climatic events, which are occurring with ever greater frequency. According to the Millennium Ecosystem Assessment (MA 2005), human occupation has decimated a good share of remaining biodiversity and threatens

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much of what remains. The CDB promotes both conservation and "sustainable use" of biodiversity and its components. But how can human society "utilize" biodiversity without destroying it in the process?

Emblematically, the history of Brazilian land occupation has been linked most clearly not to conservation but rather to destruction of biological resources for survival and domination. Exploiting and subjugating nature are synonymous with Brazil's territorial expansion; the resources exploited are little valued and have been squandered and then replaced with introduced or invasive species. Knowledge of native biological diversity and useful provision of goods and services have been progressively eroded in the name of progress.

The majority of raw materials that Brazil relies upon and exports for food (soy, coffee, Zebu cattle, African oil palm), construction, paper and biomass (sugar cane, eucalyptus, pine) are of exotic origin. Yet some of the world's most important tropical crops originated in Brazil, including cassava, peanuts, pineapple, cashew, cocoa and others. The germplasm of origin for those species depends on collection in the wild so as to assure maintenance of productivity and stability. Doubtless, the endemic flora and fauna still guard countless secrets with potential economic and social value.

Despite Brazil's historical aversion for its own native species, it is widely recognized that they represent potential new products; knowledge for pharmaceuticals, medicines and natural vitamins (herbal and personal care); genetic basis for cultivated plant resources and biotechnology (enzymes, microorganisms); and ornamental plants and substances used for natural plant protection, and foods are some of the main products that have been derived from genetic resources. The same resources are additionally responsible for generating services of inestimable national and global importance. Without native pollinators, for example, a large number of food plants could not reproduce.

From an ecological perspective, biological diversity is composed of organisms whose variability, derived from genetic evolution and their niche in ecosystem structures, enables their populations to achieve stability and compete for resources with other organisms. The UN Framework Convention on Biological Diversity (CBD) treats the potential for use of the distinct components of biological diversity by human societies, whether directly for satisfying immediate needs or indirectly through non-use conservation seeking indirect or long-term benefits (e.g. generation of environmental services or potential for curing human diseases or overcoming vulnerabilities in cultivars).

The components of the national genetic heritage are legally defined in Brazil (by Provisional Measure 2186-16/2001) as

> information of a genetic origin, contained in samples of all or part of a vegetable, fungal, microbial or animal specimen, in the form of molecules and substances derived from the metabolisms of these living creatures and from extracts obtained from these living or dead organisms, found in conditions in situ, including domesticated species, or maintained in collections ex situ, as long as collected in conditions in situ in the national territory, the continental shelf or the exclusive economic zone.[1]

[1] MP 2186-16/2001 regulates the National Constitution and the CBD, applied to access to components of Brazil's genetic heritage (translation by the authors).

This broad definition refers to organisms or extracts, seeds or strains and compounds or molecules, originating in the national territory, including marine areas in the national domain. Utilization may imply extracting active principles from the molecules, such as those that the organisms have developed for self-defence, or identification of medicinal uses derived from the knowledge of traditional societies, which can enable a reduction in the costs of pharmaceutical discoveries, thus adding value to the standing forest (Balick et al. 1996).

The stimulus for sustainable use of biodiversity comes from a widely disseminated perception in the 1980s and 1990s according to which conservation of the natural heritage could be best assured through wise use rather than by actions strictly directed towards preservation (prohibition, sanctuaries). The motto of the movement favouring sustainable use came to be "use it or lose it."[2] This concern contrasts with positions contrary to the presence of human beings in critical areas, assuming that any disturbance would lead to an irreversible loss for the evolutionary process (Durojeanni and Pádua 2001). On the other hand, observers point out that there are few wild areas left on Earth where human presence, either primitive or modern, has not manipulated the environment in some way, whether to maintain or introduce species perceived as more "useful," allowing them to remain or even to dominate the landscape (Diegues 2000).

The perspective of sustainable use as an ally of conservation brings together two distinct lines of thought that serve as arguments for its inclusion as a core component of public policies directed towards development and the environment:

- *Conserve to use* – maintaining species diversity assures productivity of the other organisms in the landscape, including those introduced by humans.
- *Use to conserve* – low-impact use allows maintenance of species diversity and ecosystem productivity in the long term.

The first approach refers to valuing natural ecosystems as integral parts of the productive landscape, due to their positive feedback into agriculture, ranching and forest production systems, providing services such as water of the quality and quantity necessary for irrigation and drinking and habitat for pollinators essential for fruiting and fertilization. The second refers to attempts to draw up good practices for natural resource management, such as low-impact logging or institutional schemes for restricting access and reducing pressure during the reproductive phase of native species, for example, local fishing agreements (Clement et al. 2007).

This chapter concerns itself with identifying the potential for sustainable use of biodiversity in Brazil, as well as its limitations, as a basis for social investment. Our focus here is upon social or community enterprises involving low-income groups that depend for their livelihood on the stability of ecosystems that shelter components

[2] This concept is also based on two connected areas: evolutionary biology considers that if an organism or ecosystem does not depend upon a given characteristic for its survival, it is fated to disappear from the evolutionary process. It is also used to refer to intellectual property rights, such as over geographic designation of origin. If the origin is not formally declared, there is a risk of loss of the right to exclusive use of the product name or quality, and it becomes generic.

of Brazilian biodiversity. We give particular attention to the potentials and pitfalls of industrial development based on sustainable utilization of forest resources for agro-extractivism, as one of the means by which resource-dependent households can benefit from expanded use of biodiversity while conserving these resources for ecosystem services of benefit to the broader society.[3]

16.2 Conditions for Sustainable Agroextractivism

A sizable portion of the products derived from native biodiversity use are goods in common use, lacking formal markets; there are in consequence no reliable data on their production, commerce and use as intermediary inputs. One of the rare sources of information, the National PLan for Promoting Socio-environmental Product Chains (*Plano Nacional de Promoção de Cadeias da Sociobiodiversidade*) (Brazil 2009), estimated that the non-timber forest product sector (NTFP) reached only R$480 million (about US$200 million at the current exchange rate at time of estimate) – a quantity sufficient to provide for a minimal monthly salary for only around 90,000 people, yet it is estimated that several hundred thousand households rely on these resources for their livelihoods. Furthermore, their value added to local and regional economies is considerable. (A recent assessment of production chains associated with açai palm products, point to total value added of as much as $1.4 billion from this plant alone.)

Activities associated with "plant extractivism" have been on the decline in recent decades (Homma 2010). Such activities have been progressively replaced by others more profitable per unit area, market intelligence and wider commercial networks. Their persistence is due to structural and situational factors: (1) continued existence of pockets of absolute poverty in rural areas; (2) complementarity between family production and plant extractivism activities in places where there is an abundance of native species with functioning markets in place (known as "agroextractivism"); and (3) attempts to add value to niche chains associated with conservation and sustainable attributes. The major motivation is undoubtedly the persistence of poverty in tandem with plant genetic resource occurrence, since the number of undertakings in "sustainable" business based on NTFP and their relatively limited success means that the audience reached is insignificant in relation to officially recorded production.

Although the domestic GDP does not reflect the importance of goods and services generated by biodiversity, the income and well-being of groups dependent on nature are strongly determined by opportunities created for their sustainable use. The expression "GDP of the poor" (Sukhdev and Gundimeda 2008), referring to the income of those living mainly from small-scale production, animal raising, informal

[3] The study on which this chapter is based (May and Vinha 2010) also includes a review of issues facing bioprospecting and pharmaceutical products, artisanal fisheries and ecotourism enterprises.

access to forest resources and fishing, has identified that such dependence may affect as much as 10% of Brazil's population. Recent research by the Center for International Forestry Research (CIFOR) indicates how groups become forest dependent in order to avoid the vulnerability associated with insertion into the global market. In those conditions, biodiversity serves as a foundation for stability in local and regional development processes and for social and economic inclusion of local communities, particularly among traditional peoples in fragile biomes (Sunderlin et al. 2005).

Despite their importance for specific groups in society, the benefits associated with those assets, when these enter into commercial circuits, are often captured by commercial firms involved in the direct utilization of substances and raw materials derived from nature, such as non-timber forest products, fishery resources and ecotourism destinations. For the most part, those activities are still informal, with problems of vulnerability associated with the lack of adequate definition of usufruct rights and competition with other forms of land use that are more profitable in the short term. They generally involve low-income groups, women and sometimes minors. The firms involved in the supply chains face complex and risky conditions, which do not offer security in terms of quality, frequency, deadline, prices, etc.

Besides the market failures associated with open-access resources or with property rights that are incapable of excluding other users, extractivists suffer from information asymmetries. They are generally unaware of the price that should be charged for the product, considering the costs associated with maintaining productivity and reproducing the productive unit, which demand setting of barriers to entry and mechanisms for enforcement and self-discipline (Granovetter 1985). Those instruments are not free and require a high degree of coordination between users of natural resources subject to such discipline, as well as transaction costs to reduce the uncertainty associated with product quality and demonstrating sustainable origin. Since the products of biodiversity frequently come from sources of debatable quality or even illegal processes, there is a difficulty in assuring adequate remuneration for such costs, due to unfair competition and the lack of organization among providers of goods and services.

Among the institutional failures, there is a notable absence of a sectoral framework especially directed towards acting in all aspects involving the business, from measures for leveraging the sector (mobilization, incentive, support and protection for undertakings) to marketing and even inspection. The last stage is the most challenging, since it depends on joint and efficient action by a series of other institutions, as well as the creation and enactment of an institutional framework directed towards protecting the resources used by the sector.

The non-capture of such benefits, besides the lack of technological development, occurs in part due to the characteristic of partial public goods (non-exclusive due to the lack of land-title regularization, but generally rivals) attributed to the majority of extractive resources. Such a condition requires mechanisms to control access that will impede exhaustion of such resources due to over-exploitation ("tragedy of the commons," Hardin 1968) and instruments that recognize rights to intellectual property over such uses and encourage investment by private stakeholders. These control instruments include collective management of common property resources, which establishes rules and responsibilities for individuals, requiring confidence and knowledge regarding the resilience of the species being managed (Ostrom 1990). In many cases, such rules existed among traditional peoples but have been eroded through greater integration with the market by predatory actions of nontraditional users or by the erosion of traditions that had previously impeded their inappropriate use. These cases require intervention by authorities so as to re-establish the rules or establish new institutions capable of regulating access and use of such resources with the intention of assuring their biological sustainability and economic profitability.

16.3 Agroextractivist Products and Enterprises

This study focuses on experiences with NTFP as a case in point of biodiversity-based enterprise. The main NTFPs of regional importance in Brazil, in decreasing order of average annual value in current US$, are described in Table 16.1 below.

Territorial concentration is the norm among this smaller group of products with significant value. They are primarily derived from "oligarchical" species (Peters et al. 1989), meaning that their occurrence is geographically concentrated in certain regions or ecosystems where such species dominate (e.g. babaçu forests in the Vale do Mearim in Maranhão, açai forests in the Amazon estuary near Belém, piaçava areas in the northern *Várzea* zones). The dependence on a given region for a certain vegetal raw material tends to be very strong. More than 80% of the value for NTFPs in each state in the Amazon, for example, is concentrated in only one product (Wunder 1999). Additionally, the principal products of vegetal extractivism (historically rubber and Brazil nut and more recently açaí, babaçu and piaçava) concentrate a significant part of the total market value for the products recorded. Furthermore, roundwood or firewood greatly exceeds the value of NTFPs, which partially explains their replacement by other land uses.

A significant problem associated with agroextractive products is their low value added due to their commonly being sold in the raw. Processing, be it by drying and cracking nuts, be it by local cold-pressing of vegetable oils or other processes, represents relatively low-cost means for assuring that the product gains in market value as well as in niches other than those accessed by the product in raw form. There are no detailed studies of the segment that characterize the potential of value added in financial terms, although recent studies to identify successful cases of local productive arrangements based on such products indicate the superiority of processing enterprises in terms of net income and employment generation (Viergever 2010).

A recurring problem with agroextractivist products or those coming from socio-biodiversity chains is their weak adherence to quality standards or norms. They are generally personalized and artisanal products or dependent on rare and uncodified information, sold directly by the producers and thus without intermediaries (Storper 1998). A fairly simple policy in favour of agroextractive enterprise is that of specifying

Table 16.1 Value of principal non-timber forest products: 1985–2008 (Current US$)

Years	1985	1990	1995	2004–2008 (average)
Maté tea	24,918,384	92,110,484	34,875,137	39,328,250
Açaí fruit	28,554,855	45,831,745	34,815,441	37,824,060
Babaçu (kernel)	40,563,277	30,111,716	38,372,301	47,989,670
Coagulated rubber	78,087,627	19,986,261	6,909,760	3,520,907
Piaçava fibre	20,362,390	69,270,337	13,660,332	37,994,790
Açaí palm heart[a]	5,406,838	16,327,742	13,136,006	3,792,167
Brazil nut	19,378,986	7,224,062	5,688,986	19,436,001
Carnauba wax	6,273,611	12,299,941	2,648,493	7,000,228
Total (eight products)	**223,545,968**	**293,162,288**	**150,106,456**	**196,886,073**

Source: Wunder (1999) and IBGE/SIDRA (2009)

[a] This amount refers only to palm heart coming from the Amazon, derived from *Euterpe oleracea* (açaí)

minimum regional sale prices for extractive products associated with product quality norms. Such a policy was adopted in Brazil recently by the National Supply Council (CONAB) in response to the National Plan for Promoting Sociobiodiverse Product Chains. Minimum prices are now applicable to Brazil nuts, açaí fruit, andiroba and copaíba oil, babaçu kernel, cashew nuts, carnaúba wax and other products that adhere to quality standards (MDA/SDT 2003).

A recent example of the importance of product valuation and normatization to stimulate production of regional biodiversity products is the formation for a network to produce and process Brazil nut (*Bertholletia excelsa*) and heart of palm from peach palm (*Bactris gasipaes*), both from trees native to the Amazon region. A plant for processing Brazil nut (drying, cracking and storage) was set up in the northwest region of Mato Grosso inside a rural land reform settlement, and agreements were signed between associations of Brazil nut collectors and owners of neighbouring forest areas, residents of an extractive reserve and indigenous areas, seeking to expand the operational scale to meet a larger demand. Due to the quality, volume and local organization, the producers are now able to receive up to twice the average regional price, with support from the minimum price programme at CONAB. Such values have received additional stimulus from certification and local processing to extract Brazil nut oil; by-product meal is used in local school lunches.

Their Brazil nuts already have a well-known brand name, having participated in the annual sociobiodiversity fairs sponsored by the federal government. In the same region, three factories for processing peach palm heart have already been set up, one of them community-based. In a typical strategy in solidarity economics, that factory has signed production and sales agreements, seeking to absorb production coming from the agroforestry systems implanted in recent years in rural settlements and cooperatives throughout the region.

Additionally, the factory is responsible for selling 10 tons of peach palm seeds per year throughout the region and the state, serving as the anchor for a cultivated palm heart chain for agroforestry systems. Planting peach palm enjoys the support of municipal governments, who set up local nurseries and provide technical guidance for interested producers. More than a thousand rural producers in northwestern municipalities of the state have planted seedlings, encouraged by the production chain that has been built up around palm heart production. Technical guidelines were approved by the state, and subsidized credit in the amount of R$3 million was provided, through the Bank of Brazil together with the regional MT programme, which were issued in Aripuanã municipality to support expansion of the area planted in peach palm in land reform settlements (Paulo Nunes, personal communication, 2009). As a result, the productive systems were evaluated as having maintained a great diversity of species native to forests in the region, protecting the mosaic of conservation units and indigenous areas, attracting wildlife and acting as a component of ecological corridors (Gonçalves 2008).

In this example, market promotion has been successful in part due to attribution of territorial and social characteristics. Such products of biodiversity are singular because they bear a mark of specificity for their territory (here understood in the French concept of *terroir*), despite the fairly ubiquitous occurrence of the species on which they are based. They could thus conceivably become candidates for geographical indication (GI) of precedence and can benefit from this differential in a globally segmented consumer market that increasingly values this attribute. The drafting of territorial denominations of origin is so far quite incipient in Brazil; there are as yet few cases of their application to products derived from sustainable use of biodiversity.

According to Abramovay (2000), artisanal products generate more value when they are able to provide health guarantees and their production transmits an aura of restoring or decoding traditional knowledge. Furthermore, territorial products have the advantage of leveraging other activities developed in the same territory, increasing comparative advantages for the local economy.

Rather than focusing on high-valued niche markets for exotic products, strengthening of short productive chains, seeking to establish links between producers and consumers who are close to each other geographically, through alliances, can be a more rewarding strategy. Confidence in the quality of the products established inside such chains can lead to creation of processes for participatory certification of conformity with regionally recognized quality characteristics, as seen in the Ecovida network in Southern Brazil, which avoids the cost associated with certification of qualification of origin by third parties.

Another mechanism of interest for qualifying territorial identity is geographical indication (GI), most often associated with establishing origin brands for products such as wines and cheeses. In Brazil, the GI instrument is a recent phenomenon, managed by the Ministry of Agriculture, seeking to value products derived from sociobiodiversity. These include foodstuffs of specific origin (e.g. wines from Vale dos Vinhedos, Rio Grande do Sul; cachaça from Paraty, Rio de Janeiro; beef from the Pampa; and coffee from the Cerrado of Minas Gerais). With support from the National Historic Preservation Institute (IPHAN), which registers culturally based products, there are plans for creating a GI for handicraft articles (clay pots from Goiabeiras, Espírito Santo and decorative artefacts made of capim dourado from

Jalapão, Tocantins – which also involves the use of another NTFP, buriti “silk” for tying the bundles, etc.). The procedures that initially involve characterizing a product and its origin lead to registering the denomination of origin, seeking to protect such knowledge and practices with the locations indicated. However, the process is complex and long and does not necessarily lead to an improvement in bargaining power.

GI has also been criticized because it has not yet been directed towards valuing the products of traditional populations. It also has little sensitivity to the ecological sensitivity of extractive production systems. For example, “capim dourado” (gilded grass) in the Jalapão region of the state of Tocantins has notoriously suffered from overharvesting due in part to its growing market popularity, without an appropriate definition for collective management practices in an open-access context.

Enterprises built on biological diversity need to adopt a strategy of sequential innovation and diversified R&D targets in order to assure sustainability. This is true because markets for most of these products are narrow, demand is quite elastic to price, and therefore, their productive life is potentially ephemeral, given the appearance of lower cost substitutes, be they natural, cultivated or synthetic. The scope of the market and the price elasticity of demand indicate how much they may benefit from exceptional rents associated with the novelty of socio-environmental origin. For example, vacuum-packed Brazil nuts with the trademark of a rubber tapper cooperative in Acre, certified organic and sustainably managed by FSC criteria and harvested in the Chico Mendes Extractive Reserve, would have some competitive advantage in this market if compared with the same product resold by land-grabbers who are rapidly converting Brazil nut groves to pastures in southern Pará.

There is a clear stated willingness among consumers in general to pay for products differentiated by such characteristics, according to a recent national public opinion survey carried out by Datafolha.[4] Even so, the principal factor influencing sales in Brazil continues to be price. However, penetration by products differentiated by beneficial socio-environmental characteristics has greater acceptance even among consumer groups with less purchasing power, as a function of their natural attributes and because they are associated with healthfulness (ibid.). The widespread demand among all income groups for the Ekos natural product line of the cosmetics giant Natura is a case in point.

16.4 Investing in Biodiversity Enterprise

The rationale for investing in sustainable businesses based on use of biodiversity is the business case for sustainable use itself: transforming enterprises that deplete natural resources into lucrative initiatives that respond to the demands of society

[4] A survey done in March, 2009, noted that 81% of the population interviewed (with ±4% error) would prefer buying products derived from processes certified as responsible, even if they cost more than a similar product lacking such a label (Datafolha 2009).

(stakeholder approach), thus valuing origin-based products. This reasoning seeks to strengthen conservation of genetic resources: their use assures that value is associated with maintaining ecosystems intact, either through the benefits directly derived or through the services generated by the remaining biodiversity for the functioning of critical adjacent agroecosystems and natural resources.

It is recognized that there are risks of irresponsible use of nature, in homogenizing ecosystems by specializing in a range with less diversity (e.g. management or enrichment with the objective of intensifying resources extracted). According to some analysts, any management may become harmful to biodiversity, with full protection being preferable to sustainable use (Durojeanni and Pádua 2001). A more realistic view is that it is probably more rational to promote some management in more resilient compartments than to lose the entire ecosystem due to the lack of value attributed to genetic resources in their natural state.

Despite recognition that potential for biodiversity to serve as a basis for important new products and services is immense, such enterprise continues to suffer from underinvestment. The few venture capital initiatives (investment funds focused on new initiatives with high rates of return, but high risk, directed towards use of biodiversity) have failed due to excessively high profit expectations and underestimated payback periods. The most successful strategies involve a combination of long-term credit with appropriate grace periods, commercial partnerships, technical-administrative support and constant nurturing.

The financing or development of commercial channels, frequently with non-refundable financing, has so far come from international, bilateral and multilateral agencies, international NGOs and partner companies. Interesting domestic examples in this regard were initiated by companies such as Natura in the cosmetics area and the major supermarket chain Pão de Açúcar (the "Caras do Brasil" programme). It should be noted that experiences of this type can prosper only if they are not treated as a residual response to corporate socio-environmental responsibility but as an integral part of business.

One area that is still little exploited is the integration of markets for environmental goods and services, notably those that involve both conservation of biomes and watersheds through avoided deforestation or environmental restoration (carbon credits, water production) and sustainable use of remaining biodiversity (direct management or extraction or indirect use for recreational purposes).

Another option for financing is in agroforestry systems (intercropping trees and crops or animals in the same area). Although this model presents restrictions as to biodiversity protection if it is excessively simplified in terms of number of species, it performs a pedagogical role in a broader scenario of transition. Since such systems can add provision of environmental services (e.g. maintenance of terrestrial carbon stocks), they may be more appropriate for undertakings that seek to reduce emissions than those for conserving biodiversity.

In structuring a financial portfolio, maintaining diversity in assets is considered advisable, taking advantage of opposing cycles of growth and decline (when the stock market falls, the dollar exchange rate rises). By the same measure, maintaining biodiversity enables greater stability for the assets used from nature. The insurance offered by maintaining biodiversity goes beyond that, since ecosystem resilience is directly related to biodiversity, meaning, nature's capacity to continue supplying all of the goods and services, even when confronted by major sources of stress or cataclysm, is greater to the extent that such environments exhibit a greater diversity of species. The inverse is also true: an agroecosystem or monospecific forest may face conditions of vulnerability that lead to loss of productivity or, in extreme situations, to collapse.

The investment experiences in enterprises seeking the sustainable use of biodiversity resources have had varying results, but the available analyses of experiences show that the most successful financial-institutional arrangements are grounded, to a greater or lesser degree, in the principles of solidarity economy.

In most cases, the success of the productive arrangement depends upon financial, technical and commercial partnerships with larger companies and/or solidarity networks that establish a link between the group having access to the natural resources and the market. Such relationships can operate through informal or contractual partnerships (supply or advances towards purchase of inputs or equipment, with commercial obligations) or financing along the supply chain.

One may establish a typology of enterprises directed towards the sustainable use of biodiversity, including compositions between the following actors and groupings:

- Cooperatives or community associations
- Solidarity networks for process conformity and short supply chains
- Corporate enterprises
- Mixed enterprises (partnerships, stimulus, chains)
- Micro, small- and medium-sized companies
- Public entities for technical and financial support

Recent new multi-institutional formats innovate through organic integration, from the initial project design, the financing agent, a local catalyzing productive agent and municipal government levels. Two arrangements have achieved special visibility: the miner/extractivist enclave and sociobiodiversity chains, exemplified below.

With the development of multi-sector investment actions, it is becoming increasingly evident that institutional arrangements capable of dealing not only with the inherent complexity of biodiversity must function not only at the project but also at the territorial level. This observation is relevant both for analysing the impact of projects that may create impediments to sustainable use of natural resources and for carrying out actions favourable for their development. This implies the need for mapping and coordinating interventions between institutions from the public and private sector.

There is a series of conceptual and instrumental formulations practised by public agencies and banks that seek to act strategically in territories, which include "clusters." In some cases, such formulations simply group existing programmes into the same region, under the supervision of some coordinating agency. In others, the territory is the starting point for distinct formulations for reinforcing productivity, adding value and commercialization for specific chains based on products of regional biodiversity.

All of those initiatives presuppose engagement by municipal governments as a prerequisite for success, although that involvement, in most cases, is not very effective and is ephemeral, because it is tied to the political-electoral calendar. One of the strategies that have been adopted to neutralize patronage interference by municipal governments is to involve a large locally based company as a preferential partner. One recent example is the Juruti Sustentável project, the result of a pioneering partnership between the national biodiversity foundation Funbio and a major mining company, Alcoa. In general terms, the project proposes a local development model for the municipality of Juruti, located in western Pará in the heart of the Brazilian Amazon, capable of being replicated in other municipalities affected by large mining operations. In one project, it combines mitigating measures required by the licencing agency and social responsibility actions defined by the company based on the results of a diagnosis and consultation with local society. The innovative component for the project lies in the creation of a fund, FUNJUS, to implement the compensation measures, which has the particularity of being directed towards stimulating entrepreneurism. Through a public call for proposals, it supports projects drawn up by the local population and chosen by a council made up of community members (CONJUS).

16.5 Conclusions

As discussed in this study, it is important for the success of enterprises built on the basis of biological diversity to adopt a strategy of constant innovation and diversified R&D targets in order to assure sustainability. This is due to the fact that the markets for most of these products are narrow and their productive life is potentially ephemeral, given the appearance of substitutes, be they natural, cultivated or synthetic. The scope of the market and the price elasticity of demand indicate how much they will benefit from exceptional prices associated with socio-environmental origin. Yet the immense popularity of natural products such as açai can surprise and result in considerable broadening of opportunities complementary with diversified agroforestry and natural forest management.

To make a difference in the fight against biodiversity degradation, investment in sustainable use of agroextractive products must take a long-term perspective, considering forest peoples' dependence on such resources for local livelihoods ("the GDP of the poor"…) and the complexities of collective action and enterprise development in such settings. Brazil hosts a tremendous range of experience in this area, with considerable variation in institutional structure, capitalization, scale and market scope. Many experiences to date have relied primarily on public or non-profit resources for start-up capital, and often are difficult to successfully wean and emancipate from such support, and survive as profitable community enterprises, generative of social capital.

In conclusion, we favour a cautious approach to investment in utilization of biodiversity-based resources as a contributor to conservation. Partnerships with business enterprises in resource-based clusters, with the right price incentives and local entrepreneurship, can make a remarkable difference to quell the local pace of deforestation and biodiversity loss. Such initiatives, when combined with a broader set of policy instruments, have considerable promise for managing territorial development and protecting remaining biodiversity in fragile biomes such as the Brazilian Amazon.

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Chapter 17
Integrating Agroecology with Payments for Ecosystem Services in Santa Catarina's Atlantic Forest

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17.1 Introduction to the Problem

Agroecology may be a uniquely viable solution to one of the most serious dilemmas currently facing humanity. On the one hand, there are a billion malnourished people on the planet. The global population is expected to increase by two billion by 2050 at that same time that income growth increases the demand for animal protein. Failure to increase food production by at least 70% by 2050 could have unacceptable humanitarian costs (FAO 2011). On the other hand, failure to restore global ecosystems and the life-sustaining services they provide poses serious threats to human civilisation. Unfortunately, with current technologies, agriculture is the greatest global threat to ecosystem services, including those that sustain agriculture (MEA 2005). Conversely, ensuring the continued provision of vital ecosystem services requires extensive ecosystem restoration, along with reductions in nitrogen, phosphorous, greenhouse gases, toxic chemicals and freshwater use (Rockstrom et al. 2009), threatening food production. On our current path, we are forced to choose between ecological collapse and widespread malnutrition or worse. Since agriculture itself depends on the continued flow of ecosystem services, the best we can do with current agricultural technologies is stave off starvation.

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The market economy is ill suited to solving this problem. While land, food and raw materials provided by nature are typically market goods with market prices, many ecosystem services are public goods with no market price. A pure public good is both non-excludable, meaning that one cannot prevent others from using it, and non-rival, meaning that use by one person does not affect the quality or quantity left for others. If people cannot be prevented from using a resource whether or not they pay, they are unlikely to pay, and markets will fail to provide the resource. This explains the rapid degradation of ecosystem services around the planet. If use of a resource does not leave less for others, then market prices inefficiently ration use, creating artificial scarcity. For example, markets will ration access to patented technologies that protect our ecosystems, reducing their benefits to humanity in exchange for profits. Markets are not an option for non-excludable resources and are not desirable for non-rival ones. As a result, the market system awards resource owners for the benefits of ecosystem conversion (e.g. timber and farm land from cleared forests), but typically fails to award them for benefits of conservation (e.g. flood and climate regulation by intact native forests). Markets systematically favour conversion over conservation, regardless of their relative benefits to society. Because ecosystems exhibit highly complex, dynamic and nonlinear behaviour, including the presence of abrupt, irreversible thresholds (Farber et al. 2002; Folke 2006; Limburg et al. 2002), excessive conversion threatens the irreversible loss of essential services.

On the socio-technological end, agroecology may be uniquely capable of solving this dilemma. Agricultural systems designed to mimic natural processes may be capable of increasing the provision of ecosystem services from farmland and the provision of food, fibre and fuel from ecological restoration while reducing the use of nonrenewable and toxic inputs. Despite minimal investments in agroecology relative to conventional agriculture, numerous studies suggest that it can simultaneously increase agricultural yields, farmer incomes, ecosystem services and resilience in the face of extreme weather events (De Schutter 2010; Gliessman 2007; Pretty et al. 2005). However, a complete solution will require economic institutions that promote agroecology and are capable disseminating it rapidly to a global scale. This chapter proposes economic institutions that reward the provision of ecosystem services generated by agroecology. Though we draw largely on our agroecology research in Santa Catarina's Atlantic Forest, for example, we believe the basic approach we propose could be readily applied elsewhere.

17.1.1 Santa Catarina's Atlantic Forest

Brazil's Atlantic Forest offers an interesting case study of the conflict between agriculture and ecological resilience. Over 90% of the original 1.5 million km^2 has been lost to economic activities (Tabarelli et al. 2005). Though forest remnants still exhibit some of the highest levels of terrestrial biodiversity and endemism ever recorded, they also harbour more threatened and endangered species than any other Brazilian ecosystem (Costa et al. 2005). A rough rule of thumb from island biogeography suggests that when an ecosystem decreases in size by 90%, species diversity decreases by 50% (MacArthur and Wilson 2001). Research in the southeastern Atlantic Forest finds that over 60% of birds are extinct, critically endangered or vulnerable (Ribon et al. 2003), while in the northeast, over a third of tree species are currently threatened with extinction (da Silva and Tabarelli 2000). Significant time lags between forest loss and extinction best explain why more extinctions have not yet occurred (Brooks and Balmford 1996; Metzger et al. 2009). While biodiversity is not an ecosystem service itself, it plays an essential role in sustaining all ecosystem services (MEA 2005), suggesting that without active intervention, the Atlantic Forest may be due for a catastrophic loss of biodiversity and the ecosystem services it sustains.

Brazil has outlawed continued deforestation of primary or advanced secondary Atlantic Forest. In addition, the Brazilian Forestry Code mandates a forest legal reserve (RL) on 20% of Atlantic Forest properties and a permanent protected area (APP) of forest cover on hilltops, slopes over 45%, for 30 m along rivers under 10 m in width (increasing along larger rivers) and for 50 m around springs. However, these environmental laws are poorly enforced (Laurance 1999; Ministério do Meio Ambiente 2011) for valid reasons: Enforcing the law would require many small farmers to reforest well over half their property, which would drive them into poverty. The region thus confronts an ecological threshold in terms of biodiversity and ecosystem service collapse in the absence of reforestation and an economic threshold in the form of abject poverty if farmers reforest.

If we look at biodiversity collapse and the loss of ecosystem services as marginal costs of agricultural production, they increase very sharply as land clearing reaches an ecological threshold. On the other hand, from the perspective of poor land owners near the poverty threshold, the marginal benefits of agriculture are the satisfaction of basic needs and hence are also extremely high. Brazil also makes a significant contribution to global food supply, where even small decreases in output can lead to dramatic increases in price. The marginal costs of food production (the supply curve in economic analysis) and marginal benefits (the demand curve) fail to intersect, as depicted in Fig. 17.1. The ecological threshold however confronts a significant time lag before it becomes irreversible, while the costs of poverty are more immediate and thus more difficult to ignore.

The results of this conflict are particularly visible in Santa Catarina state which retains 23% forest cover, mostly in secondary forest (SOS Mata Atlantica 2009), but suffers the most rapid loss of Atlantic Forest in Brazil (Meister and Salviati 2009). Abundant evidence suggests that deforestation contributes to the frequency and severity of flooding and landslides in the region (Arcova et al. 2003; Faria and Marques 1999; Frank 1995; Ministério do Meio Ambiente 2011). Small family farms, few of which comply with Brazil's forest code, account for 87% of all properties and 44% of the land in the state. One cause of deforestation has been declining incomes in rural relative to urban areas, leading farmers to clear more forests in order to increase short-term income (Frank 1995). Santa Catarina suffered from catastrophic flooding in November 2008, which official documents describe as the worst tragedy in the state's history, and again in January 2011. The major cause

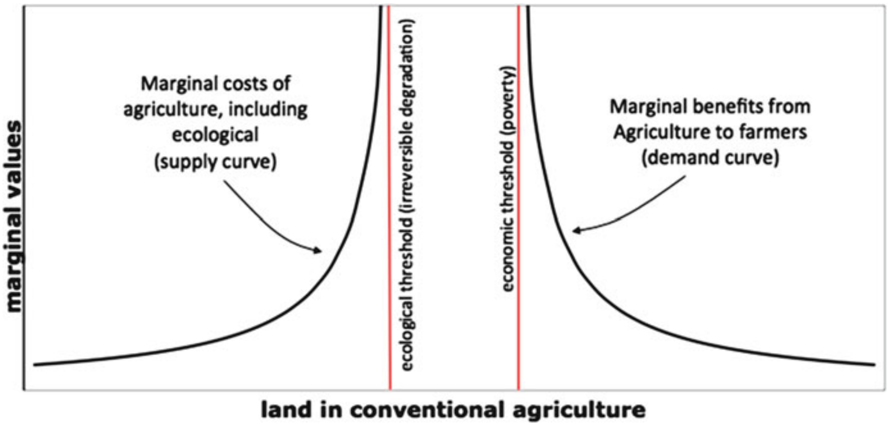

Fig. 17.1 Marginal costs and benefits of food production

of mortality and economic damage was from landslides, primarily on deforested hillsides, though also on hillsides with secondary forest (Defesa Civil Santa Catarina 2010). The state's major port remains heavily damaged, and as a result the state has lost significant port traffic to its neighbours. Nonetheless, in December of 2008, the state governor attracted national attention by announcing that the state had to choose between 'crops or slums' and would therefore significantly reduce legal protection of remaining forests in direct defiance of the national forestry code (Souto 2009). This has triggered a nationwide debate over the forestry code (Metzger et al. 2010).

17.1.2 Potential Solutions and Organisation of the Chapter

The solution to this conflict between ecological and economic thresholds must lie in developing land uses that simultaneously provide both ecological and economic services. In the context of Santa Catarina's Atlantic Forest, this means restoring some farmland with healthy ecosystems that generate economic benefits and increasing the ecosystem services generated from agricultural land. However, changing land uses will require significant investments. Small farmers have no surplus capital available, and interest rates in Brazil are among the highest in the world, so simply borrowing money to invest is not a viable option. The solution therefore requires financing as well.

Agroecology and forestry systems offer a potential partial solution to this conflict by providing positive economic returns from ecological restoration and increasing ecological benefits from agricultural land. Unfortunately, markets fail to compensate for the public-good ecosystem services provided by agroecosystems, which means there may be inadequate incentives for adopting agroecology. Payments for ecosystem services (PES) that transfer revenue from the beneficiaries of ecosystem services to the individual farmers who provide them have been proposed as a solution to this problem (Pagiola et al. 2004; Pagiola et al. 2007a). One significant challenge to PES is capturing revenue from beneficiaries, especially when the ecosystem services in question are public goods that cross political boundaries. Another challenge is that the broad dissemination of agroecology requires substantial public sector investments in site-specific research and development, agricultural extension, infrastructure required to bring products to market, and low-risk, low-interest financing mechanisms (De Schutter 2010). Individual farmers are unlikely to make public-good investments, and the private sector is unlikely to provide affordable finance options. The rapid dissemination of agroecology may therefore require a significantly different type of PES, in which public sectors of those nations that benefit from national and global ecosystem services transfer resources to the public sectors of those regions adopting agroecology practices in order to invest the public goods required to promote it. Furthermore, if funding is needed to promote agroecology, it cannot be made available only after the services have been provided. We need instead a programme of public sector venture capital, in which those governments benefitting from the provision of non-excludable ecosystem services finance their provision, thus sharing the risks as well as the rewards.

This chapter will use a case study of efforts to promote agroecology on the mountain slopes of the coastal range (Encosta da Serra Geral) of Santa Catarina to provide insights into the effective integration of PES and agroecology. Section 17.2 very briefly describes the case study region. Section 17.3 discusses Brazil's national forestry code and its implication for ecosystem services and small farmers. Section 17.4 introduces agroecology; it presents two different agroecology systems, one for farmlands and one for APPs, and provides preliminary results from research into the ecological and economic benefits of recently initiated agroecology projects in the region. Section 17.5 examines PES as a financing mechanism. It focuses on bundling the services of carbon sequestration, biodiversity and watershed regulation, and outlines potential payment schemes based on the physical characteristics of the services and institutional constraints.

17.2 Project Site Description

The Encosta da Serra Geral extends from north to south in Santa Catarina, roughly parallel to the coast. It retains the state's last vestiges of primary Atlantic Forest and sustains a wide variety of well-preserved Atlantic Forest ecosystems, ranging from broad-leaved forests to mangroves and high-altitude grasslands, which in turn support impressive levels of biodiversity and endemism. Our research is concentrated on the region surrounding the 87,405 ha Parque Estadual da Serra do Tabuleiro (PEST), the largest conservation unit in Santa Catarina, which borders the capital

Florianópolis (FATMA n.d.; Tabarelli et al. 2005). This region is the source of several important rivers, including those responsible for water supply to Florianópolis and a dozen adjacent communities.

The park is bordered by nine municipalities: Florianópolis, Palhoça, Santo Amaro da Imperatriz, Águas Mornas, São Bonifácio, São Martinho, Imaruí, Garopaba and Paulo Lopes (FATMA no date). Municipalities range from some of the wealthiest in the state to some of the poorest. Farming is one of the main sources of income and is characterised by small family farms with low productivity and few inputs, focused primarily on staple crops and pasture (Vieira et al. 2007). The Federal University of Santa Catarina (UFSC) has an active agricultural extension programme in the region.

17.3 Brazilian Forestry Code: Implications for Ecosystem Services and Small Farmers

As briefly described in the introduction, the Brazilian Forestry Code (BFC) mandates forest cover in permanent protected areas (APPs) and the legal reserve (RL). APPs are intended to preserve hydrological resources, landscape, geological stability, biodiversity and gene flows of flora and fauna; to protect the soil; and to ensure the well-being of human populations (Ministério do Meio Ambiente 2011). Small farmers are allowed to extract non-timber forest products from APPs (Resolução CONAMA 2006) and to subtract the area in APP from the area required for RL. The RL must be dedicated to the sustainable use of natural resources, the conservation and rehabilitation of ecological processes, the conservation of biodiversity and the shelter and protection for native flora and fauna, but is less restricted in its use than APPs (Metzger 2010). Unfortunately, there is very little enforcement of the BFC in general, and enforcement may be particularly lax in Santa Catarina (Souto 2009). The Brazilian congress is currently debating revisions to the BFC that would significantly weaken current levels of forest protection (Metzger et al. 2010).

If the current BFC were enforced, however, the impact on ecosystem services could be profound. The APP covers 10–20% of the land area in most Atlantic Forest states (Metzger et al. 2009; Tabarelli et al. 2005), and combined with the RL would bring forest cover to over 30%, considered the minimum necessary to avoid crossing critical ecological thresholds in the Atlantic Forest. Riparian forests increase the connectivity of existing forest fragments and their capacity to sustain biodiversity, though a 60-m corridor may be inadequate for many species. A 30-m margin does appear adequate however to capture most nitrate runoff from agricultural lands, thus improving water quality (Metzger 2010). Restoring forest cover on slopes and hilltops is likely to reduce landslides and slow runoff during storm events (Sidle and Ochiai 2006; Vanacker et al. 2007). The Atlantic Forest captures and retains airborne moisture, known as hidden rain, which can account for up to 45% of total water in the system. Forest restoration is therefore required to reduce drought and the negative impacts it has on agriculture, quality of life and the ecosystem itself (Anido 2002; Barboza 2007; Cavelier et al. 1996). Reforestation also increases carbon sequestration relative to pasture (May et al. 2005).

The problem is that many farmers in the municipalities surrounding PEST are at or near poverty level, and the APP and RL can make up the majority of the farmland for farms in our study area. In a pilot survey, we found only one farmer was in full compliance with the BFC regulations, while other farmers reported 30–90% in illegal uses, primarily agricultural production. Seventy-five percent of the interviewed farmers reported that compliance with environmental laws would decrease their income by at least 50%. In a separate, more comprehensive survey of farmers in the same region, 90% said they would only comply with the BFC if forced to do so (Farley et al. 2010a). Extensive field experience in the region supports survey results and suggests that it is extremely difficult for small farmers using conventional technologies to comply with the BFC and remain viable. Agroecology may offer a solution to this problem.

17.4 Agroecology

The transdisciplinary field of agroecology recognises that agricultural systems are subsystems of the global ecosystem and obey the general principles of ecology (Gliessman 2000). Agroecology focuses on the productivity, stability, sustainability and equity of agricultural systems (Marten 1988), paying particular attention to the needs and aspirations of poor farmers in marginal environments (Altieri 2002). This project focuses initially on two agroecology systems. Agroforestry systems in APPs and RLs can increase farmers' income from areas primarily dedicated to conserving and restoring ecosystem services. Management intensive grazing (MIG), also known as Voisin grazing, can increase both ecosystem services and economic returns on established pasturelands.

In terms of Fig. 17.1, agroecology reduces the ecological costs of agriculture, thus shifting the ecological threshold and supply curve to the right. Agroecology also increases the monetary returns to agriculture and creates a new source of revenue from the APPs, shifting the poverty threshold and demand curve to the left. The result is the potential for socially and ecologically acceptable solutions, depicted in Fig. 17.2, in which there is no longer an unavoidable trade-off between ecological and economic thresholds.

17.4.1 Agroforestry Systems

While there are a wide variety of agroforestry systems (AFS), our goal is to adopt a successional approach prioritising native species providing non-timber forest products, which seeks to recreate the structure and function of Atlantic Forest riparian

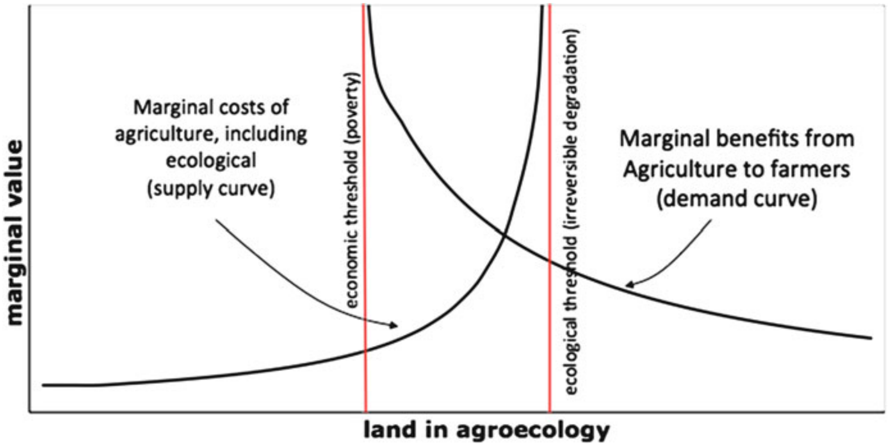

Fig. 17.2 Socially and ecologically acceptable solutions

zones and hence restore the full suite of ecosystem services they provide. Such systems in Brazil have been shown to eliminate the use of pesticides and fertilisers, filter polluted runoff into waterways, provide habitat for native flora and fauna and sequester carbon, among other benefits (Bittencourt 2007; May and Trovatto 2008; May et al. 2005; Rodrigues et al. 2007).

Campello et al. (2007) estimated that implementing a successional AFS in the Atlantic Forest in which bananas and pineapples are succeeded by other fruit trees and timber species costs about R$13,500 (~US$7,500) per hectare to implement with positive and increasing financial returns after only 2 years. May et al. (2005) estimated an internal rate of return of 18.4% for AFS relative to conventional agricultural in the Atlantic Forest of Rio de Janeiro.

A particularly promising species for AFS in Santa Catarina is the native jussara palm trees (*Euterpe edulis*), used for its fruit (marketed as açai[1]) and for heart of palm. Açai fruit is extremely high in antioxidants, shows rapid market growth in Brazil and promises even more rapid growth as an export crop. The tree grows in the shade and has a small crown allowing other crops to thrive, even when planted at high densities, and production is highly profitable (Homma et al. 2006).[2] Açai palms produce an average of 4 kg of fruit per year, with prices ranging from R$0.70 for raw fruit to R$4.00 for frozen pulp. A density of 1,000 trees per hectare allows for intercropping with other species and earns from R$2,800–16,000

[1] The true açai palm (*Euterpe oleracea*) is found farther to the north, but we will refer to the fruit of *E. edulis* by its market name.

[2] Note that Homma is referring to *Euterpe oleracea*, native to northern Brazil. However, *Euterpe edulis*, native to southern Brazil, is quite similar. All other references are to *E. edulis*.

(about US$1,500–8,900), depending on the degree of processing (Fadden 2005). The açai palm is only one of dozens of species native to the Atlantic Forest biome that can provide food and other non-timber products, though for many of these other species, the lack of markets remains a problem.

The dominant costs in all the systems described above are labour and seedlings. Campello calculates that if labour costs are borne by family farmers and seedlings are found on site, implementation costs fall to R$1,500/ha. Fortunately, through government/NGO partnerships, the region already has a 'Viveiros Nativos' (Native Nurseries) project which produces high-quality native seedlings at accessible prices (Vieira et al. 2007). Preliminary interviews suggest that regional farmers are particularly interested in açai production and prone to plant their riparian zones if seedlings are provided.

Bringing all family farmers into compliance with the BFC through agroforestry projects could substantially reduce the threats of biodiversity and ecosystem collapse in the Atlantic Forest. There are currently two sources of incentives for farmers to comply: potential returns to agroforestry and fear over penalties for noncompliance. We are in the process of developing agroforestry systems with native species in an effort to improve the returns to agroforestry and set up pilot projects for educating farmers. However, scaling up our efforts to where they could have a significant impact would require some combination of more agroforestry extensionists with more resources for farmer education, better sources of finance, payments for the ecosystem services provided by agroforestry or greater threats of punishment for noncompliance with the BFC.

17.4.2 Voisin Grazing Management

Pasture for milk and beef production accounts for nearly half the land use in the region study area. Soil erosion from lack of vegetation, applications of pesticides and fertilisers, use of rivers and springs as watering holes and continuing deforestation of native forest for pasture all have serious environmental impacts (Pinheiro Machado 2004). Furthermore, economic returns from conventional pasture are generally quite low. EMBRAPA (2006) estimates that average returns from traditional cattle production in Brazil are ranging from R$18–180/ha-year (~US$10–100).

A more ecologically and economically viable alternative is managing intensive grazing (MIG), in which pastures are divided into numerous plots with fences. Water is pumped to tanks in each plot to keep cattle away from riparian zones. Cattle are moved from pasture to pasture, mimicking their movements in nature and maximising pasture growth rates. The resulting increase in pasture-grass biodiversity both increases and stabilises production (Tilman and Downing 1994). Pasture is never allowed to be overgrazed, ensuring better ground cover, less erosion and better capture of nutrients from manure, reducing the need for fertilisers. Stock rotation interrupts the reproductive cycle of insect pests, reducing the need for pesticides, while healthier, more biodiverse pasture reduces the need for fertilisers and herbicides. More productive pastures actually increase soil carbon content, sequestering CO_2 from the atmosphere (Lenzi 2003; Melado 2000, 2007; Pinheiro Machado 2004).

On the economic side, MIG increases output while decreasing inputs. Extension professors at UFSC have implemented MIG projects in over 500 properties in the region. Initial surveys of participating dairy farmers ($n=67$) found that 91% were able to increase the number of cows per hectare, and 90% increased yield per cow total yield and revenue; 49% of farmers stated that labour requirements decreased, while 27% stated they had increased; 8% of farmers claimed that pasture grass improved in quality, 25% that it increased in quantity and 65% that both quantity and quality improved greatly. Concerning herd health, the vast majority of farmers found that ticks, horn flies (*Haematobia irritans*), worms and mastitis all decreased, in many cases significantly, while no more than 5% found that any of these diseases had increased. Over 98% of farmers said that their initial investment was generating the desired returns or more. Nearly 70% of farmers repaid the initial investment in the first year, and over 87% did so within 2 years. Perhaps most important, 85% claimed that the project improved their quality of life.

The same surveys also confirmed the positive ecological impacts. Prior to adoption of MIG, 73% of farmers used pesticides, 28% over the entire pasture; after adoption, these numbers fell to 54 and 3%, respectively. Over 72% of farmers claimed that manure decayed faster after MIG, and over 85% claimed their soil was moister during droughts. Total vegetation coverage increased from under 2% of pastures to over 72%, while areas with scant coverage decreased from over 73% to less than 2%. Over 85% of farmers noticed an improvement in soil quality.

Silvopastoral intensive grazing (SIG) systems further increase ecological and economic benefits. Silvopastoral systems combine fodder plants with trees and shrubs for animal nutrition and complimentary uses (i.e. fodder banks, live fences, windbreaks, etc.) (Pagiola et al. 2007b). Trees provide essential shade for the cows, protect pastures from drying, cycle nutrients from deeper soil layers to the surface, provide additional fodder and can also produce fruits and wood. Improved shade cover alone can increase production by 20% (Freitas 2008; Melado 2007; Pinheiro Machado 2004). We are currently initiating an experimental SIG system utilising 60 different native species, including açai.

Implementing SIG or other agroecological production techniques on all degraded pastures in the case study area could dramatically increase the flow of ecosystem services from farmland. The evidence presented here suggests that the agroecology systems are more profitable than the agricultural systems they replace, and there is a convergence between private and social land use decisions. However, the vast majority of small family farmers in Santa Catarina's Atlantic Forest have not yet adopted them. Our research suggests that the major obstacles to the spread of SIG include the up-front investment costs and the time lag before the systems begin producing, which can be particularly problematic in Brazil where interest rates on loans can easily exceed 40% (Dantas 2010); the lack of education and extension services, whose costs were ignored in the results above; and the poor infrastructure which makes it difficult to get products to market (especially milk) or to add value.

17.5 Payments for Ecosystem Services

An increasingly popular approach to improving the provision of ecosystem services is simply to pay for them or for land uses associated with their provision (Engel et al. 2008; Ferraro and Kiss 2002). Hundreds of PES and PES-like schemes exist around the world (Landell-Mills and Porras 2002; Pagiola et al. 2002). In Brazil, for example, the 'Cordão de Mata' project has negotiated forest conservation easements with dairy farmers in the Atlantic Forest (Jenkins et al. 2004), there are numerous examples of public payments for water regulation services, and a number of Brazilian states have adopted an innovative PES scheme known as the ICMS ecológico, in which a portion of state sales taxes are refunded to municipalities roughly in proportion to the ecosystem services they generate (Ring 2008). An appropriate PES scheme could finance and complement the adoption of agroecology projects.

There are two general approaches to PES, one based on trying to force ecosystem services into the market model with the goal of increasing economic efficiency and the other based on adapting economic instruments to the specific characteristics of ecosystem services (e.g. rivalry, excludability and spatial distribution) in order to achieve a variety of goals, such as sustainability, justice and efficiency (Farley and Costanza 2010). The nature of the investments required to protect or restore ecosystem services also matters. If protecting ecosystems requires investments in private goods, then payments to private landowners may be appropriate. However, if the required investments are public goods, then the private sector is likely to underinvest, and payments to individuals may be inappropriate (Farley et al. 2011).

Proponents of market approaches recognise the market failures affecting the provision of ecosystem services, but believe 'that the conditions that underlie market failure, namely non-rivalry and non-excludability, are dynamic' (Landell-Mills and Porras 2002, p. 11). The fact is however that rivalry is a purely physical characteristic and not at all dynamic. For example, information is never depleted by use, but timber always is.[3] Excludability is in some cases a dynamic policy variable, but some ecosystem services, such as climate regulation, are inherently non-excludable as an immutable physical characteristic (Daly and Farley 2010; Farley and Costanza 2010). Only a minority of ecosystem services fit the market model, and we cannot change their inherent physical characteristics to improve their fit.

Furthermore, the investments required to promote agroecology, such as R&D, extension services and infrastructure, have strong public-good characteristics and thus also fail to fit the market model of PES. There are real costs to protecting and restoring most ecosystem services and to developing and disseminating agroecology, and someone must pay them, but market-like mechanisms will generally be

[3] The error many economists make is confusing abundance with non-rivalry. For example, oxygen is currently abundant in the sense that my use does not affect your use, but it is also rival, because my use of oxygen transforms it into CO_2, leaving less for you to breath. When oxygen becomes scarce, such as when miners are trapped in a cave-in, the rivalry becomes obvious, but in normal conditions of abundance, it appears non-rival. The physical characteristic of oxygen as a rival resource cannot be affected by policy.

inappropriate. Instead, we should adapt economic institutions both to the physical characteristics of the services provided (e.g. rivalry, excludability and spatial distribution) and to the characteristics of the investments required to provide the services (e.g. public or private). We therefore follow Muradian et al. (2010) in defining PES as 'a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources' (p. 1205). This definition allows for payments by the public sector in regions that benefit from ecosystem services and payments to the public sectors of regions that generate them. Some of the literature argues that private sector PES is more effective and efficient than PES schemes that involve the public sector (Engel et al. 2008; Wunder et al. 2008), but one would expect this result simply because collective action problems concerning non-rival or non-excludable resources are inherently more difficult to solve.

In the following sections, we break down the problem of using PES to promote agroecology in Santa Catarina into two components: how to capture revenue from beneficiaries and how to disburse payments to providers.

17.5.1 Capturing Revenue from Beneficiaries

As discussed in the introduction, the private sector is unlikely to voluntarily pay for the provision of non-excludable ecosystem services such as flood regulation, climate regulation or ecological resilience promoted by biodiversity. Non-excludable services are open access by definition and cannot be rationed among users. Price rationing and hence market-based payment schemes are not an option. Instead, collective economic institutions are required, either to create and enforce excludable property rights so that market-based payment schemes are possible or to collectively pay for the provision of the open access services. The biggest challenge to collective action may well be the spatial distribution of the ecosystem services generated, which ignore political boundaries, sometimes covering only part of a political jurisdiction and sometimes crossing over into several, both national and global. In either case, conventional models for public sector provision of public goods are suboptimal (Olson 1969).

However, the fact that an ecosystem service can be made excludable does not automatically mean that market payments for the service are a good idea. If a service is non-rival, then using prices to ration access creates artificial scarcity and paradoxically diminishes the monetary value of the service as measured by economic surplus (Daly and Farley 2010; Kubiszewski et al. 2010). If a service provides a commodity that is rival but also essential with limited possibilities for substitution, such as drinking water, then markets may systematically exclude the poor, depriving them of basic needs. If we accept the law of diminishing marginal utility, this may be inefficient as well as unjust.

Ecosystem services provided by agroecology cover all possible combinations of rivalry and excludability and all possible spatial distributions. This suggests that a variety of different approaches will be required to capture revenue from beneficiaries. We explore four different types of services.

17.5.1.1 Provisioning Services

Agroecology of course provides food, fibre, and/or fuels directly to the landowners who adopt it, all of which are clearly market goods. Our research in Santa Catarina suggests that agroecology increases farmer income, which means that there is no opportunity cost from land use change. However, the investments required to develop agroecology are frequently public goods. National and state governments have historically invested in the public-good R&D and infrastructure required for agriculture with an exceptional track record. In fact, it is public support of agriculture extension conducted by the Federal University of Santa Catarina that has made our project viable. A global meta-analysis found that rates of return to public sector agricultural R&D average 43% (Alston et al. 2000). Returns to public-good investments in rural Latin America are similarly high and non-declining. However, many government expenditures are used to subsidise private goods (e.g. fertilisers, pesticides), with low or even negative social returns, in part because such subsidies are readily targeted towards politically influential (i.e. wealthy) farmers (López and Galinato 2007; World Bank 2007). Simply shifting existing expenditures from subsidies to public investments could increase agricultural output by more than 40% in some Latin American countries. In fact, research shows that 'reducing the share of subsidies to private goods in the government's budget has a large and significant positive impact on rural per capita income, reduces certain undesirable environmental effects associated with output expansion, and contributes to poverty reduction' (López and Galinato 2007, p. 1072).

Another potential source of revenue is from consumers willing to pay a premium for certified 'green', organic, fair or sustainable products. To be effective, however, the premium must cover the costs of certification and verification, which can be high. Though the market in certified products has increased 200% in the last decade, it still represents only 2.5% of the global food and beverage market. In Santa Catarina, certification of agroecological products already exists under the label Ecovida. Many consumers buy such products for their health impacts, a private benefit, rather than for the ecosystem services they provide. This may be particularly true in Brazil, which has the world's highest use of agro-toxins (Pacheco 2009). To the extent that consumers are self-interested, they are unlikely to pay extra for the provision of public goods. Relying on altruistic behaviour may contribute to solving the problem but is unlikely to generate adequate revenue by itself.

17.5.1.2 Watershed Services

The recently adopted state policy on ecosystem services in Santa Catarina (law 15.133/2010) covers a variety of ecosystem services, but its first application in March 2011 was for water regulation. Payments are made by the municipal water and sewage utility of the town São Bento do Sul to farmers willing to restore land in the APP along the Rio Vermelho river.

PES for water regulation is fairly straightforward. Water for household use is typically controlled by a water utility, which is a monopolistic (ideally publicly owned or regulated) intermediary between service providers and service beneficiaries, and can therefore serve as the monopsonistic purchaser of the land uses that improve water quality and stabilise water flow. A monopsony occurs when one buyer faces many sellers. While monopsonies in conventional market goods are undesirable because they allow the purchasers to set prices, upstream land owners can choose between current land uses or those that provide water and hence need not accept the price offered by the monopsonist (Kemkes 2008). By passing price increases on to consumers, these utilities can ensure that all beneficiaries contribute to the payment. Since municipal water use is rival, payments for each unit used are appropriate as long as the poor are still able to satisfy their basic needs.

Tap water quality is notoriously poor in some of the municipalities in our research area. For example, in Paulo Lopes, none of the major municipal water sources regularly meet basic standards for coliform content or pH, and water-borne parasites are a major health problem. There is no testing for pesticides and other chemicals that are likely present as well. Much of the riparian zone of the rivers supplying water is deforested, with direct access for farm animals to the water (Vieira et al. 2007). Reforestation could potentially improve water supply for a lower cost than filtration and purification plants, as was the case for New York City (Chichilnisky and Heal 2000). One must be cautious when charging individual households the full cost for water provisions, however, because water is essential and non-substitutable. Increasing water prices can potentially cause serious financial difficulties for the poor, who may receive the greatest marginal utility from clean water, but have the least capacity to pay.

Flood regulation in contrast is a pure public-good service. If a forested watershed reduces flooding, there is no way to exclude specific groups or individuals in the floodplain from benefitting from this service, and one beneficiary's use does not leave less for others. The spatial distribution of flood regulation and hence the beneficiaries are easily identified, but there is no collective institution that represents solely those beneficiaries. In general, municipal, state and federal governments respond to floods with assistance for flood victims and rebuilding of public infrastructure and hence are the appropriate collective institutions to pay for the reforestation which can reduce the incidence and severity of both flood events and the associated landslides that cause much of the damage. However, since watersheds typically cross numerous municipal borders in Santa Catarina, some form of state or federal payment may be most appropriate. To more accurately target revenue capture, it would be possible to impose a surtax on land in floodplains. We do not currently know of any PES schemes for flood regulation in Santa Catarina.

17.5.1.3 Carbon Sequestration

The primary goal of carbon sequestration is to provide climate regulation, but the two are distinct services with distinct characteristics (Farley et al. 2010b).

Climate regulation is an example of a pure global public good, both non-rival and non-excludable, so markets will not provide it. The global community must do so collectively. One possibility is for global institutions such as the Global Environmental Facility to finance climate regulation projects directly (UNDP-GEF 1998). The GEF is in fact financing relevant projects in Brazil, including a riparian forest restoration project in São Paulo (World Bank 2005). However, funding is based on grant proposals, reviewed by the centralised GEF bureaucracy. Grant writing skills may be more important than project viability, and the resources dedicated are negligible relative to the scale of the problem (Farley et al. 2010c).

Carbon sequestration in contrast is rival: If one country or firm uses an ecosystem's carbon sequestration capacity, there is less left for another to use. Collective institutions such as the Kyoto Protocol or the European Union are capable of making carbon waste absorption capacity excludable by capping the total amount of carbon that can be emitted and then auctioning off or assigning the right to emit in the form of emission certificates. Such caps allow price rationing of existing absorption capacity and also allow firms in relevant[4] Kyoto Protocol signatory countries to pay for carbon sequestration if it is cheaper than purchasing emission certificates or reducing emissions. This has led to the emergence of carbon markets. The price of carbon however does not reflect the marginal benefits of carbon sequestration, but rather the political will to cap emissions. Existing caps are far too lenient to prevent runaway climate change (IPCC 2007), and carbon prices are correspondingly low. Furthermore, transaction costs to negotiate, monitor and enforce sequestration projects can be very high, particularly in the case of small family farmers.

While Santa Catarina's new PES law includes carbon sequestration as one of the targeted ecosystem services, the benefits of the service clearly cross political boundaries, so Santa Catarina is likely to underinvest in its provision in the absence of national or global agreements that force it to do so. Only more stringent global agreements are likely to create adequate payments for carbon sequestration.

17.5.1.4 Biodiversity Conservation

Agroecology practices can enhance both species richness and abundance in a variety of agricultural landscapes (Batáry et al. 2011), and high yielding agroforestry projects can also promote high biodiversity (Clough et al. 2011). Furthermore, genetic information, essential for breeding new varieties of plants and animals capable of improving yields, ecosystem services and resilience, is a critical input into agroecology schemes. The Santa Catarina PES scheme includes biodiversity as a targeted service.

There are four basic types of PES schemes for biodiversity, reflecting in part the distinct physical characteristics of different aspects of biodiversity: private payments for bioprospecting rights to genetic information, biodiversity offsets, conservation

[4] Annex I countries, which are the industrialised nations, required to reduce emissions (UNFCCC 1998).

financing by collective institutions (including governments, NGOs and international institutions) that target the general public-good benefits of biodiversity, and private payments for biodiversity-friendly products (Landell-Mills and Porras 2002). Each of these has different characteristics and different mechanisms for collecting revenue.

Though genetic information is non-rival, global institutions make it excludable and hence amenable to private sector PES schemes. Clear laws and policies concerning genetic information facilitate such market-like transactions (Landell-Mills and Porras 2002). However, making genetic information freely available to all agroecology projects is required to maximise its value. Market payment schemes may provide some incentive for protecting biodiversity, but also reduce its value. National efforts to protect genetic information can further reduce its value by leading to restrictions on ecological research (Ten Kate 2002). Genetic information is best treated as a global public good, with global collective institutions contributing to its provision (Farley et al. 2011).

Biodiversity offsets function much like carbon offsets. A collective institution limits the total amount of habitat (e.g. wetlands) that can be converted for individual property owners or for society as a whole. Someone can exceed this limit only if they pay for restoration or conservation elsewhere. The Brazilian Forestry Code (BFC) currently permits such markets in legal reserves (RL). One major problem with such markets is that regulators are almost solely responsible for compliance; providers have an incentive to provide and purchasers to purchase the lowest quality that meets regulator standards (King and Kuch 2003). Another problem is the lack of incentive to engage in such markets when the BFC is not enforced.

Collective institutions currently finance most biodiversity conservation, as is appropriate for the largely non-rival, non-excludable global public good benefits it provides. The GEF is the main source of multilateral financing for biodiversity conservation, but is only able to solicit voluntary contributions from primarily wealthy nations. Global NGOs also play an important role, but collect only voluntary payments primarily from individuals and foundations. As a result, current global expenditure on biodiversity conservation are in the neighbourhood of $10 billion annually (Pearce 2007), while an estimated $317 billion/year would be required to maintain global biodiversity and evolutionary potential (James et al. 2001). Balmford et al. (2002) estimate that the social returns on the first $45 billion in annual investments would be 100:1. This suggests the need for a collective institution capable of mandating payments from all beneficiaries with ability to pay, essentially the wealthy nations, but no such institutions yet exists.

In summary, multiple funding streams are available for the different ecosystem services provided by agroecology. However, each one of them falls short of what would be required for optimal provision of a given service, much less for the optimal provision of all the ecosystem services generated by agroecology. The solution it seems would be to bundle the payments for all of these ecosystem services to generate the revenue necessary to fund the large-scale adoption of agroecology. It may cost little more to provide multiple services than to provide a single one (Venter et al. 2009). Both carbon markets and the GEF demand additionality, which is to say that one must prove the activities would not have occurred without the payment.

Since no single payment stream is likely to cover the full opportunity costs of changing land uses, a case can be made for the additionality for each separate stream. Even if it proves possible to bundle the revenue flows from each service, the challenge remains of investing the revenue where it is most capable of promoting agroecology. This will require particularly effective disbursement mechanisms.

17.5.2 Disbursement Mechanisms for Payments

While some PES schemes target community groups and cooperatives, and Brazil's ICMS ecológico targets municipalities, much of the literature on PES suggests that the gold standard is payments to individual landowners contingent upon service provision (Wunder et al. 2008). However, the appropriate recipient depends on the nature of the investments needed to promote the desired land uses, on transaction costs, and on the likely durability of the payments, which in the case of payments for public goods depend largely on political will. Furthermore, making payments contingent upon service provision will only work when the level of investment required to adopt the desired land use can be financed entirely by providers prior to receiving compensation.

As pointed out above and as discussed in the literature (De Schutter 2010; IAASTD 2008; Vanloqueren and Baret 2009; World Bank 2007), the broad dissemination of agroecology is best promoted by investments in public goods. Agroecology demands intensive knowledge of local ecosystems, cultures and markets. It is best spread from farmer to farmer, catalysed and facilitated by agricultural extensionists. The major requirements for disseminating agroecology are investments in R&D, agricultural extension, infrastructure required to bring products to market, and low-risk, low-interest financing mechanisms. Payments to individual farmers do little to provide these services, especially if they are contingent upon provision. Public sector investments are required.

Since the public goods provided by these investments cross political boundaries, payments for these investments should flow from those governments or collective institutions that benefit to those that will provide the services, supplementing resources invested by the latter. This is known as an intergovernmental fiscal transfer and was originally proposed for investments in cross-boundary public goods, not as payments for goods received (Olson 1969). Investments in agroecology promise very high returns in both crop yields and ecosystem services, but are risky. For governments in the regions providing the services, the risk is that these investments will provide lower monetary returns than those generated by public sector investments in more conventional agriculture. For the governments in the regions receiving the services, the risk is that agroecology practices will not be adopted or will not generate the ecosystem services desired. If the efforts succeed, both sides can benefit, but the initial risk should be shared, which is in fact another goal of intergovernmental fiscal transfers (Bird and Smart 2002). We therefore propose a redesign of PES as a form of public sector venture capital, in which wealthy countries and national governments that benefit from the ecosystem services agroecology provides transfer resources to less wealthy countries and local governments otherwise unable to fully finance the necessary public sector investments.

The goal of sharing risk should also scale down to the local level. Farmers investing in agroecology are risking the known returns from their current practices and must invest both their land and labour. National or local governments should provide low-interest, minimal-risk loans to farmers adopting agroecology. Repayment schedules and interest rates would be determined by the increase in market returns attributable to agroecology. Brazil has already begun to provide low-interest loans for agroforestry, but in insufficient quantities to restore the Atlantic Forest as rapidly as may be required to avoid crossing critical thresholds.

For the proposed transfers to be effective, recipient governments should have 'a clear mandate, adequate resources, sufficient flexibility to make decisions and [be held] accountable for results' (Bird and Smart 2002, p. 899). The clear mandate must be to invest these resources in the public goods required to promote agroecology. Flexibility is increased by maximising the input of local governments into investment decisions, based on the needs of their constituents. For international transfers, accountability for results is more difficult. Since many of the ecosystem services are local and regional, the governments providing the services would certainly have every incentive to succeed even without accountability. To increase this incentive, recipient governments could be allowed to sell a share of the carbon sequestered on carbon markets. Carbon payments to governments would incur far smaller transaction costs than payments to individual landowners, especially when land holdings are small and land tenure is weak.

Our suggestions are partially modelled after Brazil's ICMS ecológico, in which some Brazilian states transfer a share of the state sales tax to municipalities according to how effectively they provide ecosystem services. The approach has been very cost-effective, with minimal transaction costs. This system however rewards states after they have protected ecosystems and does not provide the up-front resources necessary to do so (Farley et al. 2010c; Ring 2008).

There are two final reasons to promote agroecology over more conventional ecosystem restoration. First, if the political will for PES falters in the future, maintaining agroecosystems is justified by their higher returns even in the absence of payments. Second, food is a globally traded commodity. If all Brazilian farmers complied with the national forestry code, it could have an impact on global food production, leading to dramatic price increases due to the inelastic demand for food. Ecosystems around the planet must be restored, and agroecology may be the only approach that will simultaneously allow continued food production. Those governments that finance agroecology will benefit both from more ecosystem services and lower food prices.

In summary, there are no longer acceptable trade-offs between agriculture and ecosystem services: Both are essential and at risk. Agroecology may be uniquely capable of providing both. There are real costs to promoting agroecology that someone must pay, but any payment scheme must recognise that many of the services provided as well as the resources required to provide them are both public goods.

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Chapter 18
Towards an Institutional Approach to Payments for Ecosystem Services: Perspectives from Two Nicaraguan Cases

Gert Van Hecken, Johan Bastiaensen, and Frédéric Huybrechs

18.1 Introduction

The concept of payments for ecosystem services (PES) has attracted growing attention in both academic and policy circles. One of its applications lies in improving land-use patterns. In this context, the premise of the approach is appealing; farmers, who have little, if any, incentives to consider the environment in their land-use decisions, may be encouraged to do so through direct payments from buyers of ecosystem services (ES). It is often argued that the conditional market-based PES approach may be more effective than alternative environmental policy approaches (Wunder 2005). The presumption of PES's superiority over other approaches to conservation is, however, not unequivocal (Redford and Adams 2009). This chapter highlights some of the weaknesses of a market-based, 'Coasean', conceptualisation of PES and questions its effectiveness and viability as a stand-alone governance alternative. An analysis of two case studies dealing respectively with the demand and the supply-side perspectives of PES in the region of Matiguás-Río Blanco in Nicaragua shows that the Coasean approach largely fails to take account of the complex and inevitable interactions between PES mechanisms and the broader institutional environment.

After a brief conceptualisation of PES, we describe the research setting and the analytical framework, whereby the supply and demand sides of the 'Coasean' PES approach are scrutinised in two case studies. This is followed by an elucidation of the research design and methods. In the final sections, we present the results and argue that the findings support the case for a more integral institutional approach to PES as part of a broader (environmental) governance structure.

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18.2 Theoretical Background

PES is generally considered to be a market-based instrument of environmental governance (Engel et al. 2008). It is founded on the assumption that environmental degradation is commonly caused by the failure of markets to take due account of the environmental externalities of private economic activities. The underlying notion is that private landowners will incorporate conservation or service provision considerations into their decision-making to the extent that this coincides with their own direct economic interests. Hence, the introduction of payments for positive externalities stemming from environmentally sound land-use practices should lead to such a behavioural change (Engel et al. 2008; Wunder 2005). Theoretically, the payments would be made by the direct beneficiaries, creating a market where demand and supply for ES meet. Thus, Wunder defines PES as a *voluntary* transaction where a *well-defined* ES (or a land use likely to secure that service) is 'bought' by an ES *buyer* from an ES *provider* if and only if the ES provider secures ES provision (2005:3). This mainstream approach to PES is based on a popular interpretation of the 'Coase theorem', where it is assumed that, in the presence of sufficiently low transaction costs and clearly defined and enforced property rights, individual and voluntary bargaining through the market will lead to the most efficient allocation of externalities[1] (Coase 1960).

The origin and popularity of the instrument are traceable to a general dissatisfaction with traditional top-down regulatory, community-based and educational approaches for being largely ineffective in halting environmental degradation (Baland and Platteau 1996; Ferraro 2001; Pagiola et al. 2002). Although the market-based PES approach has been put forward as a more effective and efficient alternative (Wunder 2005), its rapid embrace by academics and policymakers comes with rather limited empirical evidence (Redford and Adams 2009). Recent research seems to indicate that most PES initiatives generate little additional environmental stewardship and supply of ES so that the envisaged efficiency gains are unmet (Kosoy et al. 2007; Muñoz-Piña et al. 2008; Muradian et al. 2010; Pattanayak et al. 2010; Robalino et al. 2008).

This chapter focuses on a number of issues concerning the implementation of market-based PES. In particular, it presents two complementary studies, from respectively a supply and a demand perspective. The findings raise substantial doubts about the effectiveness and viability of the 'Coasean PES approach' as a stand-alone

[1] In this chapter, the terms 'Coasean' and 'market based' refer to a governance model and approach to PES that builds mainly on the belief that compliance and individual or collective action should be accomplished through the use of decentralised and individual price incentives. The term 'market based' comes from Uphoff's (1993) distinction between three main governance models (bureaucratic or command-and-control, market based and community based or voluntary action models). Each model uses different instruments and underlying philosophies to stimulate compliance and collective action. In the market-based model, 'decisions are left to individuals to calculate private advantage without reference to broader interests of the public good' (ibid: 610).

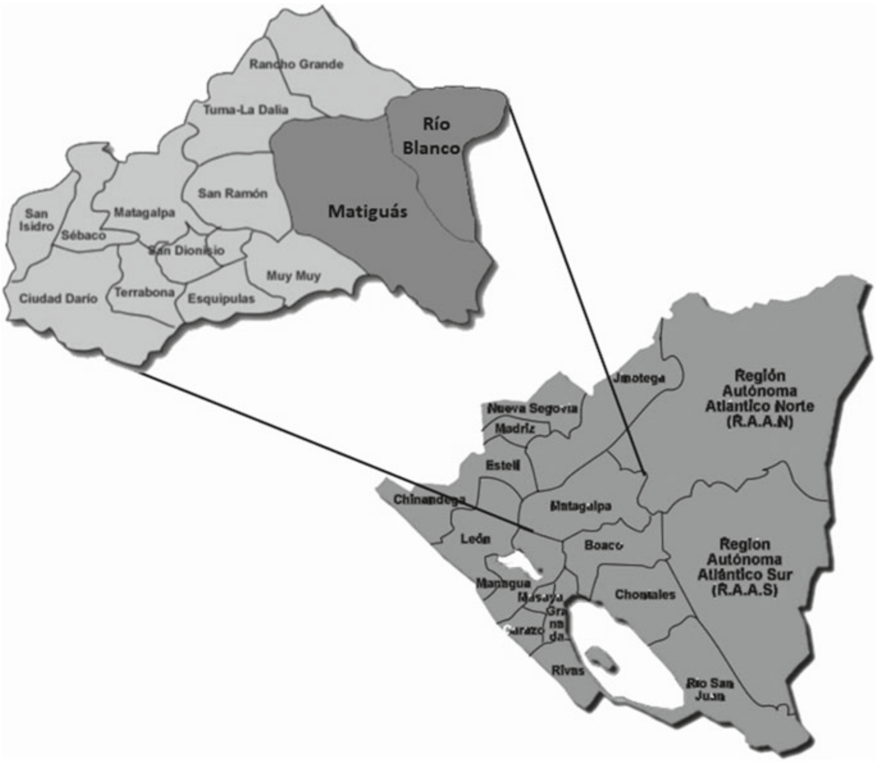

Fig. 18.1 Matiguás and Río Blanco in the department of Matagalpa, Nicaragua (Source: Van Hecken 2011)

governance alternative, disconnected from broader institutional processes.[2] Hence, the conclusions lend support to the emerging consensus that PES should be integrated into a broader and more hybrid institutional governance approach (e.g. Corbera et al. 2009; Farley and Costanza 2010; Muradian et al. 2010; Vatn 2010).

18.3 Description of the Research Setting and Case Studies

The two case studies presented in this chapter were conducted in the municipalities of Matiguás-Rio Blanco, in the Department of Matagalpa in Nicaragua (Fig. 18.1). The municipalities belong to the region with the highest poverty incidence in

[2] The term 'institutions' refers to the prevailing 'rules of the game' in a given society, consisting of both the formal and informal human-devised constraints that govern—but not necessarily determine—individual behaviour and structure social interactions (see, e.g. North 1990).

Nicaragua (INIDE 2005). They are part of the old agricultural frontier, whose colonisation began in the 1920s and 1930s (Maldidier and Marchetti 1996). Growing demand for land in the more densely populated Pacific regions of Nicaragua pushed both peasants and landlords towards the forested frontier in search of pasture for extensive cattle raising. It was the beginning of a process of deforestation, which gained momentum between 1950 and 1980 with the opening of US export markets for Nicaraguan meat. The guerrilla war of the 1980s, whose main internal front was located in the Matiguás-Rio Blanco region, temporarily interrupted the deforestation process, but it picked up speed and intensity again after the peace agreements of 1990. According to local estimates, the forested area in Matiguás has shrunk by over 40% in the past 20 years, to make way primarily for cattle farming (INIFOM 2004). Today, the region under study encompasses two nationally protected areas: the Quirragua and the Cerro Musún Nature Reserves.

Nicaragua has a long tradition of centralised command-and-control natural resource management, with strong emphasis on prohibitive measures and the creation of protected areas (Ravnborg 2010). Environmental governance in Matiguás-Rio Blanco conforms to such a top-down regulatory approach. However, excessive reliance on poorly enforced command-and-control measures has stood in the way of an effective local framework for environmental protection. The 'protected' reserves lack rangers or police officers so that they are effectively 'paper parks' where deforestation and natural degradation continue.

The prevailing perceptions of conservation also influence how environmental governance is locally embedded and how it interacts with both formal and non-formal institutions. Land entitlement continues to depend on the—albeit now largely imaginary—act of colonisation, i.e. the notion that 'wild and unproductive' expanses of forest are tamed and turned into land for arable farming and cattle production. Local farmers refer to their clearing of the forest through hard labour as introducing '*mejoras*' (improvements), from which they derive property rights and hence entitlements to compensation for expropriation, even within conservation areas (Bastiaensen et al. 2006). Moreover, the very nature of the agricultural frontier also makes it an 'institutional frontier', distant from the country's established infrastructure, with a limited state presence and an absence of extensive social networks, mutual trust and security (Ravnborg 2010). Yet, despite the geographical and institutional remoteness of Matiguás-Río Blanco from the urban society of the capital and the 'developed' world, ecological messages about endangered species, climate change and increasing pressure on water and forest resources have penetrated the local cultural arenas.

The traditional top-down approach to environmental management in Nicaragua has resulted in a general perception of nature conservation as an almost militaristic engagement (see also Ravnborg 2010). In the predominant urban narratives of environmental degradation, the onus of blame for the destruction of the forest rests firmly with the 'malicious' farmers, whose activities should therefore be subject to stricter regulations and punitive measures. Moreover, patron-client relationships prevail in this area where just a few leading actors (i.e. patron gatekeepers) mediate the information and resource flows to relatively isolated, dependent individuals (i.e. clients) and thereby dominate and manipulate local collective action (or inaction). These conditions fuel feelings of distrust, widespread opportunistic behaviour and deep-rooted pessimism about the possibility of breaking the negative, non-cooperative and non-rule-abiding dynamics that characterise vertical patron-client governance (Putnam et al. 1993).

It is within this context of weakly enforceable environmental protection and 'hard' trade-offs between conservation and development that the first case study is situated. The Regional Integrated Silvopastoral Management Project (RISEMP) was a World Bank- and Global Environment Facility (GEF)-funded pilot experiment designed and implemented to encourage the use of silvopastoral practices in cattle raising[3] (GEF 2007; Vaessen and Van Hecken 2009). Although silvopastoral practices generate long-term on-site private benefits, particularly in the current context of rising opportunities in dairy production, they are generally deemed unattractive to farmers. The main barriers are assumed to be the significant investment of capital and labour required, the time lag between new practices and higher productivity and a lack of know-how (Dagang and Nair 2003), hence the notion of introducing PES and/or technical assistance (TA) in order to stimulate the switch to the envisaged land-use practices.

The payments in the RISEMP were tied to an 'environmental service index' (ESI), which attributed ES values to 24 different land-use types in terms of their contribution to biodiversity and carbon capture. Farmers were paid an annual sum of US$75 (under a 4-year scenario) or US$110 (under a 2-year scenario) per incremental ESI point/ha compared to their baseline ESI balance (which was calculated on the basis of remote sensing imagery) (Pagiola et al. 2007). Payments were based not on the value of the generated ES to their potential users but—rather more pragmatically—on the estimated opportunity costs of the environmentally more attractive land uses and the corresponding partial subsidy required to 'tip the balance' in favour of investments in silvopastoral practices.

As analysed below, the RISEMP is generally considered to have been a successful pilot experiment (Pagiola et al. 2007). It was, however, limited in time, with farmers receiving payments over either 2 or 4 years (ibid). The temporary nature of the project raised concern about the lack of continuity of the PES, leading Pagiola et al. (2007) to suggest that a local PES system tied to an improved drinking water supply in urban Matiguás could potentially ensure long-term PES funding. The urban drinking water supply, which has come under increasing pressure through upstream agricultural activities, is a concern in Matiguás. Urban Matiguás currently relies on water from the Cusiles River, which originates in the upstream area of Quirragua. Although Quirragua is a nature reserve, about 70% of the constituting land is privately owned by around 60 farming households who use it mainly for crop growing and cattle raising (MARENA 2010). The negative consequences of these

[3] Silvopastoral practices 'combine fodder plants … with trees and shrubs for animal nutrition and complementary uses', and include living fences and 'cut and carry systems' (Pagiola et al. 2007: 375).

upstream agricultural activities are perceived locally as a threat to the downstream urban tap water supply, indicating a need for more effective, negotiated environmental governance. Preliminary ideas about a local watershed PES scheme were discussed by the municipal authorities, local NGOs and a new association of Quirragua farmers, who represented themselves as defenders of the reserve. Such a scheme would function as an exchange between the approximately 2,000 households of urban Matiguás and the 60 farmers of Quirragua, based on the perceived threat that upstream agricultural activities pose to local water provision on the one hand and the expectation that local water users could ensure long-term PES funds on the other. In this context, the second case study focuses on the demand side by gauging the willingness to pay of local water users for improved water provision and their preference for different policy options.

18.4 Research Design and Methodologies

Both case studies—i.e. the reassessment of the RISEMP project in Matiguás-Río Blanco and the measurement of willingness to pay (WTP) and policy preferences for improved water services in urban Matiguás—adopted a mixed method approach, combining qualitative and quantitative research techniques. It speaks for itself that the specific research design was adapted to the topics under study.

18.4.1 A Supply-Side Perspective: RISEMP in Matiguás-Río Blanco

The GEF and World Bank conceptualised RISEMP as a PES pilot project involving the recuperation of degraded pasture through more intensive silvopastoral farming systems. The project design encompassed a randomised controlled trial (RCT), which, in the Nicaraguan setting, focused on the behaviour of 123 randomly selected farmers. Some were exposed to one of the four scenarios of project treatment (ESI-based PES over 2 or 4 years, with or without complementary TA); the others were—in theory at least—unconnected to the project (see infra) and hence formed the control group.

The proposed study consisted in an in-depth reassessment of the research results obtained by the RISEMP research and project team (Pagiola et al. 2007, 2008). It encompassed a review of the articles and reports produced, as well as an analysis of the data from the original project surveys and a reinterpretation of the RCT experiment. This desktop analysis was complemented with an additional 2-month qualitative field study after project termination. The research activities in the field included interviews with key observers, participant observation and in-depth responsive interviews (Rubin and Rubin 2005) with 33 participating and 3 non-participating farmers. The farmers were selected on the basis of snowball and maximum variability sampling (Glaser and Strauss 1967), in which all farmers were attributed high, low and median values for variables such as type of participant group, payments received, farm size, location and accessibility, gender, herd size and type of land-use changes.

18.4.2 A Demand-Side Perspective: Hydrological PES in Matiguás

The second study focused mainly on the demand side of a hypothetical locally financed PES system within the upstream-downstream context of water governance in Matiguás. Here, the underlying logic of the mixed method approach was to first gather information about the institutional context of water and environmental governance, including perceptions of and knowledge about upstream-downstream interactions. This information was subsequently used to design a survey and to specify scenarios for a contingent valuation (CV) study. *Ex post*, it also served for the contextualisation and interpretation of the results of the CV study. The qualitative research was conducted over 6 months during 2008 and 2009 and consisted mainly in in-depth responsive interviews and three additional focus group interviews. Over 25 key informants from different local institutions and organisations were interviewed, ranging from consumer group representatives to (central) government institution delegates and from political party secretaries to Quirragua farmer cooperative presidents.

The qualitative research was complemented with a quantitative household survey in downstream urban Matiguás. Structured interviews—sampled on the basis of a geographically stratified random selection of households from a list of 1,955 plots on the latest urban cadastral maps—were conducted in August 2009 by ten (five female and five male) local university students who received 4 days of training before entering the field. A total of 1,015 surveys were completed (corresponding to a 74.5% response rate).

The survey inquired into households' views on the existing water system, their water uses and consumption patterns, expenditures on both tap and bottled water and socio-demographic characteristics. It also considered various factors impacting on the acceptance of a PES mechanism through questions designed to elicit households' perceptions and attitudes regarding local environmental degradation and entitlements, the existence of upstream-downstream externalities and preferred solutions to them.

Downstream willingness to pay (WTP) for improved water services was investigated through a split-sample, single-bounded referendum CV section in the survey. The CV method measures the price that respondents are willing to pay for changes to the provision of a publicly provided good, and it is used increasingly commonly in water supply research in developing countries (Whittington 2002; Vásquez et al. 2012). In the present instance, the method was used mainly for the purpose of evaluating whether preferences are stable under two different policy scenarios (Farley and Costanza 2010: 2063). Each survey respondent in urban Matiguás was randomly confronted with one of four (two-by-two) CV scenarios (Table 18.1). All scenarios involved a guaranteed and uninterrupted supply of safe-to-drink tap water. Half of

Table 18.1 Contingent valuation scenarios used in urban Matiguás surveys

		Type of improvement		
		Infrastructural improvement	ES protection (PES)	Total
Type of administration	Current water company	$n = 240$ Average fee = 98.0	$n = 244$ Average fee = 100.2	$n = 484$ Average fee = 99.1
	Municipality	$n = 249$ Average fee = 99.1	$n = 245$ Average fee = 100.7	$n = 494$ Average fee = 99.9
	Total	$n = 489$ Average fee = 98.6	$n = 489$ Average fee = 100.5	$n = 978$ Average fee = 99.5

Average fee presented to respondents is expressed in Nicaraguan C$ (US$1 is equivalent to about C$20.5 as on August 2009). Total observations are based on households who answered the CV question ($n = 978$)

the respondents in the household sample were presented with a scenario where this goal was to be achieved through infrastructure improvements (new pipes, tanks and filters). The other households were confronted with a scenario of improved upstream land-use practices to be achieved through conditional monthly payments (PES) to upstream Quirragua farmers. The CV study also controlled for an administration variable (management by the current departmental water company or by a new municipal enterprise), as the municipality had recently expressed an interest in managing the local water supply.

All respondents were told that funding of the proposed project would require every household to pay an additional monthly fee, to be added to the current tap water bill. The proposed additional fee varied randomly across households from C$20–180, with intervals of C$20.[4] The 1,015 respondents were then asked to vote in favour of or against the project. This elicited 978 responses, a number that was reduced on account of missing variables to 842 for estimating the logistic model.

Factors influencing the underlying WTP for improved water services are identified through an econometric model in which WTP is assumed to be a function of specific household attributes and perceptions (Vásquez et al. 2009) and to follow a log-linear form:

$$LNWTP = X\beta + e \tag{18.1}$$

where *LNWTP* represents the natural logarithm of household WTP for a change in water services. X is a vector of covariates including treatment variables (indicating different improvement and management scenarios), household income, respondent's perceptions and other relevant household characteristics. β is a vector of coefficients to be estimated, and e is the stochastic error term. The referendum format used in this study does not allow for direct observation of WTP. However, *LNWTP* can be

[4] The relevance of these amounts was determined during the preceding qualitative research and pilot survey phase. C$ refers to Nicaraguan Córdobas; at the time of fieldwork, US$1was equivalent to C$20.5.

indirectly identified given that respondents are expected to answer favourably to the referendum question only if the household's WTP is greater than or equal to the fee (*LNFEE)* presented in the contingent scenario. The direct WTP parameters from Eq. 18.1 can then be calculated by consecutively dividing the estimated coefficients of the independent variables by the estimated coefficient of *LNFEE* and by switching the sign of this resulting parameter (see Cameron 1988; Van Hecken et al. 2012).

18.5 Reassessment of the RISEMP Experience

The RISEMP project design is implicitly based upon a model of individual utility maximisation where it is assumed that the behaviour of individual decision-makers can be changed, *ceteris paribus*, by capitalising on (a) their economic incentives (conditional PES) and (b) their technological knowledge and thus their production function. The design for the RCT experiment builds on the same assumptions. However, in the course of the implementation phase, a number of things went wrong—at least from a scientific experimental point of view. First, selection of the participants in the treatment and control group was biased rather than random. Inspired by their social objectives and motivated by concerns about the manageability of their relationships with the villagers, the implementing development organisation Nitlapán selected the participants for the treatment groups (i.e. the recipients) from the poorer and medium-sized farmers, while the control group was selected from the richer farmers. This obviously violated the principle of random selection, which was key to the RCT research design. Second, there were further practical problems with the experimental design, as unforeseen 'treatment diffusion' occurred between the groups. In practice, farmers who were excluded from the TA treatment regularly substituted for other farmers in the TA workshops if the latter were unable to attend. Perhaps more importantly, although silvopastoral farming had previously not been entirely unfamiliar to the local farmers, the presence of such a substantial project in three relatively small rural communities meant that inevitably multiple 'informal' exchanges were taking place about the promoted technologies. The research objective to test for the additional effect of TA thus became problematic.

The problems with the RCT design led the RISEMP research team to drop the control group from their analysis and to focus on an *ex ante-ex post* comparison of the treatment groups (Pagiola et al. 2007). Project data shows a substantial change in land use, in particular a significant decrease in 'degraded pastures' (from 30.9 to 10.1% of the total farm area), which were mainly replaced by improved pastures with trees and fodder banks. Furthermore, the use of living fences almost quadrupled, and about half of the annual crop area was replaced by scrubland. Pagiola et al. (2007) attributed these changes mainly to PES incentives, with TA playing a secondary role in the adoption of silvopastoral practices. As will be demonstrated, a reassessment of the RISEMP experience supports the finding that the project had a substantial impact. What is more, our *ex post* farm visits confirmed that many of the silvopastoral land-use changes were maintained even after termination of the RISEMP.

However, the interpretation that this impact is attributable primarily to the monetary incentives through PES is debatable.

A first indication comes from the land-use change observed in the control group. We concur that this group was biased towards the richer farmers and therefore cannot serve as the counterfactual as intended in the initial RCT design. However, this does not mean that this group should be excluded from the analysis, especially since the results obtained are surprising. The highest reduction in degraded pastures was actually observed in the control group. Furthermore, living fences also expanded most extensively in this group (Van Hecken and Bastiaensen 2009). A comparison of the control and treatment groups shows no significant effect of PES and/or TA on land-use change, but due to the selection bias of the control group, one cannot conclude on this basis that there is no effect. Given the aforementioned result, it would certainly seem problematic, though, to hypothesise (as based on the *ex ante-ex post* comparison) that the observed changes in land use within the participant groups of the RISEMP are attributable primarily to PES and TA. Quite similar changes did after all occur among the richer control group for apparently different reasons.

Our field interviews reveal that all farmer groups have a key underlying motivation for implementing silvopastoral practices, related directly to new opportunities in the dairy sector. The growing importance of fresh milk production provides an incentive for intensified dairy farming, particularly on farms with transport links to the milk collection centres. Booming national milk consumption and the improved accessibility of milk collection centres and (semi-)industrial cheese factories have led to greater demand for milk and significant increases in regional milk prices. These evolutions justify increased investments in farm infrastructure (milking areas, galleys …), improved (denser) pastures and genetically adequate dairy cattle, which also require more trees for shade since they are less resistant to excessive heat. Milk collection centres also reward a constant year-round supply of milk. This implies a need for fodder crops that can mitigate summer food shortages. Improvements in relation to degraded pastures, fodder crops and additional (living) fences all contribute to increased milk productivity and to a more balanced milk output.[5]

Despite doubts about the differential impact of the RISEMP activities on experimental participants and non-participants, we cannot conclude that the project had no impact. Compared to similar villages in the region, changes in land use towards silvopastoral practices have occurred more rapidly and substantially in the RISEMP area. Importantly, though, many farmers stressed the decisive role of TA. The TA as such and the social momentum it generated were said to have stimulated experimentation with new practices or the expansion of already-known land-use improvements. It also lowered the perceptions of risks, impacting on the farmers' motivation to engage strongly with silvopastoral production technologies.

Although the unavoidable TA treatment diffusion to non-treatment groups is considered to be a problem in the RCT experiment, it does show that the 'silvopastoral

[5] Detailed analysis of the differential constraints of distinctive types of farmers reveals additional variations in investment strategies. See Van Hecken and Bastiaensen (2009, 2010a) for details.

noise' generated by the RISEMP created a widely felt motivational impetus in favour of silvopastoral intensification. As de Haan and Zoomers (2005) have argued, this indicates that rural change processes such as those engendered by the RISEMP initiative are not so much a matter of isolated individual innovations but rather the outcome of collective pathways of change. These collective pathways result from the emergence and articulation of a sufficiently strong social momentum crystallising into interrelated, mutually supportive individual and collective action (ibid: 40–44). While village-level interactions made it impossible to target the TA treatment and to isolate its cognitive-motivational effect on particular producers, it did generate a broad and substantial village-wide impact.

18.6 Demand-Side Perspective on Hydrological PES in Matiguás

Long-term sustainability of PES funds is often considered an important stumbling block in PES implementation (Engel et al. 2008; Pagiola et al. 2002). In the Matiguás-Río Blanco context, Pagiola et al. (2007) claim that '[w]ater services offer the most promising avenue for financing long-term PES programs'. However, the willingness of water users to compensate upstream farmers in order to safeguard the provision of water services is not self-evident. Therefore, the study inquired into the feasibility of such payment schemes in downstream urban Matiguás.

The survey data reveal serious problems with the water supply. Households say they have a reasonable 14.2 daily hours access to tap water during the dry season but only 3.8 daily hours during the rainy season, when heavy precipitation results in flooding and sedimentation, causing temporary shutdowns of the supply system. In terms of water quality, about 85% of households think their tap water is polluted, and about half of the households treat it before consumption, usually by adding chlorine or boiling it. Almost all respondents believe Matiguás is suffering the negative consequences of deforestation, and two-thirds feel water resources are badly managed, particularly by farmers. The survey also indicates that respectively 78 and 86% of downstream respondents consider agricultural activities in the upstream area to negatively affect water quantity and quality. Furthermore, 87 and 85% of the respondents agree that reforestation of the upstream watershed would improve respectively the water quantity and quality. Finally, two-thirds of urban households feel the best way of improving water quality and quantity is to invest in ecosystem protection rather than in improvements of the existing water supply infrastructure.

The willingness to pay of the inhabitants of Matiguás for an improved water supply through infrastructure investments or ES protection was determined by means of the CV method outlined under Sect. 18.4. Table 18.2 provides a description and the summary statistics of the variables used to estimate the model. The dependent variable *VOTE* has value 1 for respondents who voted in favour of the proposed scenario and 0 otherwise. *LNFEE* reflects the natural logarithm of the randomly

Table 18.2 Variables description and summary statistics for all observations

Variable	Description	Mean	Std. dev.
VOTE	Respondent's vote in the CV scenario (1 = in favour; 0 = against)	0.55	0.50
LNFEE	Natural logarithm of the additional fee charged for water service improvement in the CV scenario	4.42	0.67
PES	Respondent is presented with the payments for ecosystem services scenario in the CV scenario (1 = PES scenario; 0 = infrastructure scenario)	0.50	0.50
CITY	Respondent is presented with the decentralisation scenario (transfer of water administration to municipality) (1 = municipality administration; 0 = current water company administration)	0.50	0.50
INCOME	Aggregate household income in C$	2946.95	2788.85
EDU	Education of respondent (in years of schooling)	7.24	4.50
AGE	Age of respondent (in years)	39.09	13.94
FEMALE	Sex of respondent (1 = female, 0 = male)	0.82	0.38
HHSIZE	Number of household members	4.80	2.29

assigned fee in the scenario and is expected to have a negative coefficient, as a higher fee is assumed to lower the probability of approval for the proposed project (incentive compatibility). The dummy variable *PES* refers to the two approaches to improving water services according to the split-sample design. The estimated coefficient is expected to be positive if respondents are willing to pay more for the PES scenario than for investments in infrastructure and negative if the opposite is true. The dummy variable *CITY* indicates whether the improved water system is to be managed by the current departmental water company (*CITY* = *0*) or by the municipality of Matiguás (*CITY* = *1*). The variable *INCOME* is also included, and tap water is assumed to be a normal good (i.e. $\beta_{INCOME} > 0$). Other household characteristics incorporated into the analysis are the respondent's years of education (*EDU*), age (*AGE*), gender (*FEMALE*) and number of household members (*HHSIZE*). No specific hypotheses are formulated about the effect of these characteristics on WTP.

The regression results are shown in Table 18.3. The first column displays the 'raw' logit results, while the second column displays the WTP parameters.

The estimated coefficient of *LNFEE* is negative and significant at the 1% level and confirms the incentive compatibility assumption that water demand will be lower at a higher price. More surprisingly, the estimated coefficient of *PES* is negative and highly significant, indicating respondents' higher WTP under infrastructure scenarios as compared to an upstream-downstream PES scenario.[6] This result is unexpected, as the survey shows a majority of respondents to favour upstream ecosystem protection and adequate water management practices over infrastructure improvement as a solution to existing water supply problems. The estimate of the median WTP under the PES scenario is C$99, whereas it is C$207 under the infrastructure scenario.

[6] The results are robust across different model specifications. They are available upon request.

Table 18.3 Estimated WTP regression model

Variables	Regression coefficient	WTP parameter
LNFEE	−0.507	–
	(0.111)***	
PES	−0.375	−0.740
	(0.143)***	(0.323)**
CITY	−0.043	−0.084
	(0.142)	(0.281)
INCOME	0.046	0.091
	(0.030)	(0.062)
EDU	−0.007	−0.014
	(0.018)	(0.037)
AGE	−0.006	−0.011
	(0.006)	(0.012)
FEMALE	0.353	0.696
	(0.165)**	(0.364)*
HHSIZE	−0.008	−0.017
	(0.0314)	(0.062)
CONSTANT	1.561	5.133
	(0.665)**	(0.791)***
Observations	842	
Log likelihood	−557.36	
Pseudo R^2	0.0344	
AIC	1132.71	
BIC	1175.34	
Estimated median WTP infrastructure	C$207.45	
Estimated median WTP PES	C$98.98	

Numbers in parentheses are corresponding standard errors WTP estimates are derived using the Krinsky and Robb (1986) bootstrapping procedure (using 5,000 simulations) ***, **, * imply significance at 1, 5 and 10% levels respectively

This again indicates that respondents are willing to pay considerably more for improved infrastructure than under a PES scheme.[7]

In order to find an explanation for this apparently counterintuitive result, we think it is necessary to consider a model of analysis that goes beyond the rational actor model of decision-making and choice. To this end, the unstable revealed preferences need to be linked with characteristics of the (not necessarily coherent and articulated) local institutional environment. The cultural repertoire of the institutional context, as well as the social relationships within it, informs human perceptions and individual decision-making and is associated with the narratives of the different policy scenarios (Vatn 2009).

[7] The results for the other variables are statistically insignificant or less relevant to the argumentation presented (see Van Hecken et al. 2012 for further details).

An important consideration in the given context is the historical local predominance of a regulatory and repressive framing of environmental governance, whereby farmers are expected to act as responsible caretakers or to risk punitive measures. The notion of a PES system that runs counter to these dominant perceptions generates a new and hitherto unfamiliar frame of reference that may be difficult to embrace by the urban population. Our survey indicates that the majority of urban households (66%) consider farmers to have a limited entitlement over their privately owned land and to have an obligation not to destroy the environment. It provides evidence of a widespread sentiment among urban dwellers that it would be ‘unfair asking us to pay farmers for taking care of the natural resources on their property, as in fact, farmers are already legally obliged to do this’. Several urban water users also expressed that they ‘don’t care paying more for tap water, as long as the money does not go to the [Quirragua] farmers’.

In addition to perceiving PES as unfair, many urban respondents are doubtful that such conditional payments could function in practice. This is apparent from statements such as ‘Why would farmers suddenly start protecting the environment if they have failed to do so for the past twenty years?’ and ‘How could we ever be sure that our money is spent on environmental improvements rather than on more cows?’ Such stated perceptions of unfairness and distrust can explain why the respondents in the CV study attach a negative premium to the PES scenario, as in their perception it would reward the supposed destructors of natural resources. From this viewpoint, it is difficult for urban dwellers to accept PES as an institutional fix or ‘a carrot that makes the stick of regulations more palatable’ (Engel et al. 2008:669).

18.7 Discussion

The results of both the supply and the demand studies indicate that a PES system inevitably interacts with the broader institutional reality of the historical and location-specific context or human ‘space-time’ (Massey 1993) in which it is implemented. Within this institutional environment, three interconnected aspects can be identified: the cognitive-motivational framework (i.e. ‘culture’), the rules of the game (i.e. institutions *sensu stricto*) and the social networks and organisations (Bastiaensen et al. 2004:10). Our findings provide confirmation that all three come into play.

As pointed out in the reassessment of the RISEMP project, a reductionist market-based approach, modelling impact through a change in incentives and/or the production function of isolated individual producers, cannot adequately explain the broad-based adoption of silvopastoral practices in the targeted area. A considerable extent of social interaction is unavoidable, so that the mere notion of individually targeted PES and TA schemes becomes problematic. The emergence of a fundamentally collective pathway leading to a stronger adoption of the familiar silvopastoral-based, intensified dairy production is arguably a much more adequate conceptualisation. Beyond a possible change of production function, the social production of meaning is crucial and has been shown to be related to the substantial 'silvopastoral noise' generated by the project's presence. In this context, knowledge creation should be viewed as a fundamentally collective process in which training, animation and real world actions are (and have to be) connected. The debate should therefore focus not on whether incentives, technical training or motivational education are the best approaches, but on how they ought to be connected to emerging local development pathways.

This notion should be related to the finding that institutional arrangements, such as PES systems, cannot be treated as mere neutral transmitters of monetary incentives, since they also influence and interact with people's intrinsic motivations, which are in turn related to their sense of enjoyment, satisfaction, (social) responsibility and/or obligation (Paavola and Adger 2005; Reeson 2008; Vatn 2005). A number of scholars have pointed to a danger of 'motivation crowding-out' (Frey 1997), i.e. a negative interference of a market logic and its extrinsic price incentives with people's intrinsic motivation (Anderson 2006; Bowles 2008). Motivation crowding-out and the erosion of social norms could potentially undermine the anticipated positive effect of PES on the provision of ES. Indeed, the reported field research found some worrying indications of such an unanticipated effect in certain areas. More specifically, farmers in the Quirragua nature reserve, which plays a critical role in the local urban water supply, were found to have strategically threatened to abandon responsible environmental practices unless compensatory PES were forthcoming.

However, in a context of gradually changing local perceptions, there may also be a positive interaction between extrinsic and intrinsic motivations (see, e.g. Kosoy et al. 2008). The introduction of PES may thus transmit the message that environmental protection is highly valued by outsiders who are therefore willing to pay significantly for it. PES as such could thus induce changes in local perceptions, values and norms concerning 'accepted' and 'desirable' agricultural practices. In such a context, positive 'motivation crowding-in' would appear not to be impossible. It remains quite unlikely, however, that individualised payments alone could contribute to the strengthening and emergence of more environmentally friendly standards and 'social markets' (Martin et al. 2008). Such payments will need to be embedded in broader processes and discourses of change.

The reconsideration of the reductionist assumption that individual decision-making is based on a strict individual rationality and stable independent preferences also sheds light on the surprising and seemingly incoherent choices by the respondents to the CV survey on water demand in Matiguás. It should be emphasised here that motivation is based upon imperfect, subjective and partially collectively informed cognitive models as well as inherited social routines, underlining the crucial role of institutions in the choices made by individuals (Paavola and Adger 2005; Vatn 2005).

While the PES approach focuses on rewards for positive externalities rather than sanctions for negative externalities, little attention is generally paid to the potential implications and the local acceptance of the underlying assumption (Vatn 2010). Yet, the categorisation of externalities is not a merely technical issue (Van Hecken and Bastiaensen 2010b). The case study focusing on the demand side of a hypothetical PES scheme in Matiguás indeed shows that the prevailing perceptions of urban water users, informed by a history of command-and-control policy and narratives of environmentally negligent farmers, geared their preference away from a policy of PES, in favour of infrastructural solutions. At the same time, most upstream farmers feel marginalised from (urban) society: they are expected to bear the burden of new 'green' expectations but receive little support or even acknowledgement for good practices. In this context, Sommerville et al. (2010:1263) rightly assert that 'perceptions of unfairness can undermine the effectiveness of incentives that provide apparent net benefits … at the individual scale [and] can have a substantial impact on the participation of the wider community and thus the efficacy of an intervention'.

18.8 Conclusion

Following a lead from Ostrom and Cox (2010), the case study analysis presented here indicates that due account must be taken of the complexity of society and nature, as well as their interaction in socio-ecological systems. Reducing this complexity in order to make it fit into a market-based model can only be done at the risk of overlooking the broader institutional framework from which it cannot be separated. For this reason, a 'Coasean' PES approach, based upon a reductionist model of the individual rational actor, should not be considered a universal market-based panacea for environmental governance as opposed to supposedly less effective approaches such as government regulation, community-involvement or educational campaigns. Quite the opposite: any PES system is inevitably part of broader historical spatio-temporal dynamics. PES must therefore invariably be designed, analysed and monitored within the context of the power geographies that generate certain institutional logics and organisational forms in the human territory concerned. Vatn (2005) has rightly argued that 'choosing policy instruments is … not simply about changing incentives. … [I]t is about instituting certain logics, about understanding which institutional frames people apply, and about influencing these frames' (ibid: 215). It is about how people rally together to forge collective pathways of change, to develop widely shared principles and rule systems, as well as the motivation and capacity to comply with them, and, if necessary, to monitor and enforce them. PES initiatives cannot and should not be separated from these broader institutional change dynamics. Perhaps such schemes can fulfil an important role in a strategy towards improved environmental governance, but they should not be regarded as a market-based panacea that can miraculously assume away the inevitable complexity of social and environmental change.

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Part IV
The Special Case of Carbon Markets

Chapter 19
A Policy Mix to Finance Protected Areas in Mato Grosso, Brazil

João Andrade, Peter H. May, and Paula Bernasconi

19.1 Introduction and Objectives

The objective of this study is to discuss the potential of a REDD (Reducing Emissions from Deforestation and Forest Degradation) mechanism in combination with several regulatory instruments. This combination of command and control and economic instruments associated with carbon markets could increase provision of ecosystem services beyond the storage of carbon alone, by creation of new protected areas in one of the states of the Brazilian Amazon region. To locate the reader with regard to the specifics of the region, the study initially describes the context and legal framework relevant for environmental protection in Mato Grosso, Brazil. Following this, we apply primary and secondary data to demonstrate the relevance of the proposal for regional policy and practical results of reduced emissions from deforestation.

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Fig. 19.1 Location of Mato Grosso State in Brazil/South America and the Legal Amazon region

19.2 The Challenge of Agricultural Expansion and the Maintenance of Environmental Services

The state of Mato Grosso (MT), located in the central-west region of Brazil, occupies an area of 903,357.91 km². It is the third largest state in Brazil, bigger than Spain and Germany combined. As it lies in the geographic centre of the South American continent, equidistant from the Pacific and Atlantic coastlines and beyond the routes of European colonization (save for some minor gold and diamond mining), it is a state of relatively recent frontier occupation. This isolation enabled it to retain untouched indigenous territories, savannas and forests up until the mid-twentieth century (Fig. 19.1).

Beginning in the 1970s, through national integration policies promoted by the military regime, the state of Mato Grosso received a substantial flow of financial resources for infrastructure development. Numerous colonists arrived from traditional agricultural lands in southern Brazil, a diversified mass of small, medium and large landowners enticed by a package of fiscal policies and credit that stimulated intensive use of agricultural inputs along Green Revolution lines. The region offered comparative natural resource advantages, since it possesses a tropical climate, smooth terrain and regular rainfall patterns favourable to agriculture in a large-scale production. The government incentive policies and the natural characteristics of the region were responsible for Mato Grosso's receipt of significant public and later private investments that permitted it to become a major pole of agricultural commodity expansion. The agricultural area of Mato Grosso now covers nearly 88,000 km^2, close to 27% of the total deforested area, while ranching occupies about 233,000 km^2, corresponding to 73% of the deforested area.

Mato Grosso's main crop is soybeans, occupying nearly two-thirds of the total cultivated area of the state, followed by maize, cotton, rice and sugar cane. Favourable market conditions, allied with private sector investment and propitious credit policies towards the sector, led to a substantial increase in the soybean area. From 1990 to 2000, it went from 15,000 to 29,000 km^2, an average annual increase of 6.5%, and from 2000 to 2009 this growth was even greater, 10% per year, attaining output of 18 million tons in 2009. These numbers correspond to 31% of Brazil and 8% of total global soybean production (IBGE 2006). At the same time, the cattle herd expanded from nine to nearly 27 million head between 1990 and 2005, maintaining an annual growth rate averaging approximately 7.5% over the entire period, which makes Mato Grosso the largest cattle producer in Brazil. To calculate the area involved in this expansion, we rely on an average stocking rate of 1 head/hectare, which would account for 260,000 km^2 (livestock are recorded in number and not in area occupied by pastures in Brazil). It is projected that growth in livestock will be 100% by 2020 (IBGE 2009).

Agriculture is the main economic activity in the state, responsible for all resources mobilized in the service sector; around 70% of the gross state product is related directly or indirectly to the primary sector. Agribusiness is important not only to the regional economy, but also at the national level, where it is responsible for 8.5% of net trade, generating a surplus in 2007 of US$3.1 billion (SEPLAN 2008). At a national scale, the agribusiness trade balance was positive at $63 billion in 2010. This amount is $8.1 billion higher than it was in 2009 and three times higher than the $20 billion seen in global trade surplus in Brazil.

19.2.1 Agricultural Expansion and Deforestation in Mato Grosso

Deforestation in the Amazon is the principal problem that assails the world's largest tropical forest remnant. Of the nine states in Brazil's Amazon basin, only three (Mato Grosso, Pará and Rondônia) are responsible for nearly all of the deforestation of the remaining 18% of original forest area. These three states show similar patterns of occupation as the agricultural frontier advances from south to north. Should this pattern continue, it is anticipated that more protected areas will begin to exhibit similar rates of deforestation. Over the decade, beginning in the year 2000, the state of Mato Grosso has been responsible for around 40% of all deforestation in the Brazilian Amazon through expansion of agricultural activity. The occupation of forest is initially based on extraction of timber species of commercial value, accompanied by burning of species that are of lesser economic interest. Fire is the quickest and cheapest way to open up new agricultural areas. This logic of occupation is rooted in the cultural and economic logic of farmers who seek to extract maximum profit from the land as quickly as possible.

The fluctuating rates of deforestation in the Amazon can be explained by macroeconomic factors such as international commodity prices, the land market, inflation and financial markets (Andersen 1996; Barreto 2007; Cattaneo 2002). The deforestation dynamic is strongly dependent on the potential returns from agricultural land use (Margulis 2003). Annual deforestation rates have fluctuated in strong correlation with prices of principal agricultural and livestock commodities (beef and soybeans) (Barreto 2007). The peak prices of soybeans in the period 2001–2004, for example, were accompanied by an increase in the planted area from 31,000 to 52,000 km^2, a 69% rise in 3 years (IBGE 2006). During the same period, direct conversion of forests into agricultural fields represented 16% of deforestation in forest areas of the state, peaking at 23% in 2003. This figure is based on a consideration of only deforested areas over 25 hectare (ha) in size, which represented 85% of the total during this period (Morton et al. 2006). Besides this direct conversion of forests and *cerrados*, the conversion of pasture areas into agriculture in the north-central part of the state was also accompanied by the dislocation of ranching to new frontiers in the extreme north and northwest, contributing to the expansion of open areas in these regions.

19.2.2 Environmental Services and Payments

Agro-pastoral expansion leads to a loss in environmental services furnished by native vegetative cover; pastures and crops planted after forest clearing respire less water back to the atmosphere and absorb less solar energy, jointly effecting a reduction in rainfall and an increase in temperature in the Amazon region. Fearnside (2008) posits that this logic of land occupation associated with deforestation accelerates the process of conversion of the Amazon rainforest into savannas and, besides altering continental rainforest patterns, results in a perpetuation of natural burning, which continues to suppress forest rejuvenation. Forest fires, besides provoking an increase in greenhouse gas emissions, generate a large volume of particulate matter and export nutrients from agroecosystems. That is, besides the loss of forest-related environmental services, this interconnected series of processes can provoke an increase in forest fires, aggravating the risks of greenhouse warming and resulting in more rapid soil degradation.

The identification of the importance of environmental services and growing recognition of their continual deterioration is recent, having been exemplified through the *Millennium Ecosystem Assessment* (MA 2005). The MA concluded that more than 60% of the global ecosystems have been used in an unsustainable fashion. The MA classifies the services derived from natural ecosystems into four principal groups: provisioning, regulation, support and cultural, which assure well-being of human societies. One reason for maintaining forests and biodiversity lies in the fact that they provide a bountiful array of environmental services. Forests subsidize the functioning of agroecosystems through their provision of environmental services such as climate regulation, supply and regulation of water resources and erosion control that directly benefit humans' quality of life. When these services are lost through biological simplification, economic and environmental costs can be significant (Altieri and Nicholls 2000).

For Pagiola et al. (2002), PES consists of the sale of services provided by forests, be they public or private. PES has as its fundamental principal compensation of the provider of an environmental service for the benefit furnished to a third party or to a collectivity. It is the "provider-receiver" principle; that is, he who offers an environmental service, generating benefits to society, has the right to be compensated for not using the land for a purpose other than for maintaining or restoring the forest. The idea is to motivate the proprietor of land (be it public or private) to include environmental services in their decision-making regarding land use, making conservation a financially more attractive option. The objective of PES is not to substitute for productive activities but to motivate conservationist practices concomitantly with other land uses. It is related to a development plan based on conservation, on income generation and on furnishing environmental services.

Economic instruments based on PES will not substitute command and control instruments since application of PES requires a legal framework to delimit the economic activities involved. On the contrary, complementarity should be sought between the two types of instruments, seeking to reach the objectives of public policy at least cost to society. The operationalization of any PES instrument requires bargaining between public and private institutions to establish a market for environmental service compensation in close articulation with pre-existing command and control instruments.

19.3 Existing Policy Mix to Protect Biodiversity

19.3.1 Protected Areas for Maintenance of Environmental Services

The Brazilian government is a signatory of the UN Convention on Biological Diversity (CBD) which has as its target at least 30% of the Amazon and 10% of other biomes effectively conserved in protected areas within the National System of Conservation Units (SNUC). It also committed itself to guarantee the protection of biodiversity in at least two-thirds of priority areas for diversity through a combination of protected areas within the SNUC, as well as those lying within indigenous lands and territories of former slave communities "Quilombolas" (MMA 2007).

Within the Amazon region, the state of Mato Grosso has the smallest proportion of its total area protected for conservation. While Pará, Rondônia and Acre hold from 26 to 33% of their territory in protected areas, Mato Grosso counts only 4%. Adding up the indigenous lands that allow sustainable use, the protected area cover a total of 170,000 km^2 in Mato Grosso, corresponding to 19% of the total area of Mato Grosso. Besides their relevance of ecosystem services (provision, regulation and cultural), these areas have effectively contained the advance of deforestation in the state. Conservation units (5%) and Indian lands (4%) were deforested at a much lower rate than were private properties (44%).

Because of its recognized importance as a barrier to deforestation, new areas have been identified for protected status. In all, 15 such areas covering 63,700 km^2 were proposed for biodiversity protection, an additional 7% of the surface area of the state, of which 34,000 km^2 lies in the Amazon biome and 29,000 km^2 in the *cerrado*. They were included in the proposal for ecological-economic zoning (ZSEE-MT),[1] prepared by the state's executive branch, passed through the public consultation phase, approved by the state legislature and is currently under revision by the federal government.

These areas indicated by the ZSEE are included in the Probio 2005 listing, showing that their importance for biodiversity conservation is recognized nationally and that their conservation would be part of a Brazilian strategy for compliance with its commitment to reduce additional biodiversity loss, as expressed in the Millennium Development Goals. Each of these proposed protected areas has specific importance, since they protect ecosystems threatened by human pressure, areas of important aquifers, endemic species of fauna and flora threatened with extinction and physiognomic patterns exclusive to these environments.

But the creation of these areas is not easy; the only criterion is not based on scientific knowledge; there are political and economic interests involved. These protected areas proposed by the ZSEE-MT have generated heated and polarized discussions among those with links to rural landowners and socio-environmental entities throughout the entire process, principally in the public hearings. On a number of occasions, landowners with ties to the agricultural sector have suggested the reduction or even elimination of the protected areas proposed by the zoning bill.

Moreover, often the government, as asset manager, has other priorities when it comes to spending. One of the arguments of the state for avoiding the creation of new protected areas is the high cost due to limitations of the public budget. These costs are high primarily when one considers the perennially scarce public financing available in environmental budgets at whatever scale: municipal, state or federal (Young and Roncisvalle 2002).

[1] Ecological-economic zoning has been required since 1990 by the federal government in the nine states that compose the Legal Amazon. State zoning is an instrument of territorial planning with the objective of influencing decisions of public and private actors regarding the use of natural resources and balancing maintenance of natural capital and ecosystem services with economic activities. The spatial distribution of economic activities under ZSEE takes into account the limitations and fragilities of ecosystems, establishing restrictions and alternatives to territorial expansion of their exploitation and social benefits.

19.3.2 Reducing of Emissions from Deforestation and Forest Degradation (REDD)

Payment for storage of carbon in tropical forests, denominated "avoided deforestation", has come to be discussed as a means to make possible a rapid reduction in deforestation-related emissions (Chomitz 2006; Santilli et al. 2005). This proposed mechanism to assure financial compensation for reducing deforestation in developing countries has become known by the acronym REDD (Reducing Emissions from Deforestation and Forest Degradation). At a global level, deforestation is considered to represent as much as 20% of greenhouse gas emissions. Deforestation and land use change-related emissions in Brazil have been estimated most recently as 54% of total greenhouse gas emissions in CO_2 equivalent measures. The greater relative importance of such emissions compared to most other nations implies that for Brazil to respond to its role as a signatory of the Climate Change Convention, it must find some way to reduce these emissions. Although the federal government had not previously articulated a deforestation target, under the National Climate Change Plan promulgated in 2009, it resolved to reduce its emissions associated with deforestation in the Amazon by 80% by 2020.

There is already a market for carbon as an "environmental commodity," as an offshoot of the so-called flexibility mechanisms of the Climate Change Convention. The market value of carbon arising from these mechanisms has fluctuated and varies between that negotiated among actors associated with the European Emissions Trading Scheme and informal markets that have emerged to capture a range of different values associated with emissions reduction, including avoidance of deforestation. For a number of reasons, maintenance of forest carbon stocks was not afforded formal status in the Kyoto Protocol mechanisms. Only forest restoration or afforestation (conversion of bare or cultivated land into forest) is eligible for crediting via the Clean Development Mechanism (CDM). Following debates at the Conference of the Parties to the Climate Convention that resulted in definition of global policies for combating greenhouse warming post-Kyoto, the perspective that parties will receive compensation for their good faith efforts to reduce deforestation became more tangible.

Growing concern with the effects of carbon emissions on global warming has necessitated the creation of instruments that can revert deforestation and offer economic opportunities for those who maintain forests intact. Effective systems of property registry, tenure regularization and implementation of land use monitoring as well as the restoration of environmental liabilities (areas cleared beyond legal limits) are therefore all prerequisites to enabling REDD projects.

Discussion has already begun regarding policies and mechanisms in existence in Mato Grosso that could have REDD as an important complementary mechanism for greater control over illegal deforestation in the Amazon to protect forest cover and biodiversity. In 2010, the State Climate Change Forum, composed of various representatives of society, after 12 months of meetings and public consultations, finalized a draft state plan for climate change. Since then, the proposal has awaited presentation by the executive to the legislature. This legal framework will allow regulation of the emissions of greenhouse gases and will support REDD.

There is a strong interface between the climate change policy and the appropriate use of land to promote territorial development. Implementation of REDD demands a new set of instruments and coordinated measures that necessarily involve society and government, with due consideration for the ecological and economic specificity of each region of the state as described in the ZSEE. Spatial differentiation is of fundamental importance for the REDD instrument under discussion here, as it will greatly augment the efficiency of payment mechanisms to areas where the most critical ecosystem functions are under greatest threat. In many cases, good legislation has been enacted but with a lack of complementary resources, be they technological, financial or human, is not yet fully effective. The state of Mato Grosso was the first in the Amazon region to take the first steps towards environmental decentralization.

In response to the federal requirement that rural properties be environmentally licensed, Mato Grosso instituted a combined environmental licence (*Licenciamento Ambiental Único-LAU*) in 2000. This mechanism was linked to a technological package for monitoring based on satellite imagery that was instituted simultaneously as a means to resolve the illegal deforestation problem in the region as a whole. The Environmental Licensing System for Rural Properties (SLAPR, implemented on the basis of the LAU), entails integrated monitoring of deforestation using images provided by landowners at the time of licensing showing their properties and their protected areas. Such information was then used for forest control and environmental licensing as a requisite to obtaining authorization for additional deforestation.

If deforestation has exceeded authorized limits or made incursions into protected areas on the property, penalties require a corrective action. All of this imagery is available on-line for public scrutiny. Properties adhere voluntarily to the system, but it is mandatory for anyone seeking authorization to deforest or to conduct sustainable management of forests. Other incentives to adherence include the possibility to compensate the legal reserve in another property or in a protected area, facilities on rural credit access and the potential for market incentives and payment for environmental services (Azevedo 2009). However, besides the great importance of the instrument, adherence to SLAPR is still lower than expected. Some reasons for this are the repudiation of monitoring, licensing costs, asymmetric information and the long delay in issuance of the licence, high opportunity cost to maintain the legal reserve and the possibility of discontinuing the policy instrument (Azevedo 2009).

A programme for environmental and agrarian regularization was created targeting local governments of Mato Grosso, entitled MT Legal. This programme, implemented in the second half of 2009, seeks to motivate private property owners to register in the SLAPR, creating a market based on forest assets and liabilities and offering the potential for establishment of a trading scheme for environmental services. Despite innovations that make the state be at the vanguard, the annual rates of deforestation continue to fluctuate significantly. We will demonstrate in the case study below that the programmes and public policies of reduced deforestation in the state, if coordinated with the ZSEE proposal, would open up the possibility to undertake a series of effective initiatives for reduction and control of deforestation and greenhouse gas emissions. The proposed areas for conservation will simultaneously augment the representativeness of protected biodiversity, permitting resources to be attracted through REDD and increasing the number of properties registered in the SLAPR.

19.4 New Protected Areas and Effective Protection of Existing Areas in Mato Grosso

The 35,000 km^2 of protected areas already in existence in MT do not fully represent the diversity of fauna and flora present within the state, nor do they hold sufficient potential for reducing deforestation-related carbon emissions. The justification for creation of new protected areas arises therefore from strong arguments regarding the need to protect additional areas rich in biodiversity and to reduce carbon emissions in the state. The accumulated deforestation in these new protected areas represents 24% of their original total surface area. All this biodiversity is at risk, subject to human pressures, due to its location in private areas that have registered deforestations from the moment they were proposed for protection. In 6 of the 15 areas, this proportion was between 15 and 25%, while in four areas, it exceeded 25% and in a final five areas represented less than 15%. Of the total area, 24,000 km^2 (38%) is found within properties registered in the SLAPR. This relatively high rate of registered properties relative to the rest of the state clearly reflects the interest of landowners in assuring their property rights in the face of fear of expropriation.

19.4.1 Estimate of Private Areas That Can Be Regularized by Compensation in Protected Areas

As described previously, many rural properties in Mato Grosso have been deforested beyond limits permitted by the environmental legislation. As a result, there are a large proportion of agricultural properties with irregular status and legal reserve liabilities. Working within the context of the new institutional and regulatory framework, we assert that the creation of new protected areas would create a stock of lands fundamental to making possible the environmental regularization of already deforested areas in the state.

Based on an estimate of the total surface area cleared for production, and from available data on deforestation and property maps, it is possible to estimate the amount of deforestation beyond permitted limits for closed forest and *cerrado* in Mato Grosso. The original extent of forest cover in Mato Grosso was 525,000 km^2. Of this total, the area cleared up to 2007 was 163,000 km^2 (43%). We calculate that this area includes about 61,000 km^2 of potentially regular areas, and 102,000 km^2 of areas cleared beyond the 20% allowed on each property, deemed irregular. In relation to the *cerrado* areas of the state, their original extent was 377,000 km^2. Of this total, the area cleared up to 2007 was 136,000 km^2 (49%). We calculate that this area includes about 118,000 km^2 of potentially regular cleared areas and 18,000 km^2 of irregular areas, cleared beyond the 65% permitted on each property.

19.4.2 Options for Regularization of Legal Reserve Liabilities

State environmental legislation, consistent with the national Forest Code,[2] offers three alternatives for regularizing legal reserve liabilities: restoration on the property, compensation in another private area that holds a surplus of legal reserve or compensation in a protected area. The option for restoration of legal reserves on the property could be appropriate in small and/or degraded areas, although generally implies a high cost. Considering existing planting techniques in degraded sites practised in Mato Grosso, the cost of recuperation varies between US$1,390 and $2,220 per hectare (Hercowitz 2009). Besides this, there is the opportunity cost for the landowner in desisting from use of productive areas so as to restore his reserve, which – when added to the cost of recuperation – makes this option even more onerous, especially in areas with high productive potential.

The option for compensation in another private area, through easement or outright purchase of surplus legal reserve area, is of great interest but also has significant limitations. We calculate that the surplus legal reserve area in private properties in Mato Grosso adds up to about 24,000 km^2 in forested areas and 19,000 km^2 in *cerrado* areas. The first limitation of this option is that the surplus legal reserve area in forests is far from being sufficient to that necessary to regularize the liabilities. Besides this, this option is available only for deforestation that occurred prior to 1998 and is therefore inapplicable to the majority of liabilities, whether in the forest or in the cerrado. Besides this, such an option implies an elevated transaction cost, from searching for an area with surplus reserve area available for compensation, through negotiation and effective acquisition of an area.

Compensation in existing protected areas is an option that may appeal more to landowners, as they would not have to face the opportunity cost of reducing productive areas nor the cost of maintaining or restoring the legal reserve. It is also an option of interest to the state, as it would provide opportunities for the regularizing of tenure of already existing protected areas. According to SEMA-MT, the total area in state protected areas requiring tenure regularization represents nearly 8,000 km^2 in forest areas and 5,000 km^2 in the *cerrado*. That is, the potential for compensation for irregular land use in existing protected areas is relevant but insufficient when compared to the scale of existing liabilities. Even when all the areas offering the possibility of regularization among compensation options above are added up, there remains a deficit of 29,000 km^2. The potential for regularization considering all natural remnants in proposed protected areas in the bill to establish the ZSEE would be 26,000 km^2. Therefore, the creation of new protected areas is fundamental to enabling the regularization of already deforested areas and to implementing the MT LEGAL compensation programme (Table 19.1).

The compensation of legal reserves with state protected areas would make it possible for rural landowners to obtain the environmental licence (LAU) and for the

[2] The Forest Code is undergoing revision in the Congress, but these flexibility provisions are expected to be maintained in the revised code.

Table 19.1 Total deforestation and natural remnants by tenure type in MT, 2007

	Deforested area		Remnant area		Total area	
Tenure type	(km^2)	(%)	(km^2)	(%)	(km^2)	(%)
Indigenous territories	5,193	4	129,852	96	**135,045**	**15**
Conservation units (not including APAs)	1,869	5	33,861	95	**35,730**	**4**
Other areas (settlements, properties and squatters)	322,014	44	410,795	56	**732,809**	**81**
Total	**329,076**	**36**	**574,508**	**64**	**905,584**	**100**

Sources: SEMA-MT (UCs, TIs, SISLAM), 2007; analysis by ICV

Table 19.2 Costs of environmental regularization by property transaction option, Mato Grosso, Brazil

Item	Hectares with deficit (million)	Cost per hectare	Total cost (million)
Deficit in legal reserve area restored	6.1	R$2,500–4,000[a]	R$1,525–24,400
Liability compensated in other private area	2.4	R$5,100[b]	R$12,235
Liability compensated in protected area	3.4	R$800[c]	R$2,720

[a] Hercowitz (2009)
[b] Micol et al. (2008)
[c] (SEMA-MT, personal communication)

state to integrate private lands within protected areas that have not been indemnified, at a lower cost. To have an idea of the order of magnitude of this cost, we consider the market value of the land with native vegetation. For all of the proposed areas, the total potential cost of acquisition would be in the order of US$1.7 billion. If we divide this value by the total area of 64.725 million ha, this value comes to just under $2,830 per hectare. This is a significant amount of resources that SEMA would have to surrender from its budget if there were no lower cost alternative.

Thus, of the 61,000 km^2 that require recuperation in potentially regularized properties located in the Amazon biome, 48,000 km^2 could be compensated or exonerated within existing and future proposed protected areas, as well as in private properties that have not been deforested and that are expected to remain in that state to the extent that the agricultural frontier is consolidated in areas already defined as such by the zoning plan. The cost of these two actions in combination would require a level of resources considerably lower than the high-range restoration costs of over US$13 billion estimated above, as can be seen in the details provided in Table 19.2.

Therefore, the option of compensation of private liabilities in protected areas would imply an effective savings both for the proprietor and the state in attaining the ZSEE goals, creating a context for net gain from negotiation. The transaction costs of these exchanges would be assumed by proprietors whose lands would require regularization under the rural licensing law. Until today, few cases of legal reserve compensation within protected areas have actually taken place. In Mato Grosso, the number of cases is no more than a dozen, while an additional number have been stalled from going ahead since 2005. This possibility should be better studied to identify the principal limitations, as it opens up an important opportunity for rural landowners to resolve their environmental responsibilities through compensation at a relatively lower cost than other options presented. Besides this, it would allow for new protected areas to be created, guarding these areas against further deforestation. This form of environmental compensation is an instrument that can overcome the high costs associated with restoration and bring properties into line with environmental licensing requirements.

19.4.3 Avoided Deforestation in Areas Indicated for Creation of Protected Areas

Over the past decade, the state of Mato Grosso emitted through clearing and burning nearly one billion tons of carbon stored in biomass, or an average of 366 million t CO_2/year. This volume may account for as much as 10% of global deforestation-related greenhouse gas emissions. In this chapter, we assume that portion of the carbon emitted during the conversion of land use is totally dedicated to agriculture. According to Houghton (2005), deforestation and conversion to agricultural land use, whether agricultural crops or pasture, causes the emission of 90–100% of the carbon stored in aboveground biomass. We are not accounting for the emission of the carbon stored in soil (25% for crops and 12% for pastures).

Due to the lack of consolidated spatial data on soil carbon and land use data at an adequate scale, we propose to initially consider an emission of 100% of the carbon stored in aboveground biomass, regardless of the subsequent land use. For the amount of carbon stored in aboveground biomass, we estimate a conservative average by zone based on existing studies and maps (Saatchi et al. 2007). We produced a map (Fig. 19.2) representing the quantity of carbon stored in forest formations found in the new protected areas proposed by the ZSEE. The areas demarcated on the map contain carbon ranging from 40 tC/ha in more open *cerrado* formations up to 130 tC/ha in forest areas, considering only the carbon stored above the soil surface (not including forest litter or root biomass). Field studies carried out in the northwest region of the state show that this value can attain as much as 195.6 tC/ha when other stocks of carbon besides living aboveground biomass are considered (Scaranello 2011).

Following this, based on deforestation rates over the past decade, we projected (Fig. 19.3) an average deforestation of 1,000 km^2 per year, in all new areas proposed for creation of protected areas. Considering the deforestation rates of the past 10 years and the per hectare carbon stock in each proposed protected area, we then estimated the historical emissions associated with deforestation in these areas. The resulting calculation suggests that emissions could have reached nearly 72 million tons of carbon (265 million tons of CO_2) between 1997 and 2007, an average of 7.2 million tons of carbon per year (26 million tons of CO_2). With the conservative

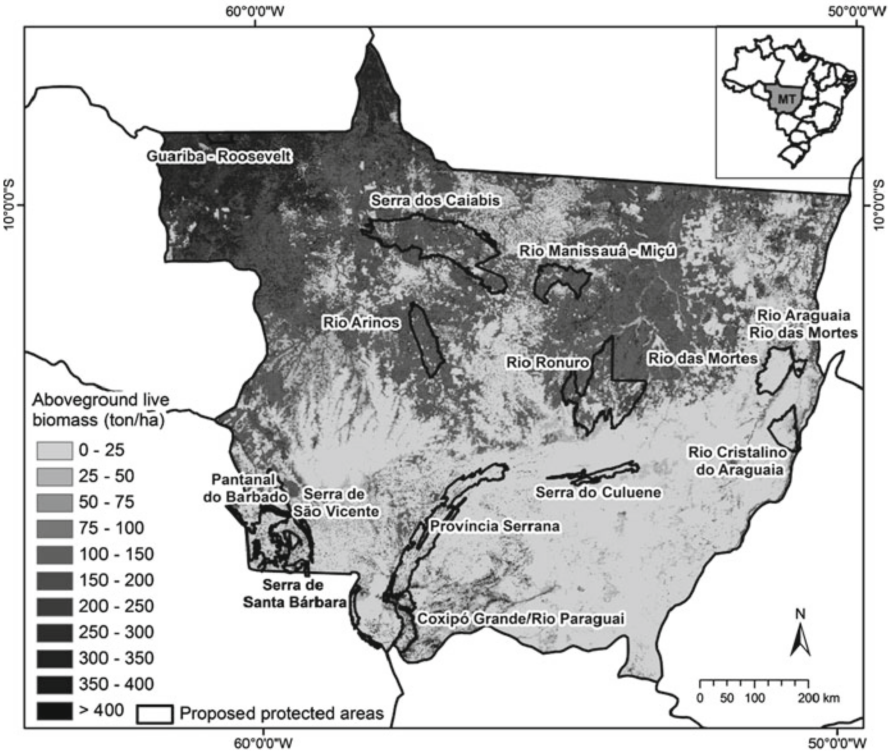

Fig. 19.2 Estimate of carbon stock in protected areas (Based on Saatchi et al. 2007)

hypothesis of an average value of US$5.00 per ton CO_2, the reduced deforestation in these areas could imply financial compensation on the order of US$130 million per year. Although the voluntary carbon market does not currently pay this amount, it can be considered conservative due to the necessity of countries with greater emissions reduction requirements finding other means to reduce their emissions.

19.5 Final Considerations and Conclusions

The creation of protected areas requires specific in-depth studies to determine their permitted use (integral protection or sustainable use), management category (park, biological reserve, forest, extractive reserve, etc.) and their demarcation. These studies would locally analyse and map the areas of greater importance for conservation, the eventual existence of natural limits, as well as the types of potential uses of areas to be created and the possible socio-economic impacts of their creation.

The process of creating protected areas must also involve local society through public consultations where studies are presented and proposals are discussed in

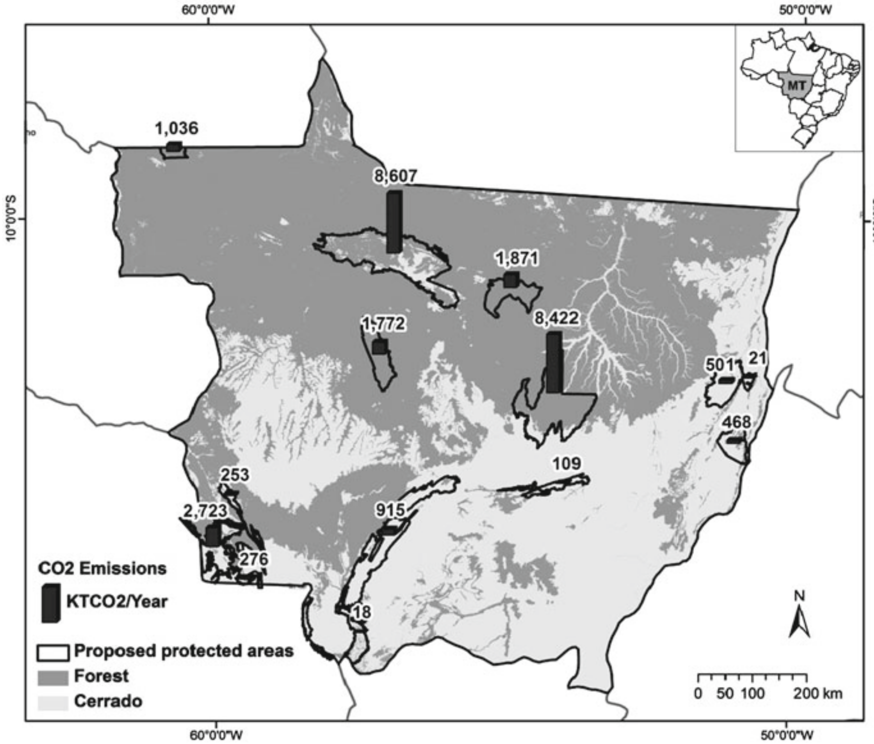

Fig. 19.3 Annual CO_2 emissions from areas proposed for creation of protected areas

order to make them appropriate to local realities. Based on the experience of the ZSEE consultations, it is in the general societal interest of all regions of the state to find adequate pathways towards socio-environmental conciliation. However, there are clearly oppositional views on the assumption of costs on the part of economic agents, some of whom would prefer to eliminate the creation of new protected areas as an option. However, a negotiation of solutions of lesser cost would, at least conceptually, be of interest to all actors. During the entire year of 2011, a legal battle was waged so that the ZSEE takes into account the proposed new protected areas. There is therefore still some hope that these areas will be defined in law, but this is still an open question awaiting judicial decision.

The protected areas proposals outlined in the law for the establishment of the ZSEE-MT are fundamental components in the strategy of environmental and territorial management for Mato Grosso. These are necessary to enable the state to effectively execute its commitments to national roles in the Convention on Biological Diversity. On the other hand, they will also be necessary to ensure the environmental regularization of rural properties in the realm of MT Legal. Therefore, the elimination of protected areas proposals from the ZSEE should be discarded as an option.

Besides the richness of biodiversity existent in these areas, they also offer the potential to generate financial resources for the state within the global carbon market.

The new protected areas would represent a direct and concrete basis for implementation of REDD mechanisms. It is very much the case, however, that the carbon market is still in process of definition, as well as is the role for forests in carbon trading. A series of uncertainties exist that affect the development of solutions in an unregulated environment. Voluntary funds, programmes of "REDD-readiness," etc., indicate this proposal can be part of the equation of emission reductions associated with deforestation. There is recognizably a long road forward towards this definition, but the analysis and mobilization of society around initiatives of this kind is the first step in this direction.

In 2011, a working group composed of diverse civil society organizations delegated formally by the State Forum on Climate Change worked to formulate a REDD law in the state of Mato Grosso. The group carried out a series of public hearings to present the proposal, and to receive suggestions that reflect social diversity, including indigenous groups, family farmers and timber companies. At the outset of 2012, the proposed law will be submitted to the state assembly for a vote, resulting in the creation of a regulatory framework for REDD as a basic instrument for functioning of a PES scheme based on avoided CO_2 emissions. However, despite state government support, some segments resist the creation of such a mechanism. In this sense, there still remains a great deal of work to be done to clarify and convince such groups of the benefits that such a programme could bring.

This chapter concludes that PES systems may be defined within the realm of a sustainable scale of economic activity within ecological constraints, defined and regulated by law, such as through ZEE. At the same time, it is clear that political will and articulation is a fundamental and inseparable principle for the implementation of such a law. This suggests that a good idea cannot prosper while powerful stakeholders remain unconvinced of its benefits. We believe that this represents one of the greatest challenges to implementation of policies for preservation of the Amazon forest.

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Chapter 20
Forest Carbon Credits Generation in Brazil: The Case of Small Farmers

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20.1 Introduction

Forests, besides playing an important role to the global environmental process, represent a valuable source of products and services to humankind (Baskent and Keleş 2009). Despite their importance, forests have been systematically deforested due to economical and cultural issues (Azuela 2006).

In this way, the deforestation in tropical areas has presented high rates (Myers et al. 2000). In Brazil, the predatory model of forest resource exploitation by clear cutting and subsequent conversion to other land uses, besides the biodiversity loss, leads to an increase of carbon dioxide (CO_2) in the atmosphere, achieving values of 0.28 (0.17–0.49) Gt C per year (DeFries et al. 2002).

Generation of carbon credits, through forestry projects, is a way to reduce pressure for logging in new forestry areas and add economic value to forest. These credits may be obtained under the Kyoto Protocol by three market-based mechanisms: Joint

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Implementation, Emissions Trading and Clean Development Mechanism – CDM (UNFCCC 1998).

CDM has a particular importance in Brazil because it is the only mechanism in which developing countries (Non-Annex I) can participate. Its purpose is to assist Non-Annex I countries to achieve sustainable development and to reduce their greenhouse gas (GHG) emissions.

On the other hand, this flexibility mechanism also intends to assist Annex I countries to comply with their reduction target emission (UNFCCC 1998). Robledo and Pfund (2004) highlight CDM as the most relevant instrument in the context of climate change, forestry and sustainable development.

Currently, 39 forestry projects are registered under the United Nations Framework Convention on Climate Change – UNFCCC (UNFCCC 2012). Small farmers are usually included (directly or indirectly) in these projects. For Brazil, it will be important to stimulate forestry projects with the participation of small farmers due to the large representation of this sector in the country.

The majority of rural settings in Brazil are occupied by small farms (IBGE 2006) with fragile livelihood conditions. They usually do not manage natural resources efficiently, often because they do not have access and knowledge regarding sustainable practices. These inadequate procedures result in several negative environmental impacts to the ecosystems surrounding their farms (Gomes 2004; Lira et al. 2006).

Besides this, in Brazil, most of the farmers aim to obtain a high income from the land in the short term. Forests are usually seen as areas that cannot provide revenues. Therefore, it is common practice in the country to convert natural areas into agricultural crops. This practice is also followed by small farmers.

Current rates of deforestation in the Amazon and other Brazilian biomes attest to the continuing erosion of local natural resources and the reduction of their ability to provide environmental services. Under the military regime (from 1964 to 1985), an aggressive strategy of regional development was implemented, favouring large landowners and commercial enterprises, but also encouraging small farmers' settlement (Hall 1989).

Overall, medium- and large-scale cattle ranching have been responsible for around 70% of forest loss in the Amazon region, small-scale farming near 20% and commercial logging and mining for the remainder (Fearnside 2005). Subsequent civilian governments have been equally supportive of settlement and commercial development policies that have led to consistently high rates of forest loss. Expansion of the highway network and soya bean cultivation in the Amazon has recently added to such pressures (Amaral and Smeraldi 2005; Fearnside 2005; Greenpeace 2006). A more detailed description of the Amazon forest cover change can be found in May and Millikan (2010).

The Amazonian deforestation is a reflection of the global economy. The current pressure for expansion of new cultivation areas in the world and growing demand for Amazonian products such as beef (free from diseases) and soya beans, produced with cutting-edge technology, may generate increased deforestation in the near future (Soares-Filho et al. 2006). In this context, the exchange rate plays an important role in the generation of deforestation. Although it is not currently the case, the devaluation of the real (Brazilian currency) against the dollar, that occurred between

2001 and 2004, indirectly encouraged deforestation (the last year with a peak of 27,400 km^2 of deforestation). With the real valued higher, world prices for beef and soya beans may be relatively lower which in part explains the reduction in deforestation rates observed in recent years (2006 onwards). On the banks of this process are small farmers who rely on their own labour and produce to meet their own consumption. Therefore, they contribute to the basic rate of deforestation in a much lesser degree (Moutinho 2009).

Forest carbon credits generation focused on small farmers is an opportunity to harness the forestry potential of the country, alleviate poverty and improve ecological conditions of degraded areas. However, there are challenges to be overcome, such as bureaucracy and high transaction costs of CDM projects.

In this chapter, we sought to identify and discuss some obstacles that Brazilian small farmers face to develop a CDM project. For this, we made a literature review focusing on national data, documents of COP, data from UNFCCC and carbon market reports. Initially, we briefly characterise Brazil under the CDM and the situation of small producers in the country. Suggestions to improve their integration in the forestry carbon market are also presented. Finally, some necessary conditions for small farmers meet the needs of the forestry carbon market were proposed.

20.1.1 Brazil and the Forestry CDM

In the UNFCCC, Brazil is the third country in the ranking of registered projects under the CDM (UNFCCC 2012). However, most of the projects are related with biomass energy, methane avoidance and hydroelectricity.

Brazil has 61.0% of its territory covered by forests (FAO 2010); from this, 1.3% (6.5 million ha) is represented by forest plantations with *Eucalyptus* and *Pinus* (ABRAF 2011). The degraded areas in the country account for near 30 million ha. Although Brazil has an extensive forest cover, the country only has two reforestation projects registered in the CDM. These two reforestation projects in Brazil are associated with private companies: Plantar S/A and AES Tietê.

Plantar S/A operates in pig iron production and eucalyptus plantation. The Plantar CDM project is titled “Reforestation as renewable source of wood supplies for industrial use in Brazil” and used the approved methodology “Afforestation and reforestation project activities implemented for industrial and/or commercial uses” (AR-AM0005). It aims to establish eucalyptus plantation as a renewable source of wood supplies to meet the pig iron industry needs. The project will have 30 years crediting period and will result in a reduction of 2,273,493 tonnes of CO_2e.

The AES Tietê-CDM project is titled “AES Tietê afforestation/reforestation project in the state of São Paulo, Brazil.” This project used the approved methodology “Afforestation and reforestation project activities implemented on unmanaged grassland in reserve/protected areas” (AR-AM0010 ver. 4). Its proponent is a large Brazilian electrical energy generator that owns and operates ten hydropower plants, with an installed capacity of 2,651 MW within the state of São Paulo, Brazil. The aim of this project is the reforestation of riparian vegetation with native tree species along banks of hydropower reservoirs that are currently covered by unmanaged grassland. The project will reduce 4,729,074 tonnes of CO_2e during 30 years of crediting period.

These two projects, despite having good proposals and aims, are essentially business projects. In reality, the major aim of both projects is to raise financial funds and improve the image of both companies in the society. They are more focused on the carbon credits generation than in social issues (the development and support of local communities).

20.2 Brazilian Small Farmers and Forestry CDM

Small farmers are considered as producers for whom agricultural activities are the main source of income. Further, this definition also assumes that the farmer family is the primary source of labour in their farm. Thus, in this chapter, the term "small farmers" was used as a synonym of familiar agriculture.

In Brazil, familiar agriculture (small farmers) is defined by the following traits:

- The area of the property is not bigger than four fiscal modules (limit of area regionally established[1]).
- The use of labour from the family is bigger than from hired labours.
- The family income is mainly derived from their farm.
- The production is under direction of the farmer and his family (Brasil 2006).

The characterisation of small farmers in Brazil was based on the Brazilian agricultural census 2006 (IBGE 2006). Small farmers represent 84.4% of the rural environment in Brazil. Moreover, they occupy 24.3% of the total agricultural area and contribute around 38.0% to the total agriculture production.

Overall, 45.3% of the small farmers' area is used as pastures, followed by forests/agroforestry systems (27.7%) and agriculture (22.0%). In the remaining area (5.0%) are betterments and inappropriate areas for cultivation and cattle raising.

Most of the small farmers (80.9%) reside in their own farm, while 19.1% probably live in nearby villages or urban centres. Regarding education, 63.3% of small farmers stated they could read; however, only 1.54% declared having some kind of professional qualification. However, illiteracy is still a challenge in rural areas, affecting more than one-third (36.7%) of small farmers.

The income of small farmers is around R$13,630 per year (~USD 7,972[2]). This income is supported mainly by the sales of vegetables (67.5%), animals and their

[1] The fiscal module consider: (1) type of exploitation prevalent in the municipality; (2) other type of exploitation, although not predominant, which is significant as an income generation; (3) concept of family property. The fiscal module serves as a parameter for the classification of rural property in size, according to Law nº. 8629/1993. Small farm – farm area is between 1 (one) and 4 (four) fiscal modules; average farm – farm area is more than 4 (four) and less than 15 (fifteen) fiscal modules (Brasil 1993).

[2] USD 1.00 = R$1.7098 (September, 2011).

products (21.0%). Moreover, an additional income is sometimes obtained with retirement/pensions (65.2%) and non-farm activities (24.2%).

Recently, the Brazilian media has given great attention to small farmers due to a draft bill that will ease restrictions in the current forest code (law nº 4.771/65; Brasil 1965). In the revised forest code (Brasil 2011a), which is underway in the Chamber of Deputies, properties up to four "fiscal modules" (i.e. small farmers properties) that illegally deforested their own legal reserve until July 22, 2008 will not have the obligation to recover these areas. Besides this, fines already issued will be permanently suspended (other polemical provisions are proposed but are out of the scope of this study).

These measures aim to stimulate agricultural expansion, instead of dealing with the core of problem that is the lack of investment in crop improvement and productivity growth (Martinelli et al. 2010). Some Brazilian agricultural sectors, as is the case of small farmers, do not use efficient production techniques, due to low education and/or technical instruction level and limited financial income. The Brazilian government is aware of this situation, but it still provides a relatively low level of support in the agricultural sector as a whole. Most of the financial support is used for subsidies to producers, while other areas such as research, extension, training, technical development and rural infrastructure receive less attention and financial funds (OECD 2005).

The consequences of this lack of investment in crucial points reflect not only in the development of agriculture in Brazil but also in the participation of farmers, especially the small ones, in the forest carbon market under the Kyoto Protocol. Thus, Brazilian small farmers still remain outside the market of CDM forest carbon credits generation. This situation is explained by technical and economic issues, besides the high level of bureaucracy and transaction costs of CDM projects.

CDM has sustainable development as one of its basic premises. Despite this, not all projects under the CDM fully satisfy this criterion, especially regarding the social aspect. This is particularly visible among forestry projects, where equity and local development objectives are not always achieved (Boyd et al. 2007; May et al. 2005).

Several factors contributed for this condition. Small farmers in Brazil, as already mentioned, usually have low education and often limited access to information. Issues such as climate change, carbon sequestration and carbon credits are still incomprehensible for most of them.

Moreover, the language used in most papers and documents regarding CDM projects is a critical problem. Most farmers do not understand English, which complicates their understanding of this subject. Thus, it results in a lack of information which makes comprehension of the physical and technical processes involved in the forest carbon credits generation difficult.

Furthermore, the vast bureaucracy and high transaction costs associated with the development and implementation of a CDM forestry project inhibit their broader participation in the carbon market. The bureaucracy is associated with the sequence of tasks performed by project developers, executive board and other

CDM institutions during the development and implementation of a CDM project. Briefly, the main steps of a CDM project cycle are (for more details see Krey 2004):

- Project activity design
- Determination of baseline and/or monitoring methodology to be used (proposal of a new/use of an approved one)
- Validation
- Registration
- Certification/verification

For most small farmers, it would be almost impossible to follow all of these stages independently. Beside this barrier are the high transactions costs. De Gouvello and Coto (2003) divided the CDM transaction costs in monetary and nonmonetary costs.

The monetary costs include the additional services necessary to meet the requirements of the CDM process. The nonmonetary costs are represented by the share of CERs corresponding to the adaptation levy, the one destined to cover the administrative costs of the executive board and, if it is the case, the share retained by the host country.

In the literature, it is possible to find few estimates of the overall costs of a CDM project. Pereira and Gutierrez (2009), based on estimates of World Bank, pointed to an average value of USD 270,000 just to attend the technical bureaucratic requirements of CDM in the case of large-scale projects. For small-scale projects, this cost would be USD 110,000.

De Gouvello and Coto (2003) suggested values for small-scale project ranging from USD 8,000 to 80,000 and for large-scale projects varying between USD 100,000 and 1,100,000. However, it is important to notice that these data are only estimates. These values may change according to several factors, such as the type of the project (size and complexity), experience of the project developer and external consultants, rates charged by the latter and participant country policy and administrative capacity (Ahonen and Hämekoski 2005).

Another important point to mention is the lack of government incentives for the participation of small farmers in national forest carbon projects. This absence of support just reflects the situation of the Brazilian forestry sector under the CDM. Brazil, despite having a legal framework covering several environmental issues, still does not have a specific legislation that stimulates the participation of the forestry sector in CDM projects. However, some initiatives are underway.

20.2.1 Brazilian National Policy on Climate Change

An initiative of the Brazilian government to deal with climate issues is the Brazilian national policy on climate change (law nº 12.187/09; Brasil 2009) in which are established the national strategies for GHG mitigation. Sectorial plans support actions for the development of a low-carbon economy. Incentives for a broader participation of the forestry sector in the CDM are mentioned, though not explicitly.

One of the primary tools for this policy implementation is the National Plan on Climate Change (NPCC). The NPCC includes actions related to the reduction of deforestation rates and enhancement of forest cover. The CDM in this context is seen as an economic instrument that can assist in reducing the net loss of forest cover (Brasil 2008). The actions proposed to stimulate the participation of the forestry sector in the CDM are still unambitious, given the potential of the Brazilian forestry segment.

Other important tools of the Brazilian national policy on climate change are:

1. National fund on climate change
2. Action plans to prevent and control the deforestation in Brazilian biomes
3. Brazilian National Communication to the UNFCCC
4. Resolutions of the Interministerial Commission on Global Climate Change
5. Lines of credit and financing of specific public and private financial agents
6. Research lines from fomentation agencies
7. Financial and economic mechanisms, at the national level, relating to mitigation and adaptation to climate change
8. Records, inventories, estimates, evaluations and other studies of GHGs and its sources, prepared based on information and data provided by public and private entities
9. National climate monitoring
10. Sustainability indicators
11. Establishment of environmental standards and targets, measurable and verifiable, to reduce anthropogenic emissions by sources and removals by sinks of GHGs
12. Evaluation of environmental impacts on the microclimate and macroclimate

20.2.2 Payment for Ecosystem Services

There are still many technical and financial challenges as well as institutional and legal framework for payment for ecosystem services (PES) gain scale in Brazil. PES emerged as an economic tool to address the market failure on the tendency to under supply environmental services due to the lack of interest by economic agents in activities of protection and sustainable use of natural resources. PES is an economic instrument discussed with great emphasis today to promote the protection, management and sustainable use of tropical forests, especially in developing countries like Brazil.

The idea behind the instrument of PES is to reward those who maintain environmental services or encourage others to ensure the provision of environmental services that would not do it without the incentive. In Brazil, there are PES initiatives related to carbon, water and biodiversity. All of them are voluntary and include participation of public, private and non-governmental organisations (Guedes and Seehusen 2011).

All stakeholders (local, regional and global) should be engaged in the formulation and implementation of PES systems. Full stakeholder awareness and participation contributes to credible, accepted rules that identify and assign the corresponding responsibilities appropriately and that can be effectively enforced (Farley and Costanza 2010).

In spite of the successful initiatives already under way, the law project n° 792/2007 (Brasil 2007), which establishes the national policy on payment for environmental services, is still in progress. In the review of the Brazilian forest code was included in the text a programme to support and encourage the preservation and restoration of the environment covering the following categories and lines of action:

1. Payment or incentive for environmental services, monetary or not, conservation activities, improvement of ecosystems and environmental services provision
2. Compensation for environmental conservation necessary to achieve the objectives of this law
3. Incentives for commercialisation, innovation and acceleration of recovery actions, conservation and sustainable use of forests and other forms of vegetation

Therefore, PES systems are booming in Brazil and represent an important advance in Brazilian environmental policy since they have a vast potential for recognition of small farmers for their actions in forest preservation and maintenance of environmental services.

20.2.3 Low-Carbon Agriculture

Low-carbon agriculture (LCA) is part of a sectorial plan (inside the Brazilian national policy on climate change) that aims to reduce GHG emissions and adapt agricultural activities to mitigate climate change. The LCA gives financial incentives and support for farmers that adopt good agricultural practices (Mozzer 2011). Six sustainable practices are encouraged in this programme: no-tillage, recovery of degraded land, crop-livestock-forest integration, forest plantations (mainly with Eucalyptus and Pinus), biological nitrogen fixation and animal wastewater treatment (Brasil 2011b).

In the LCA programme, forests have a minor role in the mitigation of climate change as this programme focuses largely on the agriculture segment. Besides this, the participation in the CDM or in the voluntary market is only clearly mentioned for the animal wastewater treatment. Other activities, including the ones with forests, are not considered in this context.

20.3 Forest Carbon Credits Generation Focused on Small Farmers

20.3.1 The Importance of Cross-Sector Partnerships

The first step to facilitate the integration of small farmers in this market is to guarantee access to information and specialised technical assistance through cross-sector partnerships between state institutions, universities and municipal city halls.

Kolk et al. (2008) suggest that cross-sector partnerships are important instruments to achieve the Millennium Development Goals and to deal with questions related to the global development. Forsyth (2007) highlights that this collaboration may be a way to reduce costs and to increase local representation regarding the social and developmental benefits associated with CDM activities.

In the case of small farmers, these partnerships are essential in order to face the challenges regarding forestry CDM projects, especially related to bureaucracy and transactions costs. In a quick review of forestry projects already registered (large and small scale), we notice that in most of the projects exist some kind of cross-sector partnerships between community groups and federal/private institutions.

Brazil only has two CDM afforestation/reforestation projects registered. The Plantar S/A project has not clearly mentioned any partnership with small farmers, which goes against the trends of other projects. This situation is probably associated to the fact that the Plantar S/A project is focused on industry supply. The same applies for the AES Tietê project in which any partnership with small farmers was not declared.

The cross-sector partnerships in several of the registered projects include financial support to the development of the project and transfer of know-how on technical and forestry management issues to local communities. This transfer of knowledge enables the local community to participate more actively in the direction of the project.

However, it is important to consider the participation of local actors not only during the implementation of a forestry project but also during the design phase. This design must assume a decentralised management of the project, including not only transfer of decision-making from the proponents of the project to local actors but also guaranteeing the distribution of resources and benefits among all (Boyd et al. 2007).

20.3.2 The Role of Community Networks

Thomas et al. (2010) asserted that the formation of community networks is essential for the success of the project to meet minimum transaction costs, even for small-scale activities (projects developed or implemented by low-income communities/individuals that result to GHG removal by sink of less than 16,000 tCO_2 per year; UNFCCC 2007).

In Brazil, cooperatives are broadly distributed in the country, with the agricultural sector representing the main branch (OCB 2011a). The agricultural cooperatives have an important economic and social role. In many regions, they represent one of the few opportunities to add value to rural production, as well as the insertion of small and medium producers in concentrated markets (Ferreira and Braga 2004).

Especially for this sector, in 2008 the Organization of Brazilian Cooperatives created a programme to stimulate the participation of cooperatives in the carbon market. Part of this programme is dedicated to forestry CDM, focusing mostly on the development of methodologies and the dissemination of information about the carbon market among the cooperative members (OCB 2011b).

20.3.3 The Need for Financing Options

Other options to facilitate the generation of forest carbon credits by small farmers are to ensure financing options that are directly related to this sector and which take into account the farmers' financial situation. Because of its long investment cycle and high investment risk of forestation, there are few credit mechanisms in place for small farmers (Table 20.1).

The PRONAF Floresta and PRONAF ECO (BCB 2008) are specifically designed for small farmers. They are part of a farm loan linked to the national programme to strengthen family farming (PRONAF – *Programa Nacional de Fortalecimento da Agricultura Familiar*). Their interest rate is the lower among the mechanisms presented, just as the cap. However, the reimbursement deadline is similar to other programmes.

In the LCA programme, as already mentioned, several sustainable practices to minimise the GHG emissions in the agriculture sector were proposed. The credit line associated to the LCA gives financial facilities for farmers to make investments and to incorporate sustainable practices in the property. Inside the LCA, two other programmes are also included: PROPFLORA and PRODUSA (Brasil 2011c).

The PROPFLORA and PRODUSA are credit lines linked to the *Banco Nacional de Desenvolvimento Econômico e Social* (BNDES) and to the *Banco do Brasil*. The PROPFLORA aims to make economically viable small and medium farms and to contribute for the conservation of native forests. Thus, it is expected an increase of settlement on rural areas and the reduction of migration to cities (BB 2011).

The PRODUSA is more focused on agribusiness development, but is also intended for farmers and their cooperatives (BB 2011). Both programmes have higher caps and interest rates than farm loans linked to PRONAF since PROPFLORA and PRODUSA are not designed exclusively for small farmers.

The Mid-West Constitutional Financing Fund (FCO – *Fundo Constitucional de Financiamento do Centro-Oeste*) aims to contribute to the economic and social development of the midwest region of Brazil (Distrito Federal, State of Goiás, Mato Grosso and Mato Grosso do Sul) by financing productive activities in various

Table 20.1 Credit mechanisms for small farmers in Brazil (valid for 2010/2011)

Lines of credit	Goals	Budget	Cap	Interest rate	Deadline
PRONAF Floresta	Agroforestry systems Sustainable ecological harvest Rehabilitation and maintenance of preserved areas Enrichment of forest areas	n/a	R$20,000 (~ USD 11,697)	1.0% per year	Until 12 years
PRONAF ECO	Creation and maintenance of forests for generation of wood products and non-timber forest products	n/a	R$80,000 (~ USD 46,789)	1.0–2.0% per year (depending on the financed value)	Until 12 years
LCA	Stimulate the sustainable agriculture in the country	R$2.0 billion (~USD 1.2 billion)	R$1.0 million (~USD 585 million)	5.5% per year	Until 15 years
PROPFLORA	Creation and maintenance of forests for industrial or agricultural purposes Rehabilitation and maintenance of preserved areas Agroforestry systems Creation and maintenance of oil palm	R$150 million (~ USD 87.7 million)	R$1.0 million (~USD 585 million)	5. 5% per year	Until 15 years
PRODUSA	Stimulate actions for environmental sustainability in the agribusiness Rehabilitation of degraded areas for increasing agricultural productivity on a sustainable basis Support actions for regularisation of rural properties according to environmental legislation Reduce the pressure for logging in new areas	R$1.0 billion (~USD 585 million)	R$1.0 million (~USD 585 million)	5.5% per year	Until 15 years
FCO nature conservation	Rehabilitation and conservation of degraded areas Support the adaptation of production processes of appropriate technologies for the environmental conditions of the region Encourage the establishment of forest enterprises, focusing on generation of employment and income Support projects involving carbon sequestration and GHG emission reduction	n/a	R$20 million (~USD 11,697,274)	5.0–8. 5% per year (depending on the activity and income of farmer)	Until 20 years

sectors, including agribusiness and agriculture (MI 2011). One of its lines of credit is nature conservation (FCO nature conservation).

This programme has a broader scope compared to the others presented. Inside, it is possible to finance conservation and agribusiness projects. Moreover, this is the only credit line for farmers which explicitly mentions the support for projects involving carbon sequestration and GHG emission reduction (MI 2011). As FCO nature conservation encompasses a greater number of project possibilities, its cap and interest rate are also higher than the other credit lines in Table 20.1.

20.3.4 The Voluntary Carbon Market Is Also an Option

The development of projects destined for the voluntary carbon market would also be an alternative for small farmers. This market includes all carbon offset trades that are not required by regulation. Thus, individuals and institutions can purchase carbon credits to offset their emissions (Hamilton et al. 2007).

The transaction costs in the voluntary carbon market are usually lower than under Kyoto Protocol mechanisms, due to the less formalised requirements. However, this will vary according to the standard used and the project type. Projects under a high-quality voluntary scheme or standard may have transaction costs similar to those projects under the CDM (Neeff et al. 2007).

In addition to a possible reduction in transaction fees, there is the chance of experimentation and innovation. Since voluntary markets do not have many established rules, they allow the development of new procedures, methodologies and technologies (Kollmuss et al. 2008). Various forest practices besides A/R are considered eligible in the voluntary carbon market, fact that expands the possibilities of forestry project establishment by small farmers.

Nonetheless, Merger and Pistorius (2011) still regard the forest sector in the voluntary carbon market as immature. Some constraints highlighted by the authors are related with political issues, technical complexity, high costs and lack of transparency on quality assurance. Forest projects, whether under compliance or voluntary markets, are intrinsically more complex than non-forest projects, due to the difficulties associated with MRV (monitoring, reporting and verification).

Further, the absence of reliable quality assurance mechanisms and the ineligibility of forest carbon credits in the EU ETS compliance market compromise the sale of these credits in the carbon market. Bureaucratic load should also be considered as it will vary depending on the standard selected. All these issues must be taken into account by small farmers before developing a project under the voluntary carbon market.

According to Peters-Stanley et al. (2012), the most popular forestry projects in the voluntary carbon market are afforestation/reforestation (10%), avoided deforestation – REDD (9%) and forest management (4%). The carbon credits originated by those projects correspond to 23% of all credits transacted in 2011.

Brazil currently has 13 forestry projects in the voluntary carbon market (pipeline or operational phase), from a total of 226 forest-related projects in the world (Ecosystem Marketplace 2011). Some of those Brazilian projects have as main goal the creation of protected areas instead of dealing specifically with rural settlements. Nonetheless, in all of them is expressed the concern with communities around the area of the projects.

Most of the projects propose training and workshops to improve livelihood and stimulate sustainable land use. Small farmers in this context may benefit from these opportunities that arise with the implementation of a project. Furthermore, it is important to mention that small farmers, through cooperatives and cross-sector partnerships, could develop projects focusing on the voluntary carbon market, so that multiple benefits are generated in the environmental, economical and social sphere. However, we stress once again that the constraints mentioned must be considered by small farmers before designing and implementing a forest project under the voluntary carbon market.

20.3.5 Other Economic Incentives for Small Farmers

20.3.5.1 Payment for Ecosystem Services

Small farmers play an important role in the context of payment for ecosystem services (PES), especially in areas where the population is directly dependent on forest resources and non-timber forest products, like in the Amazon region. Shanley et al. (2012) point out that small farmers can have a great knowledge about lesser known species, supply local markets with food of high nutritional value and protect essential environmental services. These authors examined the use of three species that are valued for timber and non-timber forest products in the Amazonian state of Pará. For the species analysed, which still do not have a formal management plan, the authors found that small farmers were able to develop innovative techniques in multiple use forest management.

Great effort has been made by governmental and non-governmental agencies for the adoption of PES that encourages small farmers in the protection of forest fragments. In Brazil there is a discussion about PES at the Environment Ministry, which seeks to ensure environmental conservation through a new system within the principle "conservative-receiver."

In this principle, those farmers who have a different management of their farm in relation to soil, water and forests receive financial incentives for ensuring the ecosystem services promotion. Table 20.2 shows some PES initiatives that are implemented in Brazil.

Considering the scope of our study, more focus will be given on PES related directly or indirectly with carbon sequestration and storage. As a case study we present some carbon PES initiatives that are being implemented in the Atlantic Forest in Brazil (Table 20.3; Guedes and Seehusen 2011).

Table 20.2 PES Projects implemented in Brazil

Project name	Location	Goals	Amount paid	Financial sources	Biome
Forest grant *(Bolsa Floresta)*	Amazonas	Promote the involvement of traditional communities living in protected areas to reduce deforestation and to value the standing forest	R$50.0/month.family (~USD 29.2/ month.family)	State fund for climate change, the Amazon fund, sustainable Amazon foundation	Amazon
Green grant *(Bolsa Verde)*	Minas Gerais	Encourage farmers to preserve important areas for the protection of water resources, biodiversity and ecosystems through financial benefit	R$200.0/ha.year (~USD 116.9/ha. year)	Fund for recovery, environmental protection and sustainable development of watersheds in Minas Gerais; compensation for the use of natural resources	Atlantic Forest, Cerrado (Brazilian savanna)
Water conservator *(Conservador das águas)*	Minas Gerais	Increase the state vegetation cover, implant ecological corridors, reduce pollution caused by erosion and lack of sanitation, ensure sustainable management of systems and practices implemented	R$159.0/ha.year (~USD 93.0/ ha.year)	Fund for recovery, environmental protection and sustainable development of watersheds in Minas Gerais, water supply company for São Paulo state; charging for water use	Atlantic Forest
Water producer *(Produtor de água)*	Espírito Santo	Encourage farmers to contribute to the reduction of erosion and increase water infiltration, through conservation of standing forest	R$80.0 up to 300.0/ ha.year (~USD 46.8 up to 175.5/ ha.year)	Water resources fund of Espírito Santo state; royalties from oil, natural gas and electricity sector; capital acquired in environmental penalties	Atlantic Forest

Table 20.3 PES Projects focused on carbon being implemented in Brazil

Project name	Location	Goals	Deployment costs	Maintenance costs	Amount paid
Carbon park: forest restoration on the state park of Pedra Branca	Rio de Janeiro	Recover degraded areas within the park, through assisted regeneration and enrichment with native species, encouraging the creation of ecological corridors	R$27.000,0/ha (~USD 15,791.3/ha) including costs of maintenance and certification	Included in the deployment cost	Undefined
Carbon, biodiversity and ecological corridor community on Monte Pascoal	Bahia	Protect important areas of Atlantic Forest through the formation of ecological corridors between two national parks	R$ 15.000,0/ha (~ USD 8,772.9/ha)	R$6.000,0/ha.year (~USD 3,509.2/ha.year; 3 years)	Undefined
Ecological agriculture and socio-environmental services	Paraná, Rio Grande do Sul	Avoided deforestation of protected areas; rehabilitation of degraded areas through the implementation of agroforestry systems	R$650,0/ha (~USD 380.1/ha)	R$500,0/ha.year (~USD 292.4/ha. year)	Approximately R$257.0 (~USD150.3/ha; for up to 3 years)
Carbon, biodiversity and income on the Pontal of Paranapanema	São Paulo	Stimulate the generation of income through the establishment of agroforestry systems on small farms	R$5.000,0/ha (~USD 2,924.3/ha)	R$1.800,0/ha.year (~ USD 1,052.7/ha; 3 years)	Undefined
Safe carbon programme	São Paulo	Create carbon stocks in areas intended for dairy cattle	R$10.000,0/ha (~USD 5,848.6/ha)	R$256,0/ha.year (~USD 149.7/ha. year)	Farmers will receive an amount equal to C stored in their forest
Projects to combat global warming on the coastal zone	Paraná	Restoration of degraded areas and conservation of protected areas to offset GHG emissions	USD 230/ha	USD 45/ha	Undefined

Guedes and Seehusen (2011) report that PES initiatives focused on carbon in the Atlantic Forest are structured in a peculiar way and do not show payment differentiation according to characteristics or quality of services from different providers. The authors highlight that normally PES initiatives do not transfer the total value of carbon sequestered to the project participants. Part of this resource must be used to recover the investment and administrative costs of the applicant organisation.

Allied to these PES initiatives, it is of utmost importance to invest in the dissemination of knowledge and capacity building. These are inputs which are presently poorly supported by the low levels of investment in extension services in Brazil and other developing countries. Only with education it will be possible to secure the ecological integrity of forests and the improvement of small farmers' livelihoods.

20.3.5.2 Reducing Emissions from Deforestation and Forest Degradation (REDD)

Reducing Emissions from Deforestation and Forest Degradation (REDD) is a mechanism that gives an economic value to the carbon stored in the forests. It represents an economic incentive for forest-rich countries to reduce GHG emissions from forest areas while promoting sustainable development.

More recently, the concept of REDD was expanded to also include the sustainable management of forests and the enhancement of forest carbon stocks (REDD+). REDD+ has arisen as a key issue in the international climate change negotiations and entered into the public media. Forest ecosystems cover one-third of the Earth's land surface, storing more carbon than both the atmosphere and the world's oil reserves combined. However, ongoing deforestation and forest degradation, which the FAO estimates to amount to 5.2 million hectares net per year, accounts for up to one-fifth of global anthropogenic carbon emissions (WBI 2011).

Brazil is composed of six different biomes (Amazon forest, Cerrado, Caatinga, Atlantic Forest, Pantanal and Pampa) which were or are under severe deforestation pressure. Initiatives that stimulate the reduction of deforestation and the restoration of degraded areas, such as REDD/REDD+, are important mechanisms that can supplement the Brazilian government actions against the drivers of deforestation.

Currently, seven REDD projects are under development in Brazil. A brief description of each of these projects is presented in Table 20.4 (Cenamo et al. 2009).

Although hosting some REDD projects, Brazil still does not have a REDD's national strategy, nor a political system to regulate this mechanism. This issue is under discussion in the political sphere of the Brazilian government, and some studies have been developed to support the debate (e.g. CGEE et al. 2011).

In this new national offset policy, two key issues need to be addressed to provide incentive for the participation of small farmers: (1) the empowerment of local governance and (2) the guarantee that REDD benefits really get to the individuals that are protecting the forests. For this, Kanowski et al. (2011) suggest the appliance of some principles of good forest governance (e.g. accountability, inclusion and transparency) to the existing national and sub-national commitments for forest conservation and management.

Table 20.4 Brief description of REDD projects under development in Brazil

Project	Goal	Vegetation type	Size	Net emission reductions	Project status
Acre state carbon project – payment for environmental services	Add value to the standing forests of Acre and turn them into a viable source of environmental services for current and future generations	Amazon forest	5,800,000 ha	62.5 million t CO_2e in 15 years	Design
Ecomapuá Amazon REDD project	Conservation and restoration of an Amazon forest area that belonged to a timber company before the purchase of the property by the project owners	Amazon forest	94,171 ha	6.0 million t CO_2e in 20 years	Design
Genesis REDD project	Guarantee the protection of a natural Cerrado area located in the *Serra do Lajeado* environmental protected area located in the state of Tocantins, northern Brazil	Cerrado (Brazilian savanna)	121,415 ha	57,389 tCO_2e in 20 years	Implemented/ under validation
Avoided deforestation on small rural properties in the region of the Trans-Amazon highway	Transform the historic model of development for rural properties in the region, currently based on slash and burn agriculture with low productivity and minimal value added production, into a model that primarily involves improvements in agricultural and ranching practices and decreasing pressures for new deforestation	Amazon forest	31,745 ha	3,136,953 tCO_2e in 10 years	Design
Juma reserve REDD project	Consists of the creation and implementation of a sustainable development reserve, to contain deforestation on a region under strong pressure of land use, located in the Novo Aripuanã municipality, in the south of Amazonas State, Brazil	Amazon forest	589,612 ha	189 million t CO_2e in 44 years	Validated/sale of VERs (voluntary emissions reductions)

(continued)

Table 20.4 (continued)

Project	Goal	Vegetation type	Size	Net emission reductions	Project status
Conservation of the Atlantic rainforest, pilot project for reforestation in Antonina and the action project against global warming in Antonina	Transform areas originally used for raising buffalo into privately owned nature reserves. Implement reforestation, protection and enforcement measures against land grabs and impacts from external activities, as well as degradation caused by the buffalos in primary forests. This is a collection of three projects that takes place in different Atlantic rainforest locations in the municipalities of Antonina and Guaraqueçaba, in the State of Paraná, in southern Brazil	Atlantic forest	Atlantic rainforest conservation project: 8,600 ha Pilot project for reforestation in Antonina: 3,300 ha Action project against global warming in Guaraqueçaba: 6,700 ha	Atlantic rainforest conservation project: 181,095 tCO_2e in 40 years Pilot project for reforestation in Antonina: 65,456 tCO_2e in 40 years Action project against global warming in Guaraqueçaba: 137,713 tCO_2e in 40 years	Implemented/sale of VERs
Suruí project	Protect the Indigenous Territory *Sete de Setembro* which is currently under great threat of deforestation from land grabbing and illegal logging. The project is located in the municipalities of Cacoal and Espigão d'Oeste in Rondonia State and Rondolândia in Mato Grosso State	Amazon forest	248,000 ha	16.5 million t CO_2e in 44 years	Design

These principles would guide actions to promote the empowerment of local communities and prevent the misuse (e.g. elite capture, corruption) of the financial resources available under REDD projects. Another point that must be taken into account is the impact of the revised forest code in REDD+ projects in Brazil. As already mentioned before, small farmers will be exempted to recover areas that were illegally deforested until July 22, 2008. Thus, small farmers that kept their standing forests following the prescriptions of the previous forest code will receive no reward. On the other hand, small farmers who have illegally logged their forest areas will gain amnesty. This contradictory situation goes against the principles of REDD+ and may affect the image of Brazil in the COP 17 and in the fourth Earth Summit in 2012.

The establishment of an economy based on the valuation of forest and its environmental services will only be possible with the inclusion of different parts of society. The small farmers must not be left apart from this process.

20.4 Conclusion

Offering alternatives for the participation of small farmers in the forest carbon market is of great value in developing countries, such as Brazil, where there are high rates of poverty, illiteracy and environmental degradation. However, there are still many barriers to the inclusion of this segment of society in the forestry carbon market.

Strong action needs to be taken by the Brazilian government to overcome critical challenges faced by programmes for the generation of carbon credits. This is necessary to ensure that mechanisms are in place to enable small farmers and the forestry sector as a whole to take full advantage of the market.

Initiatives to increase access to information are necessary, since most small and medium farmers are unaware of the steps needed to develop and implement CDM projects. The training of agricultural technicians by local, state and federal agricultural extension services would be an alternative to promote this. After the training, agricultural technicians will be qualified to provide the necessary technical information to small farmers for the development of CDM projects.

Policies to reduce the project transaction costs would also be important as Brazilian small farmers have a low income. Different approaches could be included in these policies in order to lower the project costs: (1) empowerment of public environmental agencies to assist in the development of CDM projects and (2) provision of qualified technicians to give support in all steps of the project cycle, including the identification of potential buyers for carbon credits.

The creation of more financing options is also important. Nevertheless, the lines of credits should be specifically for carbon credit projects, considering both large- and small-scale projects. In the case of small-scale projects, special conditions should be given to small and medium farmers.

As Brazil has a very large territory, a regional assessment of the potential to generate forest carbon credits in the country would be recommended. An evaluation of small farmers' communities in each of these regions could also be carried out. Based on the results, it would be possible to establish national policies appropriate for each region, respecting differences among biomes, climate and even cultural aspects.

Further, to ensure the effectiveness of forest carbon projects in Brazil, strong governmental actions are necessary. The enforcement of environmental laws, through supervision and monitoring by municipal, state and federal agencies are a key issue. The definition of land tenure is also important, especially in the north region of the country, where most of the small farmers do not have the title of the land. The inclusion of all the parties involved or affected by the carbon projects must be assured, especially the economically marginalised groups. All the benefits generated by the project must be distributed among all parties.

The stimulation of forest carbon projects is a way to collaborate on the development of the country. When those projects are designed respecting local needs, the chance of success increases, just as the benefits that arise after the project implementation. Through forestry projects, it is possible to keep the man in the field, generate income, knowledge and also contribute to mitigate the intensification of the greenhouse effect.

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Chapter 21
Carbon Sequestration Projects in Peruvian Tropical Forests

Teresa Rojas Lara and Thomas Berger

21.1 Introduction

In many tropical countries, projects that protect or increase carbon stocks receive high development priority. Payments for ecosystem services could generate significant revenue, contribute to the alleviation of poverty and preserve various ecosystem services (Baker et al. 2010).

Fifty-four per cent of the Peruvian territory is covered with forests, and around 40% of the population still lives in poverty (INEI 2009). The rate of deforestation is around 150,000 ha per year (Velarde et al. 2010). Deforestation and land use change account for around 50% of Peru's greenhouse gas emissions (MINAM 2009). Most deforestations in Peru are the result of subsistence agriculture (slash-and-burn agriculture), which is exacerbated by migration of farmers from the highlands, as well as development activities, such as commercial agriculture, logging, mining, gas and oil operations and road construction (Velarde et al. 2010). Moreover, forest areas are also confronted with the problem of illegal coca cultivation. Many farmers are attracted by the potentially high revenues of this crop, especially because alternative economic opportunities are scarce (UNODC 2011). Payments for environmental services (PES) can be part of the solution to this problem. Currently, some efforts to mitigate greenhouse gas effects through the establishment of forest-based systems are being developed, involving poor rural communities. It is, however, necessary to determine whether these projects can increase farmer's income. Based on

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insights from institutional economics, this chapter presents analyses of data obtained from surveys carried out in two regions: (1) the Peruvian Amazon and (2) Peru's Pacific coast. A mathematical programming model is applied to assess the impacts of PES on the economic situation of households involved in PES schemes. An analysis of the institutional context in which PES schemes are implemented in Peru is presented, and householders' perception of PES projects and their possible implementation are discussed.

In Sect. 21.2, the theoretical framework used to analyse PES draws the links between PES schemes and poverty reduction. Theoretical concepts from institutional economics are presented along with a discussion of the implications of the presence of transaction costs for these schemes. Section 21.3 describes the research context of the present study and the current legal and institutional framework of Peru, which could have a potential role to play in the implementation of PES forest-based schemes. In Sect. 21.4, the methodology is explained, focusing on household modelling with mathematical programming. Section 21.5 presents the results of this study, and Sect. 21.6 draws policy-relevant conclusions from the results obtained.

21.2 Theoretical Framework

Externalities are costs or benefits arising from an economic activity, in most of the cases attributed to human activities, that affect somebody other than the people engaged in the economic activity and are not fully reflected in prices (Perman et al. 2003). Externalities are recognized as exceptions to the standard economic theory and represent an important class of market failure in the field of environmental and resource economics (Woerdman 2004). Different policy instruments such as taxation, subsidies, tradable permits or charges are possible solutions to overcome these market failures (Perman et al. 2003). Payments for environmental services (PES) belong to the group of market-based mechanisms and have been promoted as an environmental policy instrument for climate mitigation (Wunder 2005).

As mentioned in the introduction of this book, these schemes are used as tool to finance conservation and management of natural resources in developing countries (Pascual et al. 2010). There are studies that show how they can improve the welfare of rural people and to play a role in solving social conflicts (Wunder 2005; Pagiola et al. 2005; Cacho et al. 2003; Rosa and Dimas 2003). Policy makers in several developing countries are enthusiastic about the potential for PES schemes to mitigate environmental degradation and combat rural poverty (Landell and Porras 2002). Nevertheless, the impacts of PES schemes on poverty depend on whether or not the poor can benefit from markets for environmental services. Poor smallholders in developing countries often face constraints related to market access, lack of willingness/ability/capacity to pay for environmental services, high transaction costs, insecure property rights and inadequate policy and legislation (Wunder 2005; Scherr et al. 2007).

Institutional factors play an important role as to whether poor people can actually be involved in and benefit from these schemes (Smith and Scherr 2003; Bracer et al. 2007). Some studies show that PES schemes can positively strengthen existing institutions for ecosystem conservation, through the provision of a framework for management and regulation and by providing incentives to change behaviour (Corbera et al. 2009; Engel et al. 2008). PES schemes require the participation of various stakeholders, and transaction costs can act as a barrier to involvement of small stakeholders (Scherr et al. 2007). This is considered in the theory of institutions of which the transaction cost theory constitutes an important component (North 1990). In case of PES schemes, transaction costs involve costs of drawing attention to potential buyers, costs of working with project partners and costs of ensuring parties fulfil their obligations (Bracer et al. 2007). Taking this into consideration, reducing transaction costs is an important consideration for the potential viability of PES schemes that will impact their potential to deliver new sources of income to rural communities (Woerdman 2004; North 1990). Participation of local communities in these markets for environmental services can contribute to a reduction in transaction costs, specifically of monitoring and compliance activities (Smith and Scherr 2003; Ostrom 1990; Ballet et al. 2007).

21.3 Research Context and the Case Studies

Peru is extremely biodiverse and has a wealth of natural resources. However, its natural resources have not been used efficiently to develop the economy, and the national economy has relied heavily on mining since colonial times (World Bank 2007). According to the World Bank, the economy of Peru is classified as upper middle income and is the 42nd largest in the world (World Bank 2011). Peru is a market-oriented economy with a high level of foreign trade. The main exports are copper, gold, zinc, textiles and fish meal. Although exports have provided significant revenue, the distribution of income remains very skewed. Around 40 per cent of the Peruvian population lives below the national poverty line; of this, 14% live in extreme poverty. Although Peru ranks 80 of 180 countries, with a Human Development Index score of 0.723 (UNDP 2009), it is characterized by stark disparities, reflected in a Gini coefficient of 0.48 (1 indicating complete inequality). Furthermore, 42% of the population cannot cover the minimum required caloric intake (2,100 kcal) (World Bank 2011; United Nations Development Programme 2009; INEI 2009).

Fifty four per cent of Peru's territory is covered by forest; however, the contribution of the forestry sector to the Peruvian GDP is only 1% (FAO 2009). Moreover, deforestation and forest degradation from subsistence agriculture are significant threats to forest estate in Peru. A leading cause of deforestation and forest degradation in Peru is the migration of farmers from Andean regions to the Amazon basin, who take advantage of Peru's land-tenure law which allows people to own land by occupying it for 5 years. Deforestation and degradation are also the result of development activities, especially logging, commercial agriculture, mining, gas and oil operations and road construction. The indirect drivers of deforestation are unclear land tenure, limited access to information, lack of certainty in the forest legislation and limited involvement of stakeholders in decision making (Velarde et al. 2010).

Nevertheless, during the last years, Peru has tried to implement an environmental management framework in order to overcome these problems. Diverse entities and legal instruments have been created to address specific issues, ranging from forests and biodiversity to the regulation of sectorial activities and the consolidation and integration of policy and institutions involved with natural resource management within a national environmental system.

21.3.1 Peruvian Legal Framework

Governments have an important role in establishing the legal framework that defines the governance of natural resources, such as institutional arrangements, responsibilities, requirements, contracts and mechanisms for resolving conflicts or disputes (Bracer et al. 2007; FAO and ITTO 2010). Although globally there are many individual cases of PES schemes operating without a formal legal framework, greater impacts for ecosystem service provision and the generation of local benefits require a clear legal framework (Bracer et al. 2007; Smith and Scherr 2003).

The Peruvian government has signed many international agreements and treaties relevant for the implementation and regulation of PES and REDD schemes.[1] Peru has subscribed the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol and participates in the international debate on the implementation of a new binding agreement on climate change (MINAM 2009). At the Conferences of the Parties (COP) 15 of UNFCCC, the Peruvian government submitted a proposal to reduce gradually its greenhouse gas emissions, originating from deforestation activities to zero. According to the statement of the Minister of Environment, this goal could be reached in 10 years[2] with the support of international financial aid (MINAM 2009).

In Peru, the state has ownership on natural resources, including forests. The most important laws that could contribute to the implementation of forest-based PES schemes include the Environmental and Natural Resources Code (Legislative Decree No. 613), the Natural Protected Areas Law (Law No. 26834), the Forestry and Wildlife Law (Law No. 27308), the National System for Environmental Impact

[1] Peru has signed the Convention on Biological Diversity, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the ILO C169 Indigenous and Tribal Peoples Convention.

[2] In Peru, MINAM is promoting REDD schemes, due to the high amount of tropical forest (Amazon) and the lower cost of conserving forest as compared to afforestation and reforestation activities (Capella and Sandoval 2010). MINAM is interested in implementing PES schemes through a national protected areas network, including schemes to fund indigenous communities to preserve standing forest like the "Programa Conservando Juntos" and "Programa de Bosques."

Evaluation Law (Law No. 27446), the National Environmental Management System Framework Law (Law No. 28245), the General Law of the Environment (Law No. 28611) and the Organic Law for Sustainable Use of Natural Resources (Law No. 26821) (Capella and Sandoval 2010; USAID 2010). The Peruvian Ministry of Environment (MINAM) has already submitted a proposal entitled "Environmental Services Law," which is currently being considered by the Peruvian congress. This bill aims to establish the general framework for regulating the provision and use of environmental services in order to contribute to the conservation, restoration and sustainable use of natural resources and biodiversity in Peru.

21.3.2 Peruvian Institutions

Although institutions and organizations have different meanings, they are often understood as being the same. Organizations are material entities and include political, economic, social and educational bodies. Institutions can be defined as entities that devise constraints that structure political, economic and social interactions. Institutions achieve their objectives through informal (e.g. traditions, cultural values) and/or through formal rules (e.g. legal rules) which govern individual behaviour and structure social interactions and thereby provide an institutional framework (Woerdman 2004; North 1990). An adequate institutional framework can enable the minimization of transaction costs of natural resource management and specifically of PES schemes. Bracer found various ways to reduce these costs and risks through institutional mechanisms in pilot PES projects oriented towards poor sellers (Bracer et al. 2007).

Peru has done some efforts to consolidate an organizational structure that can respond to the country's environmental necessities. The Ministry of Agriculture (MINAG), through its General Directorate for Forestry and Wildlife (DGFS), establishes national policies related to the promotion, management, monitoring and evaluation of forest resources and coordinates with regional counterparts for the implementation of these policies (Capella and Sandoval 2010). The Ministry of Environment (MINAM), established in 2008, has the administrative authority for the national environmental resources and has responsibility for the evaluation, design and implementation of PES schemes (Capella and Sandoval 2010; USAID 2010). There are public-private organizations providing financial support through projects designed to promote conservation or sustainable forest management. The National Environmental Fund (FONAM) is Peru's focal point for the World Bank Carbon Finance Unit, and the National Fund for Natural Protected Areas (PROFONANPE) is working in 46 of the 63 current natural protected areas (Capella and Sandoval 2010; USAID 2010).

The public ministry, the national policy and the Peru's national ombudsman programme are institutions dealing with environmental offences, whereas the Supervision Office for Forest Resources (OSINFOR) is responsible for the supervision of granted rights under the Forestry and Wildlife Law and under the law for environmental services (Capella and Sandoval 2010). Currently, the central government is transferring forestry management functions to regional governments.

Therefore, regional governments are responsible for granting rights to forest resources, approving management plans. They are able to develop and implement programmes for the sale of environmental services in forest regions and/or protected areas (Velarde et al. 2010; USAID 2010). The regional agricultural directorates are working also at the regional level, granting property titles to indigenous and peasant communities.

The civil society and the private sector are involving actively in these schemes. NGOs have initiated at least 17 carbon-based projects in the country (Baker et al. 2010), while private enterprises have contributed to at least five projects in the country. NGOs are also working on the legal and institutional framework for REDD schemes at the country level. At the local level, several rural communities are carrying out sustainable management activities that help maintain or produce environmental services. Some are organized into rural patrols and forest management committees, which are working on conservation and sustainable forest management activities and also for the control of illegal logging (Velarde et al. 2010; Capella and Sandoval 2010). Nevertheless, more support from the government is necessary to strengthen their activities.

At the same time, some institutions and policies could negatively impact the establishment of PES schemes. With regard to property titles, there is a small percentage of forestry land with land-tenure rights granted by the state. The Ministry of Housing, Construction and Sanitation, through the Agency for Formalizing Informal Property (COFOPRI), is in charge of the national programme for the formalization of property rights. In Amazonia, COFOPRI has not performed too well with respect to forest conservation. The procedures for granting titles are based on land use changes to agriculture activities,[3] which have encouraged deforestation activities[4] (Velarde et al. 2010).

Likewise, the Ministry of Energy and Mining is granting rights for exploration and exploitation of nonrenewable natural resources (hydrocarbons and minerals) in forest areas. This situation is creating an overlap in the use of land,[5] generating

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[3] That is, the Law of Private Investment in the Development of Economic Activities in the Lands of the National Territory and of Rural and Indigenous Communities (Law No. 26505) and Legislative Decree of Investment Promotion Law in the Agricultural Sector (Legislative Decree 653) (Velarde et al. 2010).

[4] During the data collection carried out for the present study in the Peruvian Amazon, the Campo Verde Beekeeper Association, for example, mentioned its desire to conserve 600 ha of primary forest, which is not recognized by COFOPRI as land suitable for the entitled.

[5] There are 64 active oil and gas blocks under contract with multinational companies in the Peruvian Amazon, covering an area of two thirds of the Peruvian Amazon. Of these, 20 blocks overlap communal reserves and conservation reserve zones, 58 overly lands titled to indigenous peoples and 17 blocks overlap areas that have been proposed as reserves for indigenous groups in voluntary isolation (Finer et al. 2008). During the data collection in Campo Verde, one native community received an official visit from an oil company, which will start exploration in the area. In San Lorenzo, Piura, a fruit-growing region, close to the research area, the Peruvian government granted three blocks of land to Manhattan Minerals of Canada in 2000. In San Lorenzo, agricultural production creates about US$2 billion in annual revenues and permanently employs roughly 15,000 people, and more during the harvest season. In 2001, the local population held a referendum, and the result was an overwhelming rejection of the proposed mining project (No Dirty Gold 2008). The proposed mining project was abandoned in 2009.

enormous pressure on forest, including natural protected areas and indigenous community lands. Although coordination between stakeholders and relevant actors is mandatory, in practice, this is not achieved (USAID 2010). Furthermore, the Ministry of Economy and Finance and the Ministry of Transportation and Communications are expanding the road infrastructure in forested areas, which has led and will lead to dramatic changes in forest cover. Biofuel production has been promoted in Peru as a mitigation option for climate change. In 2007, the government promulgated legislation requiring mandatory blending of 5% biodiesel in diesel by 2011 and 7.8% ethanol in gasoline by 2010. In order to accomplish this goal, growth of crop areas of oil palm, jatropha, canola and sugar cane is necessary. This policy is a potential threat to forest areas that encourages changes in land use practice (Velarde et al. 2010).

21.3.3 The Case Studies

Two case studies of forest-based projects oriented to timber production and carbon markets were selected as case studies. The project "Ignacio Tavara Dry Forest Reforestation, Sustainable Production and Carbon Sequestration" covers 8,989 ha on communal land of the Ignacio Tavara Pasapera community. It is located in Chulucanas District, in the department of Piura, where precipitation is 327 mm/year and the mean annual temperature is 25.7 °C (AIDER 2010). This project was registered in 2009 with the Clean Development Mechanism (CDM) of the UNFCCC, and it employs native species from the dry tropical forest, such as algarrobo (*Prosopis pallida*), zapote (*Capparis scabrida*) and overo (*Cordia lutea*). The stakeholders involved are the Ignacio Tavara community, the AIDER and the Peruvian National Environmental Fund (FONAM).[6] The community holds the rights to carbon credits, whereas AIDER (an NGO) is providing technical support. FONAM is in charge of Peruvian portfolio of carbon projects, therefore is negotiating with different private investors to sell the carbon credits. According to the Peruvian land use map, the project area is entirely classified as protection land with grazing aptitude and low agriculture quality. The community has 8,589 inhabitants and an area of 52,269 ha divided in 16 villages. The land-tenure system is based on communal lands. Livestock and crop production are oriented towards subsistence. Livestock raising depends on algarrobo production and seasonal pastures, whereas agricultural activities take place over small areas, only during the rainy season (January–April). Beans is the most important cash crop, while maize and watermelon are grown for local consumption. Forestry activities are restricted; timber harvesting is not allowed without an approved management

[6] A formal agreement has been signed between the community, AIDER and FONAM to assure that the project will continue for the next 40 years.

plan (AIDER 2010). Nevertheless, evidence of illegal logging practices was found during data collection.

The project “Reforestation of Degraded Areas in Campo Verde with Native Species” is located in the district of Campo Verde, department of Ucayali (Peruvian Amazon), where annual precipitation is 1,862 mm, with a mean annual temperature of 27 °C. In this area, deforestation has been caused mainly by the expansion of the agricultural frontier (slash-and-burn agriculture), forest substitution for illegal crops (e.g. coca) and conversion of secondary forest into grassland (Ramos 2009; Velarde et al. 2010). The carbon project belongs to a private enterprise which owns 16,000 ha of degraded land, of which 2,600 ha have been reforested with mahogany (*Swietenia macrophylla*), tornillo (*Cedrelinga catenaeformis*), marupa (*Simarouba amara*) and guaba (*Inga edulis*). This project has sold some Voluntary Emissions Reductions (VERs) to voluntary carbon markets. The stakeholders involved are SFM-BAM enterprise, which holds the rights of the project and carbon credits, AIDER and FONAM (Fondo Nacional del Ambiente). The area contains 15 villages, where farmers are primarily engaged in subsistence agriculture. Cassava and rice are the most important cash crops, whereas maize and bananas are grown mainly for local consumption. Livestock is another important activity, where around 30% of the families have cattle (SFMBAM 2010). In this area, illegal coca cultivation is widespread; therefore, the government is trying to control these activities, using a variety of methods, which range from military interventions to introduction of productive projects.[7]

21.4 Methodology

21.4.1 Data Collection and Research Design

The data collection was carried out in 2010, using a combination of qualitative and quantitative research methods. The qualitative research concentrated on the institutional setting for natural resource management. For this purpose, in-depth and key interviews have been undertaken. For the quantitative research, 163 household interviews were carried out using a detailed structured questionnaire. For each individual household, all the necessary data for household modelling was collected in this survey. Secondary data provided by the NGO AIDER and SFM-BAM enterprise was obtained in order to quantify the amount of carbon sequestrated for each project. The household surveys carried out in the Ignacio Tavara community (Piura) and in Campo Verde (Ucayali) took the form of random samples of 90 households

[7] A cocoa programme is established with the support of United States Agency for International Development (USAID) through the Alternative Development Program (ADP), providing technical and financial support (fertilizers and pesticides) during the first 3 years of the plantation. Ucayali´s regional government is implementing a palm oil programme in the area, working in the same way as the cocoa programme.

in 16 villages in Piura and 73 households in Campo Verde. The information gathered focused on general aspects of the household and farm characteristics, availability of land resources and their use, agricultural production activities, forest use, carbon projects, assets and savings, credit and institutional embeddedness as well as households' perception of the forest and its functions.

21.4.2 Mathematical Programming

Mathematical programming (MP) was chosen to evaluate the behaviour of the farmers and their resource allocation. MP is a simulation approach for finding the best course of action, in terms of maximum profit or minimum costs, taking into account the various constraints that households face in their decision making (Hazell and Norton 1986). The approach has been favourably used to assess the potential smallholder's adoption of forestry technologies, taking into account socioeconomic characteristics and the influence of policy activities, that is, for the case of carbon payments (Bellow et al. 2008; Vosti et al. 2002). Nevertheless, it is important to note that as with all modelling methods, there are some limitations, like the assumption of certain values and preferences when specifying the objective function, the possibility of non-linearity and feedback between variables, as well as the dynamics of systems.

As an input for the model, the gross margins for the main cropping activities maize, beans (Piura) and cassava and maize and rice (Ucayali) were calculated. In Ucayali, perennial crops also play an important role; therefore, cacao, oil palm and fruit trees were considered as a component of the gross margin. As livestock activities are important in both regions, they were included in the gross margin calculations. The model is designed to maximize the total gross margin of the farm by finding the optimal set of the different agricultural activities under the respective restrictions such as farm size, suitability of the land for various crops, credit limit and family work force. The credit limit is the maximum amount of credit that a household expects to be able to borrow from formal and informal sources. It is considered that the farmer has information about alternative production activities and input and output prices; therefore, risk is not accounted for in the model (Vosti et al. 2002). The model allows off-farm labour activities, but it does not currently incorporate nonagricultural investment such as schooling.

21.4.3 Carbon Accounting

The methodology used for accounting the amount of sequestrated carbon is AR-AM003: "Afforestation and reforestation of degraded lands through tree planting, assisted regeneration and control of animal grazing" (UNFCCC 2006). This is one of the methodologies approved under the Clean Development Mechanism by the UNFCCC for forest projects. There are five carbon pools: living biomass (above and below ground), dead biomass (dead wood and leaf litter) and soil carbon. For the purpose of this study, only living biomass is considered. The calculation of carbon sequestrated in living biomass was done using different allometric equations for the different species in both study areas. These equations are mathematical functions that relate oven-dry biomass per tree as a function of a single or a combination of three dimensions (Chave et al. 2005). For this study, the field inventory data was provided by the non-governmental organization AIDER. The biomass can be converted to carbon using a conversion factor of 0.5 g for 1 g of biomass. All carbon measurements for above and below ground were added up to obtain an estimate of the total carbon per hectare. Finally, this amount was converted to CO_2e, which is the basis to calculate the amount of certificates to be obtained for the different forestry systems. This is translated later into monetary terms, using the Certified Emission Reductions (CERs) and Verified Emissions Reductions (VERs) values with a discount rate of 10%. For the mathematical programming model, the net present values were converted to annuities, in order to show the annual payments which the farmer would receive from a 20- and 30-year sequestration project.

21.5 Results

21.5.1 Carbon Sequestration Potential

The results show that in Piura the project removes approximately 498,675 tCO_2e in 20 years or 2.8 tCO_2e per ha per year. The resulting payments for carbon sequestration in turn depend then on the CER and VER prices, which vary considerably on carbon markets. A price of US\$5%/tCO_2e is comparable to the lowest price, whereas US\$30 represents the trading prices in the European Climate Exchange for 2011–2012 in February 2011. At low carbon prices of US\$5 tCO_2e, this would amount to an annuity payment of US\$200,000, at a price of US\$15 tCO_2e to US\$ 600,000 and at US\$30 tCO_2e to US\$1,200,000 for a 20-year project. In Campo Verde, 531,888 tCO_2e will be removed during 30 years of project, which means an annually removal of 6.8 tCO_2e per ha. A low carbon price of US\$5 tCO_2e represents approximately an annuity payment of US\$49,000, at a price of US\$15 tCO_2e to US\$149,000 and at US\$30 tCO_2e to US\$290,000.

21.5.2 Farm Households

In this section, the farm households of the research areas are described, and best decisions are derived from using a mathematical programming model. A first look at the data obtained from the household surveys reveals some basic characteristics

Table 21.1 Characteristics of farm households

	Jose Ignacio Tavara community	Campo Verde
Total land (ha)	2.5	39.9
Cultivated land (ha)	1.82	7.8
Grassland (ha)	0.78	13.8
Forest (ha)	0.06	19.3
Family size (members)	5.2	4.47
Mean age (years)	29.8	28.3
Illiteracy (%)	40	20
% migrant households	3	44
Family labour days per month	68.9	58.8
Credit limit (US$)	925	2,750

Source: Own data

Table 21.2 Total gross margins for household for different carbon payments scenarios

	Jose Ignacio Tavara community	Campo Verde
Baseline	790	1,253
Scenario 1 (d 10%, US$5)	808	N.C.
Scenario 2 (d 10%, US$15)	843	N.C.
Scenario 3 (d 10%, US$30)	895	N.C.

Source: Own data
N.C. not calculated

of the households. Furthermore, in part substantial differences between the households in the two research areas become evident (Table 21.1).

As the table shows, households in the Jose Ignacio Tavara community have lower credit limits and less land at their disposal as compared to Campo Verde. Differences also occur with respect to land use. In the Jose Ignacio Tavara community, more than 95% of the agricultural area is allocated to annual crops, mainly cowpeas and beans, watermelon and maize. Cowpea is the major cash crop destined for local markets, whereas the other crops are used for home consumption. In Campo Verde a mere 20% of the land is dedicated to agriculture activities, whereas 34% of the land is grasslands, leaving the major part covered by forest. The most important cash crops are cassava, oil palm and cocoa citrus and to a lesser degree pepper. Cassava and banana are the most important food staples in the area.

While family sizes and age structure are similar, striking differences between the two areas occur with respect to human capital and migration. In Piura, the rate of illiteracy is 40%, double the rate in Campo Verde. The per cent of households that have migrated to the respective areas, however, is substantially higher in Campo Verde, with 44% versus 3% in Piura. This latter aspect reflects the high importance of immigration from other parts of the country to the Peruvian Amazon.

The baseline of the total gross margin of the main farm activities were calculated (Table 21.2), using three scenarios with three different levels of carbon payment

(US$5, 15 and 30). The gross margins increase with the level of the payment. However, farmers in the region do not only cultivate crops with the highest gross margin. There are some reasons for crop choice that are currently not reflected in the model, such as traditional land use practices and cultural preferences, which could play a potentially important role in the household's decisions with respect to forestry projects.

In the Ignacio Tavara community (Piura), the median annual income of a household is US$790, whereas in Campo Verde (Ucayali) is US$1,250. The median was chosen instead of the arithmetic mean income as it is less strongly affected by unusually high or low values (U.S. Census Bureau 2003). Agriculture activities provide 60% of the gross income of the householders in the Jose Ignacio Tavara community, while in Campo Verde the contribution is at around 76%. To assess the potential impact of carbon payments on economic activities, three scenarios were tested. In these scenarios new activities are introduced into the baseline model. With the introduction of the carbon payments: US$5, 15 and 30, the rise in total gross margin increases between 2 and 13% in Piura for 2,000 families in the area. Forestry projects aim to produce timber, which is extracted from forest during logging operations. The time of harvesting depends on the forest rotation management system, which entitlement requires long periods; thus, carbon payments could provide a source of income between the harvesting seasons.

In Campo Verde, as the project belongs to a private enterprise, villagers benefit mainly from the generation of employment, which is more than 200 wages per day and reaches 500 during the peak season. This can increase their gross margin up to 30% in some cases. Employment opportunities may additionally reduce the need of poorest households to practise migratory agriculture in forested areas.

In the Ignacio Tavara community, the median income per day is US$2.1, which is below the poverty line in Peru (US$3 per day) (INEI 2009). Among the respondents, 43% are living in extreme poverty (US$1.5), whereas 26.6% are living in poverty. In Campo Verde, the median income per day is US$3.4. In this area, 23.3% of the respondents are below extreme poverty, and 20.5% are below the poverty line. As the introduction of carbon payments appears to have a positive impact on household income, PES schemes can contribute to the reduction of poverty. For the community in Piura, with payments of US$5, extreme poverty could be decreased by 5%, and with payments of US$15 and US$30 scenarios, extreme poverty could be decreased by around 11% and 13%.

21.5.3 Impacts and Incentives for Forest-Based Projects

In this section, the potential impacts and incentives of adopting forest-based projects are described. We have found, in both areas, that forests play an important role in households' livelihoods. In the quantitative survey about their perception of PES, farmers mentioned that individual payments in cash or in kind could act as incentives for implementing new forest projects in the communities. Differences, however, were found regarding the type of payment schemes. In the Ignacio Tavara community, which has a strong social organization, 55% of the respondents viewed favourably communal payments, whereas in Campo Verde, only 28% of respondents favour communal payment for ecosystem services. Some concerns were raised on whether communal projects would be carried out according to their objectives and whether funds would be handled efficiently and distributed fairly. In Campo Verde, a few respondents had concerns about land tenure and the potential loss of their land. In Piura, where there is an inherent problem of land scarcity, associated with the need to work, obtain food and pass on land to the children, some fear that not enough land will be available for their children.

In spite of these concerns, most of the farmers mentioned that forest projects could generate employment, reduce deforestation and protect the remaining forest. They thought that financial support is the most important incentive for forest-based projects, followed by training and strengthened social organization. Thus, most of them would like to participate in the projects. Regarding enforcement and incentive measures to stop villagers from cutting trees in the forest, most of them considered that payments for ecosystem services is the best solution, followed by individual payments of penalty and physical punishment following. Physical punishment is not allowed by Peruvian law, but this kind of informal institutional arrangement is very common in some rural communities in Peru.

21.5.4 Institutional Arrangements for Carbon Sequestration Projects

Participative governance involves different stakeholders, where all parties join in a common decision-making process to achieve agreement. In Campo Verde, we did not find communal organizations working on forest-based carbon projects. However, some local organizations are working on agricultural issues, like the Campo Verde Beekeeper Association and the Organic Farming Producers. In one village, a rural patrol is working to limit illegal logging activities. According to local authorities, villagers are interested in forest-based projects, but they also mentioned that mistrust is engrained in the people due to corruption, previous failed projects and illicit crops (coca). This perception raised the question whether communal projects will be carried out according to their objectives and whether the funds would be handled efficiently and distributed fairly. The private firm through its Rural Community Relationships Office, is providing workshops to the surrounding communities on reforestation and environmental issues with excursions to the project. SFM-BAM enterprise is interested in developing a pilot communal carbon project with one community. During the interviews, high expectations about carbon payments were expressed, as well as some concerns that the enterprise might appropriate land, especially in areas surrounding the project.

In Piura, the Ignacio Tavara community was involved into project formulation and has the rights to the sequestered carbon. The stakeholders signed a formal agreement, where they commit themselves to use the revenues from CERs to cover operational costs, as well as fund social projects. As mentioned in Sect. 21.2, transaction costs of carbon projects could be reduced with communal agreements, where monitoring and enforcement are key issues for the success of these projects. We found that the representative board,[8] the board of directors[9] and the rural patrol (Ronderos) are performing successfully these activities in the area, but that more support is necessary to back up their efforts. In one village, a female rural patrol operated with great success against illegal logging activities.

These findings allow some judgments as to whether the institutional arrangements of the community could benefit the carbon sequestration project. It was found that the regulatory framework established on the basis of the traditional customary institutions provides an important framework for the implementation of a PES project, including REDD projects. For an internationally financed REDD project, monitoring activities have to be strengthened through financial support, associated with a more transparent organizational structure, where the objectives and responsibilities have to be clearly defined. In the case of the Amazon, carbon payments can be used as an incentive to reduce deforestation, which ultimately will lead to avoided greenhouse gas emissions. In some areas, villagers are interested in protecting the remaining primary forest.

21.6 Conclusions

The case studies show that carbon payments could increase adoption of these projects, although the degree of participation would depend on the price of carbon and other factors such as transaction costs and economic conditions. With low carbon prices of US$5 tCO_2e, the increment in the gross margin is low. However, with upper certificate prices of US$30, households could increase their gross margin by about 13% with the introduction of carbon payments.

In Campo Verde, no community agreements related to natural resource management were found. Nevertheless, some local organizations could be used as a starting point. In Piura, the present institutional arrangement of the community could be useful for carbon project implementation and further PES projects. It seems that they could provide a framework based on the rules and regulations of the traditional customs. They are already addressing the issue of illegal logging and are actively involved in rule enforcement. Extractive activities have declined since the establishment of rural patrols, and environmental awareness has increased, but not in all

[8] For every 50 locals, one representative is elected.

[9] The board of directors is elected with the votes of all members of the community.

villages. Thus, for a potential REDD project, the institutional framework needs to be strengthened and community participation in the conservation activities fostered. Additionally, the information flows have to be improved in order to reach more people. Negotiations can be done much more efficiently when contractual arrangements are made with the community rather than with individuals. This can substantially decrease transaction costs and has the advantage of using known institutional arrangements that can ensure familiarity for the participants, as they have trust in these established institutions.

Although Peru has ratified important agreements at the international level, at the country level, the Peruvian forestry governance lacks on an effective mechanism to implement environmental policies and appropriate control systems. Verification activities are needed to ensure that Peruvian forest resources are being used sustainably. An efficient implementation and adoption of regulations and adequate systems for managing land use and forests are necessary. Recently, in Peru negotiations have emphasized the need to establish clear national mechanisms for accounting, recording and monitoring carbon sinks and REDD projects. It is clear that this can only be achieved with a strong national system working effectively with the relevant regional counterparts.

Peru's institutional framework for PES schemes is diverse, with several overlaps. While several institutions could contribute to the implementation of these mechanisms, such as the Ministry of Environment and the Forestry Authority (national and regional), others could undermine the efforts to implement PES schemes, such as the governmental organizations promoting mining and bioenergy (oil palm) production and the government departments building roads. Therefore, it is imperative to formulate and approve measures to strengthen inter-agency coordination between the relevant authorities that oversee the use of renewable and nonrenewable natural resources and also to coordinate policies that affect natural resources.

Unrealistic expectations about the potential benefits that the carbon market could generate were found, especially among smallholders. An information campaign to address this point is necessary, using accessible formats and languages. This also has to be related to the promotion of financial mechanisms for the sale of carbon credits that are inclusive.

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Chapter 22
On-Farm Tree Planting in Ghana's High Forest Zone: The Need to Consider Carbon Payments

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22.1 Introduction

In 1994 Ghana reformed its forest policy by enacting the Forest and Wildlife Policy which encouraged economic tree planting (Forestry Commission 2006a). After that the then Minister of Lands, Forestry and Mines launched several economic tree-planting programmes in the Offinso Forest District (FD) in the Ashanti Region. These efforts were strengthened in 2001 when the government launched the National Forest Plantation Development Programme to stimulate reforestation through the establishment of forest plantations and the planting of trees on farming land (FC 2008). Section 3(3) of the Forest Plantation Development Fund Act 2000 makes provision for timber ownership rights to individuals who plant trees on farmlands (Agidee 2011). A revision of the Timber Resource Management (Amendment) Act 2002 (Act 617) (Forestry Commission 2006b) led to tree ownership being vested in

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the farmer or planter of the tree. This represented an exemption to common practice in Ghana of tree ownership being vested in the state. Since then, on-farm tree-planting initiatives have mushroomed throughout Ghana's high forest zone. In response to farmers' interest, both state actors (e.g. the Forest Services Division (FSD) of the Ghanaian Forestry Commission (FC) and the Forest Research Institute of Ghana (FORIG)) and non-state actors (e.g. non-governmental organisations (NGOs), timber and mining companies) stimulated tree planting among small farmers in off-reserve areas.[1] Many land-owning farmers in Ghana's high forest zone benefited from the policy reforms and incentives from private companies and governmental and non-governmental organisations and have adopted various agroforestry[2] models over the years.

Farmers and supporting organisations in Ghana are facing several challenges in the implementation of these on-farm tree-planting schemes, with suboptimal livelihood benefits being the result. According to Boni (2006), these challenges include extra work and costs involved in tree planting and maintenance, absence of short-term benefits due to the time gap between investment (planting and weeding) and profit (from harvesting), bureaucratic procedures to obtain loans for tree planting and land rights documentation, ambiguous legislation regarding tree ownership and insecure timber rights for tenant farmers. Given that it is a scheme that is potentially eligible for carbon payments, it is important to gain a greater insight into the livelihood implications of these challenges. This chapter aims to provide this insight by addressing the questions of (1) how on-farm tree planting contributes to rural peoples' livelihoods, (2) what are stakeholders' perceptions regarding the performance of these schemes and (3) what strategies can be followed to improve the livelihood outcomes of tree-planting schemes. The next section of this chapter provides information about the methodology and characteristics of the study area. After that we briefly discuss the theories that underpin this chapter. This is followed by a presentation and discussion of the results as regards livelihood benefits and farmers' perceptions of the scheme. The final section makes recommendations for enhancing the contribution of Ghana's on-farm tree planting schemes to rural livelihoods.

22.2 Methodology and Background to the Study Area

Fieldwork for this study was carried out between August 2009 and November 2010 in Ghana's high forest zone. Below we justify the selection and describe the main socioeconomic characteristics of the study sites and the methods used to gather the data.

[1] Ghana's forests are divided into reserved and unreserved forests, commonly denoted as on and off-reserve areas.

[2] Following Somarriba (1992, p. 240), agroforestry is defined in this chapter as a form of multiple cropping in which at least two plant species interact biologically, with at least one of them being a woody perennial and at least one plant species is managed for forage, annual or perennial crop production.

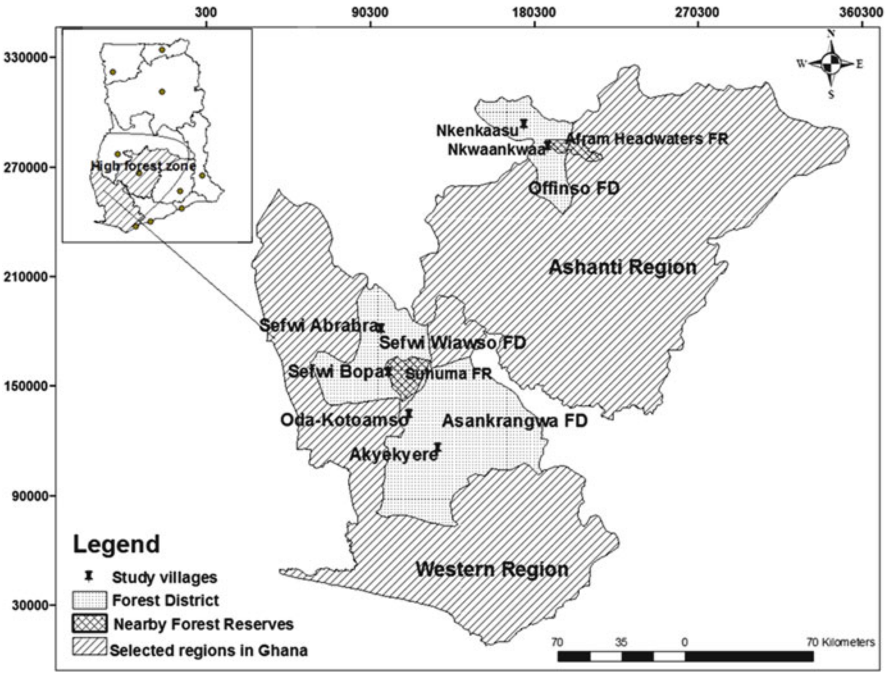

Fig. 22.1 The study sites

22.2.1 Selection of the Study Areas

Three forest districts with more than a decade of history with on-farm tree-planting programmes facilitated by both state and non-state actors were selected: the Asankrangwa and Sefwi Wiawso FDs in the Western Region and Offinso FD in the Ashanti Region (Fig. 22.1). In each forest district, two villages actively involved in on-farm tree planting were selected: Oda-Kotoamso and Akyekyere (Asankrangwa FD), Sefwi Abrabra and Sefwi Bopa (Sefwi Wiawso FD) and Nkwaankwaa and Nkenkaasu (Offinso FD) (Fig. 22.1).

The selection of the villages was based on the prevalence of on-farm tree-planting modes under different kinds of institutional support (Table 22.1). These modes encompass different modalities, ranging from planting trees in pure stands to various agroforestry systems. The latter combine the planting of exotic tree species (mainly teak (*Tectona grandis*) and cedrela (*Cedrela odorata*)) and indigenous tree species (mainly ofram (*Terminalia superba*), emire (*Terminalia ivorensis*) and African mahogany (*Khaya ivorensis*)) with perennials and food crops. Cocoa (*Theobroma cacao*) and oil palm (*Elaeis guineensis*) are the most common permanent crops, but black pepper (*Piper nigrum*), cola (*Cola nitida*) and orange (*Citrus sinensis*) are also interplanted with trees. The food crops in question are mainly plantain, cassava, vegetables, maize, cocoyam and yam.

Table 22.1 Four on-farm tree-planting modes identified in Ghana's high forest zone

Mode 1: Timber company-supported on-farm tree planting
Farmers in the Samartex concession area are supported by the Samartex Agroforestry Unit (SAU), which is part of the company's Forest Development Division (FDD), in the planting of timber trees on farmlands. Stakeholders (chiefs, farmers, company) jointly agree on the tree-planting modalities.
The SAU negotiates with chiefs to release land to farmers interested in tree planting and mediates in obtaining titles of the tree farms.
In the case of farmers who use individual or family/clan lands for tree planting, their share in the benefits is based on 100% of crops (all types) and 100% of the tree benefits.
In the case of farmers who use the chief's land, benefit-sharing regarding timber trees is based on 33% for the chief/landlord and 67% for the farmer and 100% of food crops for the farmer.
In the case of planting timber trees in cocoa farms under a sharecropping arrangement, benefit-sharing is based on the *Abunu* sharing system (50% for the landlord and 50% for the tenant for both timber and crop benefits).
The supporting timber company has the first right to buy the planted trees at prevailing market prices.
Identified in the Asankrangwa FD.
Mode 2: NGO-supported on-farm tree planting
The NGO facilitates on-farm timber tree planting in selected communities as part of its project mandates.
Farmers use individual or family/clan lands for tree planting and are entitled to a 100% share of the crops (all types) and a 100% share of the tree benefits.
Farmers find their own market for planted timber trees.
Identified in the Sefwi Wiawso FD.
Mode 3: State government-supported on-farm tree-planting initiative
The government employs workers to plant timber trees in both on and off-reserve areas. Facilitated by the Forest Services Division of the Forestry Commission and the District Assemblies. Launched in January 2010. The government (investor) uses chief's land in off-reserve areas, with benefit-sharing being based on 33% for the chief/landlord and 67% for the investor for tree benefits and 100% of the food crops for the workers.
Workers are paid for tree planting and maintenance, but have no benefit in terms of tree revenues.
Identified in the whole high forest zone.
Mode 4: Farmer initiative with little government support
Farmers plant trees on farms on their own initiative, incentivised by policy reforms and the importance of economic trees and with a little support from the government (FSD/FC).
Farmers use individual or family/clan lands for tree planting and therefore receive 100% of the crop benefits (all types) and 100% of the tree benefits.
Farmers find their own market for planted timber trees.
Identified in the Offinso FD.

In Asankrangwa FD farmers were involved in a private reforestation scheme, i.e. a company-community partnership with Samartex Timber & Plywood Co. Ltd. (Samartex in the rest of this chapter). This firm – one of the largest timber companies in Ghana, which became Forest Stewardship Council (FSC) certified in 2008[3] – supports farmers

[3] URL: info.FSC.org, Retrieved November 24, 2011.

in tree planting as part of its corporate social responsibility policy. In order to realise its sustainability and social responsibility aims, Samartex established the Forest Development Division (FDD) with the aim being to (1) collaborate with communities and farmers to develop agroforestry systems and plantations, (2) establish plantations on degraded lands and (3) promote the development of non-timber forest products (NTFPs) such as *Thaumatococcus daniellii*, which is used as a natural sweetener, and honey. To realise the first aim, a pilot project known as the Oda-Kotoamso Community Agroforestry Project (OCAP) was set up in 1997[4] and later expanded to other communities in the firm's concession area under a public-private partnership (PPP) with the German Agency for Technical Cooperation (GTZ) known as the GTZ/Samartex PPP (Suglo 2009). Within this context, efforts were made to grant property titles to the tree-planting farmers (with 212 farm plots mapped and processed for registration by November 2010), to inform farmers and traditional authorities about land rights and rules that regulate tree planting in off-reserve areas and to explore opportunities for farmers to engage in carbon credit schemes.

The villages of Sefwi Abrabra and Sefwi Bopa in Sefwi Wiawso FD were selected because they are representative of the 59 villages in Sefwi Wiawso FD that were supported by Ricerca e Cooperazione, an Italian NGO which stimulated tree planting on farmlands under its Forest Resource Creation Project from 2000 to 2004. The general aims of this NGO – established in 1985 and active in Ghana since 1987 – centre on safeguarding biodiversity and the cultural heritage of indigenous cultures and on promoting fundamental human rights and good governance. During its presence in the study area, the NGO promoted tree planting on crop land, in degraded cocoa plantations and in oil palm plantations, with a view to reducing pressure on natural forests and improving soil fertility by planting nitrogen-fixing tree species (Da Re 2005). It did so by organising farmers into tree-grower associations, providing seedlings and promoting agroforestry by providing technical advice and equipment. Moreover, it promoted alternative livelihoods, like black pepper cultivation, beekeeping, vegetable growing, grasscutter rearing and snail farming (Ibid., pp. 24–25). Due to low returns resulting from a lack of institutional capacity and policy support, suitable credit schemes, markets, and skills and sustainable interest among beneficiaries (who preferred to invest in perennials like cocoa and oil palm), the project was discontinued in 2004 with no exit plan that could guarantee follow-up by the FSD (Da Re 2005, p. 25; Mr Jones, former RC project officer, personal communication 2011).

Offinso FD was selected because there are ten villages in this forest district where a good number of farmers have adopted on-farm tree planting through their own efforts, with some support from organisations like the FSD. Farmers organised themselves into the Offinso Teak Growers Association (OTGA), the leadership of which maintains good contacts with the FSD. Via the OTGA leadership, occasional support from the FSD was obtained in the form of training and mediating in tree

[4] URL: www.samartex.com.gh, Retrieved November 24, 2011.

seedling supply. Overall, however, tree-planting farmers in this forest district rely on their own or on hired experts to survey and document their land and to find a market for their mature timber.

The fourth, government-supported, tree-planting mode in Table 22.1 has not been included due to it being too recent an initiative (initiated in January 2010) to enable any meaningful data collection.

22.2.2 Socioeconomic Characteristics of the Study Areas

Agriculture is the major economic activity in all study sites, employing 70–85% of the people who are mostly peasant farmers. Traditionally engaged in slash-and-burn cultivation (Quansah et al. 2001), increasing scarcity of farming land forced farmers to engage in sedentary farming (Da Re 2005). In the Asankrangwa and Sefwi Wiawso FDs, cocoa and oil palm are the major cash crops, although coffee is also grown. In the much dryer Offinso FD, prospects for cocoa farming are less favourable. Oil palm is all-weather resistant, but the region has a problem with the provision of good seeds.[5] Consequently, farmers focus mainly on vegetables (tomatoes, peppers, garden eggs and okra) and, to a lesser extent, on cashew and timber trees as their main cash crops. Particularly in this region, timber has become an interesting option due to declining cocoa yields. The differences in ecological circumstances and agricultural opportunities mean that in the first two areas, timber trees are mainly interplanted with cocoa trees, whereas they are planted in pure stands in Offinso FD, with a preference for fire-resistant teak (*Tectona grandis*). In all areas, the major staple crops are cassava, cocoyam and plantain. In Offinso FD farmers also cultivate maize, vegetables and yam.

The villages in Asankrangwa and Offinso FD are close to regional market centres. Asankrangwa town is a large market centre located 4–5 km from the two villages. Nkenkaasu is one of the major market centres in Offinso FD; Nkwaankwaa village is about 5 km from Abofour town which has a vibrant weekly market. In both districts, there are ample opportunities for market-oriented production. This is less so for the more isolated study villages in Sefwi Wiawso FD, which have a poor road connection with Sefwi Wiawso town that has a large market. In these villages, trading mainly takes place through middlemen who come to the villages during the peak of the harvesting season.

22.2.3 Research Methods

Data was obtained through a household survey involving 106 on-farm tree-planting smallholders from the six villages (Table 22.2), open-ended interviews with key informants and some validations through group discussions in the villages and additional

[5] URL: http://offinso.ghanadistricts.gov.gh, Retrieved November 24, 2011.

Table 22.2 Overview and characteristics of the study sites

Study village	No. of resp.	Stool land owners	Forest district	Administrative district	Region	Eco-zone	Prevailing tree-planting scheme
Oda-Kotoamso	16	Asankrangwa	Asankrangwa	Amanfi West	Western	Wet evergreen to moist evergreen forest	Company-community partnership
Akyekyere	14	Akyekyere					
Sefwi Abrabra	32	Sefwi Asanteman	Sefwi Wiawso	Sefwi Wiawso/ Akontombra	Western	Moist evergreen to moist semi-deciduous forest	NGO-facilitated tree-planting and agroforestry scheme
Sefwi Bopa	21						
Nkwankwaa	9	Offinso	Offinso	Offinso	Ashanti	Moist semi-deciduous, semi-evergreen forest	Farmer's own initiative
Nkenkaasu	14						

key informant interviews. Key informants included chiefs and a queen mother, project officers affiliated to supporting organisations, FSD officers and leaders of tree-planting associations and steering committees. In each village, respondents were randomly selected from the group of tree-planting smallholders who were identified with the help of farmer leaders. Of the respondents, 61% were males and 39% females, which corresponds with the overall gender ratio in on-farm tree planting. Three village level focus group discussions were held with farmers, during which Tool 4 of the PROFOR 'Poverty-Forests Linkages Toolkit'[6] was employed. This tool is a qualitative method employed for one group of ten males and one group of ten females per village, through which the participants list their income-generating economic activities and clarify the proportion of income generated by each of them by assigning 20 stones (representing 'money') to the listed activities. This was done twice for each group, for both cash and non-cash income, respectively. Finally, in-depth interviews were held with six smallholders from the three forest districts with a view to obtaining detailed data on planting and maintenance costs and revenues from the scheme.

22.3 Theoretical Outlook

This study uses the sustainable livelihood approach based on Carney (1998) and Scoones (1998, 2009) to analyse the livelihood effects of on-farm tree planting. Attributes considered in this study include assets (natural, human, financial, physical and social capitals) formed through the on-farm tree-planting schemes. The sustainable livelihood approach also considers stresses and shocks that can affect planted trees and crops (in this case, drought, fire and timber theft), as well as fluctuations in markets (prices of crops and timber). For an on-farm tree-planting scheme to act as a sustainable livelihood, it should, among other things, include timber species like teak that can withstand drought and fire outbreaks. The tree-planting scheme should also incorporate all kinds of marketable annual and permanent crops in an agroforestry setting, in order to generate income during the period between tree planting and harvesting.

In terms of livelihood potential, we draw on the distinction made by Sunderlin et al. (2005) between poverty mitigation and poverty elimination. In the first case (poverty avoidance or mitigation), forest resources serve as 'safety nets' or 'gap fillers', whereas in the case of poverty elimination, forest resources help lift the household out of poverty by functioning as a source of permanent increase in income, assets, services, civil and political rights, voice and the rule of law. In this

[6] The Program on Forests (PROFOR) is a multi-donor trust fund based at the World Bank with the aim being 'to support in-depth analysis, innovative processes and knowledge-sharing and dialogue, in the belief that sound forest policy can lead to better outcomes on issues ranging from livelihoods and financing, to illegal logging, biodiversity and climate change' (URL: http://www.profor.info/profor/content/our-mission, Retrieved December 20, 2011).

chapter, we emphasise the prospects for pulling people out of poverty through the creation of a high-value forest resource by economic tree planting on farmlands.

Several studies have been carried out that have shed light on the actual and potential livelihood effects of reforestation and agroforestry schemes. For instance, Smith and Scherr (2003) highlight the cash and non-cash income benefits from food, fuel and construction material; inputs for farming (e.g. green manure and fodder); and environmental services (e.g. windbreaks, erosion prevention, soil fertility enhancement and soil recuperation). In some cases, smallholder tree planting even allows farmers to accumulate assets that can be invested in farmland and children's education or be used to pay off debts (Saxena 1997, cited in Smith and Scherr 2003). Chambers et al. (1993) stress the role of trees in dealing with contingencies, either through direct use, sale of timber for cash or as a source of savings and security. At the same time, several livelihood risks inherent in tree planting have been noted, such as a reduction of the land available for crop production (Smith and Scherr 2003), theft (Kusters et al. 2008) and fire (Insaidoo et al. forthcoming). More recent literature stresses the notion that the economic feasibility of agroforestry and small-scale plantations, as well as their livelihood benefits, can be increased by their engagement in the carbon market (e.g. Swallow et al. 2006; Schroth and McNeely 2011). This issue will be addressed in more detail in Sect. 22.5.

22.4 Contribution of On-Farm Tree Planting to Participants' Livelihoods

This section analyses the contribution made by on-farm tree planting to the five livelihood assets of human, social, natural, financial and physical capital.

22.4.1 Human Capital

Figure 22.2 presents the competencies respondents claimed to have acquired through their contacts with partner organisations that supported them in tree planting. The primary skill acquired is in planting and agroforestry techniques, followed by tree nursery establishment techniques. Remarkably, this score is lower among the company-supported tree growers in Asankrangwa FD than among the farmers in Offinso FD who receive minimal support from the FSD. This can be explained by the strong leadership of the Offinso Teak Growers Association to which the tree-planting farmers are affiliated and the close contacts that these leaders maintain with the FSD.

Engaging in tree planting also involves attending meetings and workshops in which information and training are given that add to farmer's skills, including in Offinso FD. Training in alternative livelihood ventures applies only to the externally supported modes, mentioned more frequently in relative terms by the company-supported tree growers in Asankrangwa FD (47%) than the NGO-supported ones

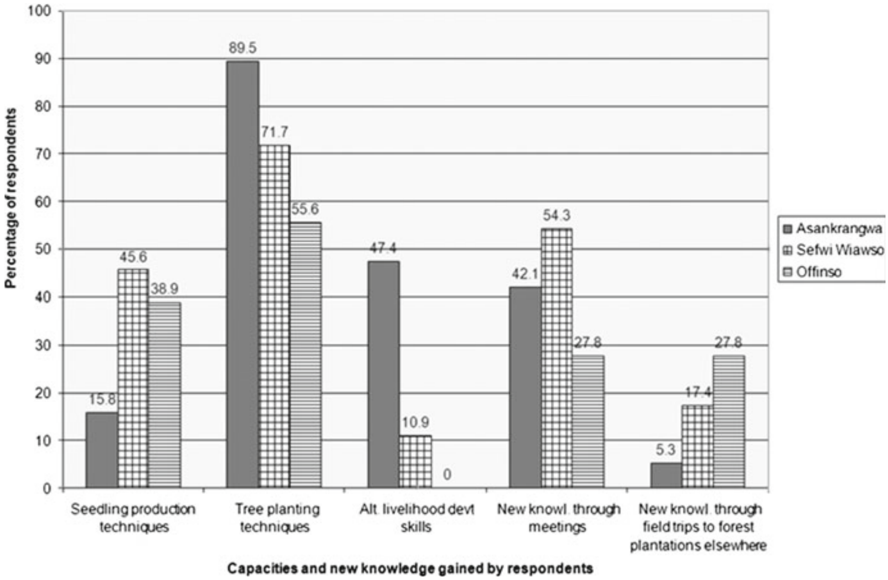

Fig. 22.2 Competencies (human capital) acquired in the tree-planting project

in Sefwi Wiawso (11%). Supporting organisations, including the OTGA in Offinso FD, also organise field trips to forest plantations elsewhere, and this was also referred to as a source of acquired skills (Fig. 22.2).

Survey results indicate that training or guidance in tree planting and/or agroforestry techniques has resulted in increased crop yields, income and fodder for animals, as well as improved skills in farming. In Asankrangwa, Sefwi Wiawso and Offinso FDs, 71, 21 and 29% of respondents, respectively, indicate higher crop yields from both agroforestry and farm plots as a result of guidance in agroforestry and tree-planting skills. Respondents in Offinso FD (86%), who are actively involved in small ruminants (sheep and goats) rearing, indicate that guidance in agroforestry techniques from the FSD via their leaders has helped them to improve fodder production for their animals.

22.4.2 Social Capital

All three tree-planting modes helped create social capital in the form of producer groups and/or tree-grower associations.

Tree farmers in Oda-Kotoamso (Asankrangwa FD) were organised into a group that operated under the OCAP. Until 2004, when financial support from Samartex was ended, the OCAP's steering committee consisted of a nine-member executive body (all participating farmers), the Samartex representative who acted as the manager and a representative of the chief. The local steering committee's tasks included (1) the administration of annual requests for seedlings on behalf of participating farmers (old and new), (2) mediating in encroachment and boundary disputes, (3) monitoring illegal felling of timber trees from nearby forests and farmlands, (4) patrolling to prevent fire outbreaks during the dry season (November–March), (5) distributing beehives to members who wanted to go into beekeeping and (6) monitoring tree planting and alternative livelihood venture activities. The local steering committee still exists and continues to function despite the withdrawal of Samartex's financial support, with a focus on processing annual requests for tree seedlings from the timber company and on protecting the trees and forests in the area against fire and illegal felling.

Tree farmers in Akyekyere (Asankrangwa FD) under the company-community partnership organised themselves into a group under the leadership of the village chief (a tree farmer himself) and his assistant. The two leaders mediate between Samartex and the tree farmers in the village in requests for tree seedlings and beehives (which they have to pay for since Samartex ended its financial support for the agroforestry project in 2004), free extension services and other needs. This group has no appointed or elected executives or bye-laws.

Under the NGO-supported tree-planting mode in Sefwi Wiawso FD, the farmers were organised into a group during the period that the NGO (Ricerca e Cooperazione) actively facilitated tree planting. The group's executive body coordinated project management with the NGO in order to ensure a smooth implementation of tree-planting activities by participating farmers. The executives mobilised members for meetings, training workshops and other group-based activities organised by the project. The producer group dissolved when the NGO ended its activities in the region, and the few farmers who continued to plant trees after the exit of the project did so individually.

In Offinso FD, the majority (73%) of the respondents are members of the Offinso Teak Growers Association (OTGA) that is composed of on-farm tree farmers from ten villages in Offinso FD. The OTGA liaises with the district FSD office for the provision of technical advice and inputs (e.g. tree seedlings) to its members. The main reason (given by 32% of the respondents) for individual tree farmers to join the association is to expand their access to external support. One respondent mentioned participation in decision-making and providing a strong voice for the welfare of tree growers as a reason to join. Although the association's requests for support from the FSD do not always generate the expected results, it plays a role in promoting tree planting, as a result of which farmers continue to plant trees annually and the area planted with trees is slowly but gradually expanding.

22.4.3 Natural Capital

In focus group discussions, respondents in the Asankrangwa and Sefwi Wiawso FDs indicated that all participating farmers receive timber tree seedlings from their supporting organisations. This is only 17% of the interviewed self-organised

Table 22.3 Average size of land planted per tree-planting respondent in acres (and ha)

Study village	Asankrangwa FD	Sefwi Wiawso FD	Offinso FD	Total
Sefwi Abrabra		4.25 (1.7)		
Sefwi Bopa		4.02 (1.61)		
Oda-Kotoamso	5.27 (2.11)			
Akyekyere	4.99 (2.0)			
Nkenkaasu			13.3 (5.32)	
Nkwaankwaa			4.93 (1.97)	
Average size planted	5.16 (2.06)	4.12 (1.65)	10.64 (4.26)	6.17 (2.47)

Figures in brackets is size in hectares (ha)

farmers in Offinso FD. Most tree growers in Offinso FD obtained the seedlings from individual or group nurseries, some of which were established with support from the FSD.

Respondents' estimates of the amount of land planted with trees are presented in Table 22.3 and indicate that the average area planted since the time of entry amounts to 6.17 acres (2.47 ha) per farmer ($n = 82$). Virtually all respondents planted trees on their own land. Only one planted trees on the land of her spouse and one on the land made available by the chief. None of the people in question were involved in sharecropping arrangements which are very common in Ghana (Amanor 2001). Remarkably, the average area planted by tree growers in Offinso FD is more than twice the average of farmers in the other forest districts, despite the fact that they could not count on continuous outside support from a company or NGO. This can be explained by their strong motivation to plant teak rather than perennial crops like cocoa, as the fire resistance of teak makes it a less risky investment in the dry deciduous environment that characterises Offinso FD.

Information from Samartex (annual reports and personal communication with staff) indicates that, by November 2010, a total of 1,820 ha of off-reserve land in the Asankrangwa FD had been planted with economic timber trees. Of this total, 1,120 ha, involving 226 participants from 12 communities/villages in Samartex's operational area, were planted under specific agroforestry projects initiated by the company, such as the OCAP and the GTZ/Samartex PPP in Akyekyere. The other 700 ha were planted by about 700 individual farmers from 27 communities who had benefited from extension services provided by the Samartex project staff in collaboration with local staff of the FSD and the Ministry of Food and Agriculture (MOFA).

In Sefwi Wiawso FD, no records were available at the NGO or FSD on the total size of farmland planted with trees. However, records from the Ricerca e Cooperazione project office indicate that 466 farmers from 59 villages in this forest district had benefitted from the NGO's support for tree planting. Based on the average size planted with trees per farmer (1.65 ha) that came out of the survey, the total size of tree farms in Sefwi Wiawso FD can be estimated at 746 ha (466×1.65).

Records on the number of on-farm tree planters and area planted could not be obtained from the FSD office in Offinso FD either. According to an OTGA leader, who is well versed with people engaged in tree planting in the area, about 470 individuals from ten communities in Offinso FD are involved in this tree-growers association. Using the average size of 4.26 ha that respondents said they had planted with trees, we estimate the total size of tree farms in Offinso FD to be 2,002 ha (470×4.26).

22.4.4 Financial Capital

Financial capital denotes the availability of cash or equivalent that enables people to adopt different livelihood strategies. It is the asset that tends to be the least available to the poor (DFID 1999). Since it is difficult to collect accurate data on incomes in rural settings where farmers do not keep records, and because non-cash income is a large share of total income, the contribution of the on-farm tree-planting scheme to peoples' cash and non-cash incomes was assessed in relative terms using Tool 4 of the Program on Forests (PROFOR) 'Poverty-Forests Linkages Toolkit' (Shepherd and Blockhus 2008; see Sect. 22.4.1). In Asankrangwa and Sefwi Wiawso FDs, where trees are generally interplanted with cocoa or other perennials such as oil palm or orange, income from tree farms refers to food crops interplanted with trees during the first years of agroforestry establishment (until canopy closure) and to income from the perennials once these start to produce (after 5 years for cocoa). In Offinso FD, where timber trees are planted in pure stands, data refers to the first 3 years until canopy closure when food crops can still be grown between the trees. In pure timber stands, food crops can no longer be cultivated once the canopy closes and will not generate income until the timber can be harvested.

The results as regards the proportional contribution of each activity to cash and non-cash incomes for the three study sites are presented in Figs. 22.3, 22.4, and 22.5, for men (a) and women (b), respectively. These results indicate that the participants derive their cash and non-cash incomes from land exclusively used for farming, tree farms, (off-reserve) fallow land, the forest reserve, wage earnings (mainly from day labour on other people's farms and plantations, locally referred to as 'by-day' and informal labour like masonry and carpentry), petty trading and remittances.

Agriculture, both from plots exclusively used for farming and tree farms, contributes the lion's share to people's cash and non-cash income (75–80% for men and 80–90% for women) across the three study forest districts. Income from tree farms is relatively more important to women than to men in Offinso FD, as it is based mainly on food crops, which are mainly the domain of female farmers. In the other study sites, the trees are mainly integrated into cocoa farms, which are relatively more important to men.

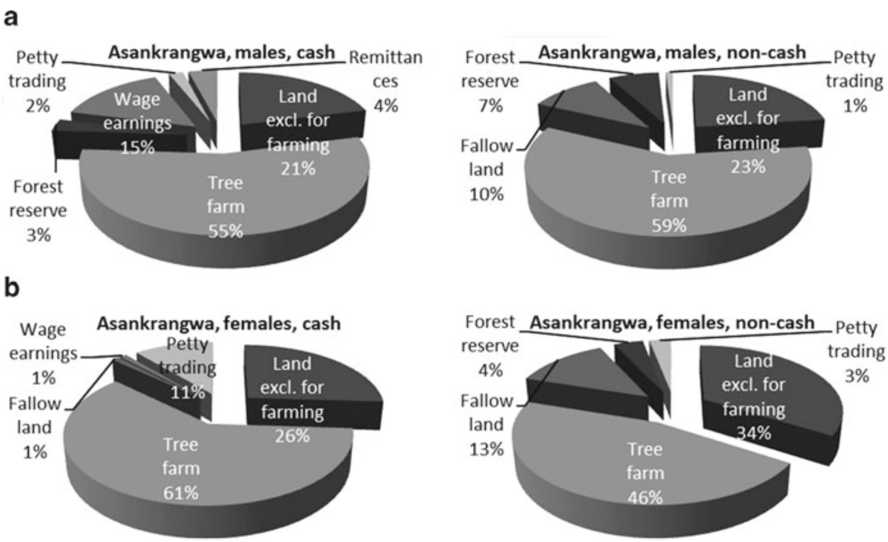

Fig. 22.3 Cash and non-cash income among tree-planting farmers in Asankrangwa FD (**a**) Males (**b**) Females

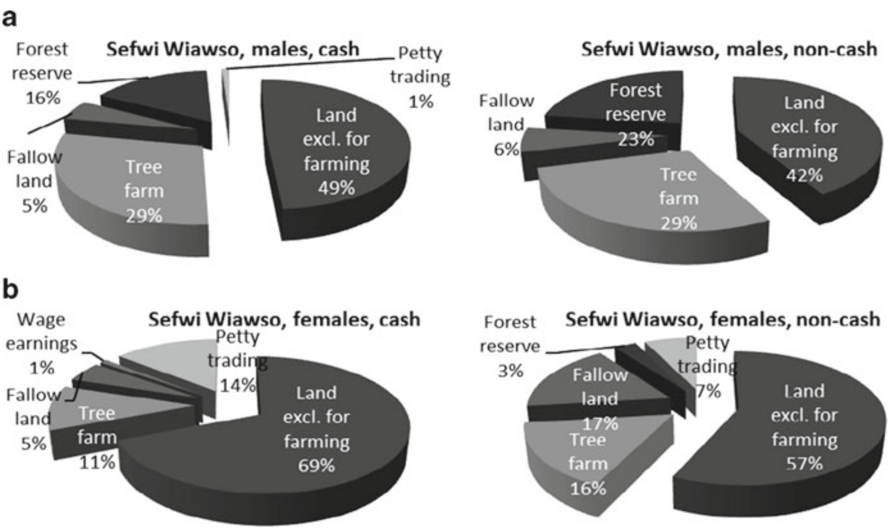

Fig. 22.4 Cash and non-cash income among tree-planting farmers in Sefwi Wiawso FD (**a**) Males (**b**) Females

As long as food crops can be harvested from them, the tree farms generate between 29 and 55% of cash income for males and between 11 and 61% of cash income for females, with the relative contribution for both sexes being lowest in Sefwi Wiawso FD and highest in Asankrangwa FD. The contribution to non-cash

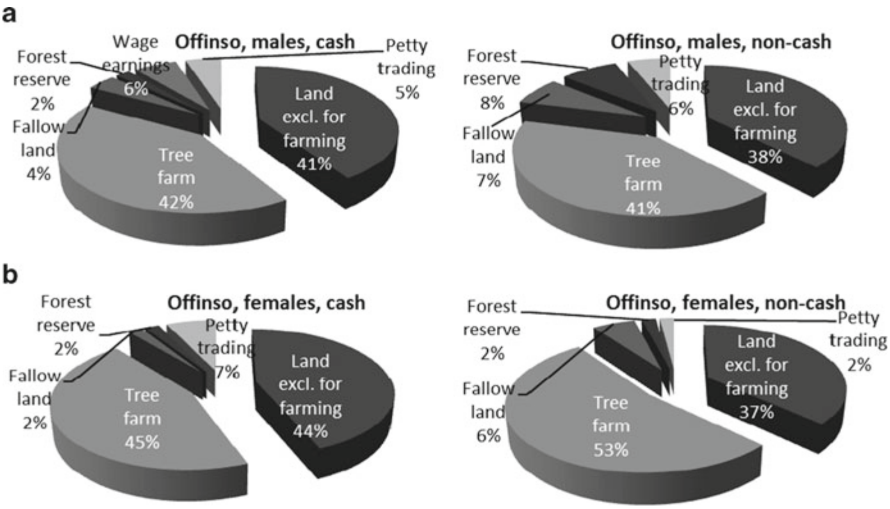

Fig. 22.5 Cash and non-cash income among tree-planting farmers in Offinso FD (**a**) Males (**b**) Femalesing farmers in Offinso FD

income is between 29 and 59% for males and between 16 and 53% for females, with the lowest relative contribution noted in Sefwi Wiawso FD and the highest in Asankrangwa FD for males and Offinso FD for females. Tree farms play a less important role as a source of cash and non-cash income in Sefwi Wiawso FD than in the other forest districts. This corresponds to the fact that tree planting is regarded as a low priority among farmers in this district (where cocoa is the priority crop) and has not been additionally stimulated since the departure of the NGO in 2004. Remarkably, the tree farm plots in this forest district are relatively less important for women's cash incomes (11%) than for those of men (29%), whereas the opposite is true for the relative contribution of land used exclusively for farming (which contributes 49% of the cash income of males and 69% of the cash incomes of females). This difference, which is not evident in the other forest districts, can be attributed to the fact that men have continued to invest more than women in tree farms since the departure of the NGO.

In Asankrangwa FD, wage earnings contribute a larger share (15%) to men's cash income than in the other forest districts (nil in Sefwi Wiawso FD and 6% in Offinso FD). This can be explained by the employment created by the timber company that operates in this area. Wage earnings do not play a role in women's cash incomes, but they do engage more in petty trading instead, which contributes 7–14% to their cash incomes across the three study areas, as well as a modest share (ranging from 2 to 7%) to their non-cash incomes.

The forest reserve contributes a larger share to male cash (16%) and non-cash income (23%) in Sefwi Wiawso FD than in the other study sites, where the forest reserve contributes 3%/7% (Asankrangwa FD) and 2%/8% (Offinso FD) to the cash and non-cash incomes of males respectively. This can be explained by the fact that the villages in this study site are located closer to the forest reserve and that the reserve is relatively richer in biodiversity and therefore provides opportunities for hunting bushmeat and NTFP collection. The forest reserves hardly contribute to female income as women consider entering the reserve to be dangerous and the collection of NTFPs from the reserve to be physically demanding. Instead, they collect NTFPs from fallow lands, notably in Sefwi Wiawso FD, but also in Asankrangwa FD, where fallow lands contribute in particular to their non-cash income (17 and 13%, respectively).

22.4.5 Physical Capital

Physical capital comprises the basic infrastructure (e.g. roads, affordable transport, secure shelter and buildings, adequate water supply) and producer goods (e.g. tools) needed to support livelihoods (Scoones 2009). In Asankrangwa FD, the timber company occasionally reconstructed the feeder road network leading to the tree farm plots under the agroforestry project. In Sefwi Wiawso FD, the NGO provided a number of water storage tanks in some of the communities, as a way of ensuring adequate and continuous water supply in the agroforestry plots. No such infrastructural support was available for tree growers in Offinso FD. Physical capital also took the form of houses which respondents had built with revenues from the tree farms. However, this was only possible for a minority, i.e. 10, 7 and 15% of the respondents from Asankrangwa, Sefwi Wiawso and Offinso FDs respectively.

22.4.6 Respondents' Perceptions of Livelihood Outcomes from On-Farm Tree Planting

Respondents perceive on-farm tree planting as having both positive and negative effects on their livelihoods (Fig. 22.6). Positive effects include employment creation, increased incomes, increased food production and enabling farmers to build houses and educating their children to higher levels. Positive environmental effects such as trees serving as windbreaks and stakes for crops and improved soil fertility, which indirectly impact positively on livelihoods, were also mentioned. Negative effects include the high costs (which are perceived as draining money from the farmer), reduced crop yields due to shade effects and an increased workload. An overview of cost categories involved in establishing tree farms is given in Appendix 1.[7]

Respondents from Asankrangwa FD supported by Samartex appear to be most satisfied with the scheme. Substantial numbers mentioned increased food

[7] We considered the financial data that we collected on costs to be unreliable due to the failure by farmers to keep books and provide financial transparency and therefore decided to provide an overview of items and labour time only.

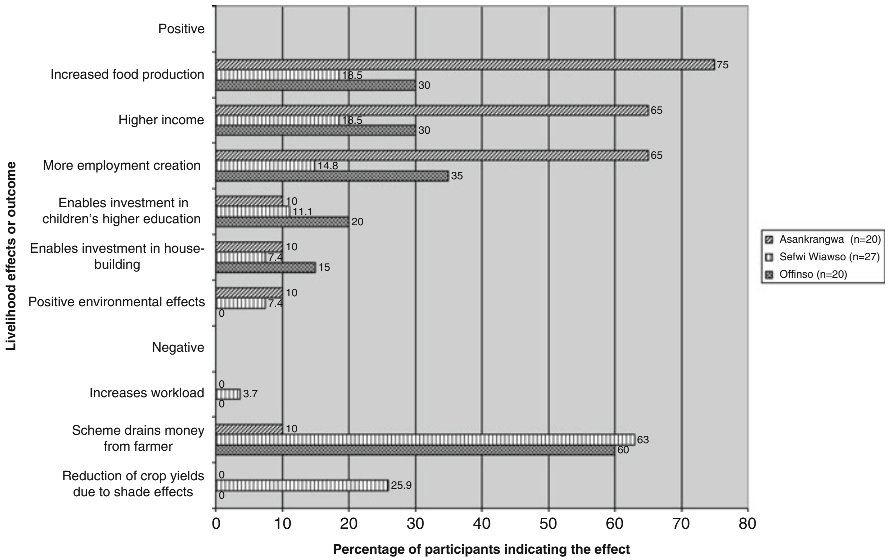

Fig. 22.6 Effects of on-farm tree planting on respondents' livelihoods

production (75%), increased income (65%) and employment creation (65%) as positive effects, whereas only 10% mentioned a negative effect in the form of the scheme draining money from them. Only a minority of 10% indicated that they were able to accumulate assets in the form of investments in children's education or house building.

The NGO-supported tree-planting scheme in Sefwi Wiawso FD generated the least positive perceptions of livelihood outcomes. Two of the three negative effects (increased workload and reduced crop yields due to shading) were only mentioned in this forest district (4 and 26% of respondents respectively), whereas no less than 63% stated that the scheme costs them money. Only a minority indicated positive effects, such as increased food production (19%), increased income (19%) and employment creation (15%), while even fewer tree growers indicated that they were able to accumulate assets in the form of investments in their children's education (11%) or house building (7%).

The perceptions of livelihood effects among respondents from Offinso FD indicate that about one third of them see positive effects in terms of increased food production (30%), increased income (30%) and employment creation (35%), whereas some were able to accumulate assets in the form of children's education (20%) and houses (10%). However, on the negative side, a majority (60%) indicated that the system drains them of money.

Only a few respondents from Asankrangwa and Sefwi Wiawso FDs (10 and 7%, respectively) mentioned positive environmental effects, such as trees serving as windbreaks and stakes for crops like yam and black pepper, improving soil fertility and providing shade for cocoa trees.

Despite the fact that there are mixed feelings about current livelihood outcomes in two of the three forest districts, the majority (86%) of the respondents consider on-farm tree planting as a potentially important source of livelihood for the future, with the main reason being that it serves as a source of future income and creates a legacy for their children.

At the same time, the majority (80%) of respondents think that on-farm tree planting is not a reliable source of livelihood as there is still a lot of uncertainty about future income from trees (mentioned by 48%). They regard it as a safety net rather than a stable source of livelihood because they do not depend on it for their daily expenses (mentioned by 26%). Other reasons to be pessimistic about the stability of on-farm tree planting are based on perceived challenges, including (1) the lack of financial means for tree farm maintenance, especially after the third year when annual food crops can no longer be grown between the trees; (2) high labour and/or maintenance costs; (3) the time lapse between tree planting and harvesting with no immediate or regular benefits from trees in the meantime; and (4) a lack of support and incentives for farmers from the government. Specifically in Sefwi Wiawso FD, where farmers plant timber trees in cocoa farms, there is a problem of shade effects of trees (notably cedrela) which result in reduced yields of intercrops. Field verification revealed that this was due to an excessively dense stocked spacing design. A problem mentioned only in Offinso FD – the only region where farmers were already harvesting teak – was the bureaucracy associated with obtaining harvesting rights and conveyance permits and uncertainty about prices for timber from off-reserve areas.

22.5 Discussion

Current rates of deforestation and forest degradation and the associated loss of goods and environmental services underline the need for new approaches to reforestation that address rural poverty (Lamb et al. 2005). Ghana implemented several of these approaches by launching its Forest Plantation Development Programme in 2001. Combined with laws which acknowledge farmers' rights to trees, this particularly encouraged smallholders with secure rights to their lands (Owubah et al. 2001) to use their land for tree farming.

The presence and encouragement of supporting organisations – public (the FSD) or private (timber company, NGO) – appeared to be decisive for the livelihood outcomes of tree-planting schemes. Even in the case of self-organisation in a tree grower's organisation, outside support in the form of occasional supply of tree seedlings and regular technical advice from the FSD was crucial. In terms of the relative importance of income from the tree farms (Figs. 22.3, 22.4, and 22.5) and farmer's perceptions (Fig. 22.6), the timber company performed best. The explanation is related to the company's interest to make the scheme a success. Due to it having a stake in securing its source of raw material for the future, the company put all the necessary resources in place to ensure the success of the scheme. It established a separate agroforestry unit and contracted professional foresters to supervise the agroforestry programme and cooperated effectively with government agencies like the MOFA and FSD. Officers in the Samartex agroforestry project paid regular visits to the villages involved, ensuring that the right technical advice was given. By contrast, the NGO had a broader scope with no specific interest or expertise in tree planting. Project officers were not necessarily professional foresters, nor did they have the financial means to visit the study villages on a regular basis. Furthermore, they misjudged farmer's interest in tree planting in a region where cocoa and oil palm offer better perspectives. As far as government agencies are concerned, respondents attributed a lot of the challenges that they faced to the passive nature of state involvement and the lack of technical advice and incentives from the state.

Literature identifies secure land and tree tenure as another major factor in adopting on-farm tree planting, not only in Africa (e.g. Fortmann 1985; Brasselle et al. 2002) but also elsewhere (e.g. Dewees and Saxena 1997; Potter and Lee 1998). Moreover, in Ghana farmers with secure rights to land are more likely to plant trees on farmlands (Owubah et al. 2001). This is confirmed in this study, where virtually all respondents were owners of the land on which they planted trees, and none of the land where trees were planted was subject to a sharecropping arrangement. Samartex acknowledged the importance of secure tenure and deliberately facilitated the registration of land and tree rights for participating farmers in Asankrangwa FD.

On-farm tree planting has potential to become an important element of farm livelihoods (Dewees and Saxena 1997) by creating high-value tree assets for the future. However, several challenges adversely affect participants' livelihoods and explain the mixed feelings among farmers about current livelihood outcomes. In the first place, it should be realised that on-farm timber tree planting is not based on farmers' traditional farming systems, which in Ghana's high forest zone is mainly reliant on a combination of food crop farming (with cassava, maize and plantain as the main crops) and cocoa farming (Chamberlin 2008). Tree planting has primarily been driven by a government interest in addressing timber deficits and rural poverty and has been implemented from above. Since colonial rule vested land ownership (and hence the right to a share in royalties from timber concessions) in customary authorities (the stool) and postcolonial legislation (Act 1962 (124)) vested custody over trees in the state (Amanor 1999; Boni 2006), trees have not played a role in farmers' livelihoods as no exploitation rights or benefit-sharing arrangements for farmers were in place. Against this background, it comes as no surprise that farmer's experience, skills and knowledge of input, output and credit markets related to tree planting are limited (Chamberlin 2008). This partly explains why tree-planting schemes are extremely dependent on outside support.

Second, in view of limited benefits thus far, farmer's motivation for tree farm maintenance is restricted. Farmers in this study started tree planting mainly because of high expectations of future income (93%), with having wood for house building (19%) and creating a legacy for their children (18%) as secondary motivations. However, a combination of high costs for tree farm establishment and maintenance, the long gestation period of trees and the lack of funds for tree farm maintenance once food cropping between the trees is no longer possible has discouraged farmers and hampered the continuity and success of the scheme. The lack of income between canopy closure and timber harvesting is a problem in most reforestation schemes that focus on trees in pure stands (Boni 2006).

Third, the few farmers (in Offinso FD) who engaged in tree farming long enough to harvest some trees faced several bureaucratic hurdles. Procedures for obtaining harvesting and conveyance permits were lengthy, leading to low prices being paid by timber companies.

Among the options suggested in literature to address the financial challenges of tree-planting schemes is the advanced or gradual purchase of timber from farmers (Boni 2006; Montagnini et al. 2005). More concretely, Boni (2006, p. 6) suggests to facilitate the institution of a timber 'co-ownership agreement policy', which allows the timber company (buyer) to purchase timber gradually by paying a small yearly maintenance support fee (e.g. 30–50 dollar cents per tree per year) to the farmer (planter). In exchange for the yearly maintenance fee, the buyer acquires joint ownership of the timber and the right to purchase the remaining half at harvest at the prevailing market price. Alternatively, the maintenance fee can be considered a loan scheme to the farmer, to be paid at harvest with interest. If such a scheme is considered, it is important to put an insurance system in place in order to deal with the risks of drought, fire and timber theft.

Another option is to link on-farm tree planting to climate change mitigation programmes and carbon schemes under the Clean Development Mechanism (CDM) or non-Kyoto compliant voluntary carbon sequestration projects (Jindal et al. 2008). The Kyoto Protocol recognises carbon sequestration through forestry as a way of mitigating global warming (Jindal et al. 2008). Forestry is allowed under the CDM in the form of tree planting or reforestation, with special attention given to small-scale reforestation projects under Article 12 (Decision 19/CP.9) of the Kyoto Protocol, which is meant to assure that low-income communities also benefit from CDM projects (Boyd et al. 2007). Although the tendency is to be optimistic about the potential that carbon markets offer to agroforestry and smallholders (see e.g. Scherr et al. 2004; Montagnini and Nair 2004), several authors have also noted the hindrances related to smallholder's participation in carbon markets, such as the regulatory burden and high transaction costs (Smith and Scherr 2003). In Ghana, the inclusion of smallholders in carbon trade occurs on a pilot scale only and was not identified in the study villages. Considering the importance of generating income from tree farms for the farmers, supporting institutions need to proactively facilitate such carbon projects, as Samartex has done in Asankrangwa FD by facilitating research on assessing carbon contents of planted trees.

Considering the challenges identified in Ghana's on-farm tree-planting scheme, there is a need for a partnership approach (Ros-Tonen et al. 2007) based on cooperation between a wide range of stakeholders. In our view, co-management is not necessarily limited to state and community actors but may also involve a company-community partnership or NGO-community partnership as described in this chapter. Public actor involvement in such partnerships is crucial for the success of tree planting and agroforestry schemes. Firstly, the public sector can create an enabling environment by providing agroforestry extension, by improving road networks and means of transportation that enable efficient marketing of products from the system and by facilitating agroforestry research and the dissemination of its results. Secondly, joint and coordinated action by the FC/FSD and the Ministry of Agriculture may help develop the on-farm tree-planting scheme into a multipurpose agroforestry system from which cash and non-cash incomes can be derived on a more continuous basis than currently is the case. Thirdly, despite efforts to promote co-management between the FC and rural communities, hierarchical governance still prevails in Ghana (Ros-Tonen et al. 2010), as a result of which farmers take a wait-and-see attitude and a dependent stance towards government agencies.

In our view, the kind of agroforestry scheme that is most appropriate for Ghana's high forest zone, especially on fertile lands, is the integration of economic timber trees with permanent agricultural crops such as cocoa, cola and oil palm, with initial integration of food crops. In this case, the adoption of an appropriate tree-crop mix spacing is important to ensure reduction of shade effects, especially on adjacent agricultural crops. Such an agroforestry system has potential for effective livelihood improvements since farmers can benefit from short-term income from food crops, medium-term income from permanent agricultural crops (that have an early maturity period ranging from 3 to 5 years) and long-term income from both the trees and permanent agricultural crops. On marginal lands that are less suitable for permanent agricultural crops, the establishment of woodlots for charcoal and firewood, using species like *Senna siamea*, would be helpful in ensuring a regular provision of incomes to the farmers involved.

It is fair to question whether the introduction of tree planting from above is feasible at all. Tree planting is a relatively new venture for smallholders in Ghana. Integrating tree planting into their farming system therefore requires an adaptive approach to co-management, with mechanisms in place that ensure feedback, joint learning and building of mutual support among the partners (Berkes 2004). Folke et al. (2002, p. 20) define adaptive co-management as 'a process by which institutional arrangements and ecological knowledge are tested and revised in a dynamic, ongoing, self-organized process of learning-by-doing'. Berkes (2008, p. 1698) argues that 'adaptive management and co-management have been evolving toward a common ground because adaptive management without collaboration lacks legitimacy, and co-management without learning-by-doing does not develop the ability to address emerging problems'. Co-management arrangements gradually evolve into adaptive co-management through continuous learning-by-doing (Ibid., p. 1699). In essence, one of the ways of ensuring high productivity and sustainable management of long-term projects like on-farm tree planting, whether under external or self-organised support, is to adopt an attitude of learning from feedback and the experiences of others.

22.6 Conclusions

This chapter reviewed the livelihood outcomes of on-farm tree planting in three forest districts in Ghana's high forest zone. It showed that the scheme contributes substantially to cash and non-cash income based on food crops but that these livelihood benefits are mostly limited to the first 3 years of tree farm establishment, unless trees are interplanted with perennials such as cocoa. Four issues came to the fore as conditions for ensuring adoption and continuous farmer involvement in the scheme.

First, considering the lack of experience and skills in tree planting among smallholders, continuous professional support is crucial for the time being, be it in the form of technical advice, seedlings supply or training in nursery establishment and tree-planting skills. This support can come either from the state, private sector or the leadership of a tree growers association, preferably in a partnership approach.

The importance of secure land tenure and tree harvesting rights (including reducing the bureaucratic requirements of obtaining harvesting and conveyance permits) emerged as a second important condition for farmers' successful engagement in tree planting. Such rights need to be ensured, and the playing field has to be levelled in order for farmers to capitalise on planted trees. Further research is needed to assess how farm size and tenure arrangements affect the adoption and continued engagement in tree planting, in order to obtain more insight into the question of whether tree planting is feasible for poorer farmers – usually migrants – who are engaged in sharecropping arrangements or who have small plots where the planting of trees may compete with food crops.

Third, we identified numerous uncertainties as regards future benefits from tree planting. Together with the lack of experience in tree farming, this highlights the need for an adaptive management approach in order to deal with these uncertainties.

Finally, income sources need to be found to deal with the time lapse between investment and returns from timber trees. This requires further research into (1) possibilities to turn on-farm tree planting into a full agroforestry system that provides non-timber forest products and proceeds from shade-tolerant crops throughout the rotation period, (2) possibilities for farmers to obtain timber benefits before the trees are actually harvested through thinning or gradual purchase or advance payments that compensate them for tree maintenance and (3) possibilities to link tree planting to carbon and other payments for environmental services (PES) schemes.

For sustainable benefits from on-farm tree planting, both in terms of livelihoods and carbon sequestration, it is of utmost importance that on-farm tree planting develops into an agroforestry system from which multiple benefits – including carbon credits – can be obtained to bridge the period between tree farm establishment and timber harvesting. Otherwise, there is the risk that the carbon sequestered will leak away due to farmers encroaching on the forest in search of additional farmland.

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Appendix 1: Activities Involved in On-Farm Tree Planting (Planting and Maintenance Costs)

Scenario 1[a]		Scenario 2[b]	
Activity	Input/quantity/schedule	Activity	Inputs/quantity/schedule
1. Land preparation		1. Land preparation	
Clearing	Labour (20 md)	Clearing	Labour (20 md)
Stumping/debris removal	Labour (20 md)	Stumping/debris removal	Labour (20 md)
2. Planting		2. Planting	
(a) Maize	15 kg seeds + labour (15md)	(a) Maize	15 kg seeds + labour (15 md)
(b) Cedrela/teak	1,111 seedlings[c] + labour (10 md)	(b) Mahogany/ofram	67 seedlings[d] + labour (1 md)
(c) Cocoyam	900 corms + labour (13 md)	(c) Cocoyam	1,000 corms + labour (17 md)
(d) Yam	200 setts + stakes + labour (8 md)	(d) Plantain (3 m × 3 m)	1,111 suckers + labour (25 md)
(e) Plantain	1,111 suckers + labour (25 md)	(e) Cassava	2 ½ head-loads + labour (4 md)
(f) Cassava	2 ½ head-loads + labour (4 md)	(f) Vegetables (pepper, etc.)	250 seedlings + labour (4 md)
(g) Vegetables (pepper, etc.)	250 Seedlings + labour (4 md)	(g) Cocoa (at 3 m × 3 m)	1,111 seedlings + labour (15 md)
3. Maintenance		3. Maintenance	
(a) Weeding – labour (three times each in 1st–3rd yr)	90 md each in 1st–3rd yr	(a) Weeding – labour (three times each in 1st–3rd yr)	90 md each in 1st–3rd yr
(b) Filling in[e]	120 seedlings + labour (2 md)	(b) Filling in timber	20 seedlings + labour (1 md)
(c) Pruning	Labour (5 md) each in 3rd yr	(c) Filling in cocoa	200 seedlings + labour (4 md)
		(d) Cocoa spraying[f] (two times per yr at 2 l, start at 2nd yr)	Chemical + labour (10 md)
		(e) Cocoa – fert. application	Two bags fert + labour (2 md)
		(f) Pruning of cocoa	Labour (2 md)
		(g) Pruning of timber trees	Labour (1 md each in 3rd yr)

(continued)

(continued)

Scenario 1[a]		Scenario 2[b]	
Activity	Input/quantity/schedule	Activity	Inputs/quantity/schedule
4. Harvesting		4. Harvesting	
(a) Maize	Labour (12 md in 1st yr)	(a) Maize	Labour (12 md in 1st yr)
(b) Vegetables (pepper, etc.)	Labour (4 md each in 1st and 2nd yr)	(b) Vegetables (pepper, etc.)	Labour (4 md each in 1st–4th yr)
(c) Cocoyam	Labour (8 and 6 md each in 2nd and 3rd yr)	(c) Cocoyam	Labour (8 and 6 md each in 2nd–4th yr)
(d) Yam	Labour (2 md in 2nd yr)	(d) Yam	Labour (2 md in 2nd yr)
(e) Cassava	Labour (5 md each in 2nd and 3rd yr)	(e) Cassava	Labour (5 md each in 2nd and 3rd yr)
(f) Plantain (2/3 of planted)	Labour (4 md each in 2nd and 3rd yr)	(f) Plantain (2/3 of planted)	Labour (4 md each in 2nd and 3rd yr)

Source: Based on in-depth interviews with six smallholders from the three forest districts

Key: md man days, *yr* year

[a] Based on exotic timber species planted at 3 m × 3 m, with initial stocking of 1,111 seedlings/ha and referring to the first 3 years of establishment (2002–2004).

[b] Based on indigenous timber species interplanted with cocoa at a spacing of 10 m × 15 m, with initial stocking of 67 seedlings/ha and referring to the first 3 years (2002–2004).

[c] Spacing of 3 m × 3 m = 9 m^2.

[d] Spacing at 10 m × 15 m = 150 m^2.

[e] Filling in means planting timber trees in open spaces that are due to trees that failed to survive (in forestry referred to as 'beating up').

[f] Cocoa spraying is done free by the government, but farmers supplement the spraying.

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Chapter 23
Ecosystem Services and Environmental Governance: Some Concluding Remarks

Roldan Muradian and Laura Rival

From a historical perspective, the use of "ecosystem services" as a key concept for describing the relationship between the human societies and the natural environment is very recent (Gómez-Baggethun et al. 2010). Since its introduction, the concept has spread rapidly, and it has become both a heuristic analytical tool for academicians and a powerful discursive tool for conservation practitioners and policymakers interested in the preservation of the natural heritage. The concept is expected to induce a paradigm shift in the management of natural resources (Cowx and Portocarrero-Aya 2011) and to expand the audience for the conservation message by means of showing the links between the natural systems and human well-being (Amsworth et al. 2007; Skroch and Lopez-Hoffman 2009). The utilitarian emphasis of this framework on the economic benefits humans derive from ecosystems and the role of humans and local social institutions in the provision/degradation of these services (Gómez-Baggethun and Kelemens 2008) stands in contrast to the paradigm that previously dominated the field of environmental conservation, which stressed the human/nature division, the trade-off between economic development and the conservation of natural ecosystems, and the corresponding emphasis on the creation of protected areas, set up to exclude all human activity (Sunderland et al. 2008). The new framework is expected to facilitate the creation of novel partnerships, particularly between civil society organizations, local dwellers and corporate entities (Tallis et al. 2009), and therefore to mobilize additional human and financial resources for the conservation of natural ecosystems.

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We identify below some key features of the ecosystem services framework and outline their importance for policy design. The ecosystem services framework aims to:

1. Acknowledge and communicate (e.g. through quantified measurements) the dependency of economic processes on ecosystem functions.
2. Make explicit the linkages between different stakeholders, in particular the users of the resource base (on which the provision of ecosystem services rely) and the beneficiaries of the ecosystem services derived from these resources.

In order to achieve these broad objectives, the ecosystem services approach typically "compartmentalizes" ecosystem services following a classification of values (provisioning, regulatory, etc.) and the type of contribution to economic processes (carbon sequestration, water regulation, etc.).

From a policy perspective, the approach is meant to achieve two critical goals:

1. To help in solving the tension between economic development and environmental conservation
2. To influence the decisions made by the users of a resource base, so that they align their practices with the interests of the beneficiaries of ecosystem services

These two goals constitute the core of the governance agenda that comes associated with the ecosystem services approach. This agenda corresponds to two distinctive areas of action, that of (a) creating linkages between different layers and stakeholders in order to deal with complex economic, social and ecological interdependencies and that of (b) inducing changes in the use or the property rights of the resource base which provides the concerned services, in order to align the interests of different social agents.

Though not necessarily inherent to the ecosystem services framework, this governance agenda has come along with two associated measures, (1) the economic valuation of these services, and (2) the promotion—and increasing use—of market-based policy tools, especially the so-called payments for ecosystem services. The goal is to convert hypothetical (and unrecognized) market values into actual cash flows (Gómez-Baggethun and Ruiz-Perez 2011). Though market-oriented policy approaches are not inextricably linked to the ecosystem services framework, the adoption of this type of policy instrument has been facilitated by two important components of the framework, namely, (a) the compartmentalization of services—which has allowed their commoditization—given that the identification of a tradable "commodity" is a prerequisite for the implementation of market-oriented instruments and (b) the need to create linkages between various levels and stakeholders (with differing interests) and to induce changes in property/use rights among the users of the resource base. In principle, monetary transfers seem to be appropriate tools for both establishing links between social groups and negotiating changes in rights over resources, either through trade or incentives. The promotion and use of market-based policy instruments in the governance of ecosystem services may open new opportunities, but it also entails some threats and challenges, the most important of which we outline below.

The compartmentalization and commoditization of ecosystem services involve a substantial reduction of complexity. Ecosystem functions are typically complex, due to the multidimensional, non-linear and multi-scale (both geographical and temporal) nature of ecological dynamics (Wilson 2006). The capacity of ecosystems to deliver a variety of services depends on a particular combination of features and properties. These complexities (that markets are usually unable to grasp) have been the subject of ecological research for decades. Despite the ecological knowledge gained, our understanding on ecosystem functions, including their drivers and trade-offs, is still very limited. Such complexity, and the associated cost of gathering information about the relationship between ecosystem functions, services and human welfare, is part of the explanation as to why we miss empirical evidence about the link between the conservation interventions and the status of ecosystem services. The lack of data about the effects of interventions on the provision of ecosystem services is pervasive (Brouwer et al. 2011; Farley et al. 2011). Such a gap has led Tallis et al. (2008) to conclude that “most of the current enthusiasm for ecosystem service projects in the conservation world is an act of faith”.

Trade-offs between the provision of different ecosystem services are very common (Rodriguez et al. 2006), as, for instance, between carbon sequestration and water provision (Jackson et al. 2005) or between carbon sequestration and biodiversity (Kanowski and Catterall 2010). An overemphasis on the commoditization and trade of a particular ecosystem service (such as carbon sequestration) may induce changes in the structure and function of the resource base that may in turn jeopardize the supply of other services and even the service whose provision is being promoted. The fact that markets tend to be concentrated on few services may affect negatively the resilience of ecosystems. For instance, large-scale carbon accumulation in forests might favour disruptive fires (Holling 2010). These fires then may eventually undermine the capacity of forests to provide a variety of ecosystem services. Furthermore, the current “obsession” with carbon puts non-forested ecosystems at risk. It may also put at risk the complex and not yet well-understood structure of tropical and other types of forests since, in forests managed for carbon, most species are viewed as superfluous (Putz and Redford 2009). From the point of view of adaptive ecosystem management based on the application of ecological knowledge, the compartmentalization of services is probably the main caveat of the ecosystem services approach. A narrow division of ecosystem services is exacerbated by the use of market-based instruments for environmental governance since markets tend to be focused on very few services; markets are usually myopic to ecological dynamics, that is, unable to grasp their inherent complexity.

In addition, the commoditization of ecosystem services also requires a high level of understanding and predictability of the relationship between the practices of resource use, the ecosystem functioning and the provision of ecosystem services. This information is, in many cases, costly to obtain. As a result, there is often a trade-off between the intention to establish markets for well-defined ecosystem services (which involve verifying that they are actually delivered) and the transaction costs of setting them up (Muradian et al. 2010). More often than not, this makes the commoditization of ecosystem services unfeasible in practice. Furthermore, the assumption that a generalized compensation for the provision of ecosystem services—that is, the internalization of their positive externalities—will lead to a more efficient provision of such services is structurally defective. One might easily argue that the economic system is feasible only because of the "free-of-charge" benefits humans derive from natural ecosystems. A generalized internalization of these positive externalities would lead nonetheless to enormous costs and therefore to economic collapse. Compensation (internalization) is only possible at the margin of the provision of ecosystem services.

The need for coordination between different social actors for the governance of ecosystem services comes from the fact that though ownership of the resource base might be of any kind (private, public or communal), most ecosystem services fall within the type of goods that are considered "common-pool resources". This term refers to goods (i.e. services) with two particular features: potential beneficiaries cannot easily be excluded and there is a high subtractability of use (Ostrom 2010). The fact that the beneficiaries of locally supplied ecosystem services might be in distant locations and often belong to different social groups creates the need for governance systems that transcend the local realm and encompass different geographical and governance scales, even at the global level, such as the emerging regime for reducing carbon emissions from deforestation and forest degradation, or REDD+ (Corbera and Schroeder 2011; Agrawal et al. 2011). Paradoxically, however, in the case of forests, such global institutional arrangements focused on a single service (carbon sequestration) seem to create political incentives towards centralized governance (Sandbrook et al. 2010), thus threatening to reverse a positive trend towards decentralization of resource management in the developing world (Phelps et al. 2010).

The common pool nature of most ecosystem services implies that market mechanisms are not always suitable as governance tools, since markets tend to be more effective in dealing with private goods. The new institutional economics has devoted considerable efforts to explaining why hybrid (i.e. intermediary governance structures positioned between markets and hierarchies) and hierarchical (either firms or states) forms of governance emerge over time, when markets lose their power of coordination. Because of transaction costs and the complexity of the concerned transactions, markets are generally not the most effective coordination mechanism when a high level of cooperation is necessary (Williamson 1991). In addition to the need for cooperation, and given the above-mentioned complexity of ecosystem functioning, the provision of ecosystem services often involves a high level of uncertainty, imperfect and asymmetric information between transacting parties, and cognitive barriers for assessing the service itself (for instance, the extent to which it has been supplied). Consequently, the governance of ecosystem services demands to be approached in terms of nested layers, in a context of uncertain and complex interactions. Due to the high transaction costs involved in the coordination between parties under such circumstances, and as explained at length here, we expect markets to be relatively less effective governance structures for this kind of situations.

A corollary of the argument developed above would be that for the governance of ecosystem services, we could derive more useful insights from the literature on institutional arrangements for governing common-pool resources than from the literature on Coasean approaches to resolve environmental externalities. For instance, we argue that it is analytically more appropriate to conceptualize payments for ecosystem services as incentives for collective action (Muradian et al. 2010) instead of quasi-perfect market transactions to solve market failures (Engel et al. 2008). Such a different point of departure has important implications, not only from a conceptual point of view—the way the problem of ecosystem degradation is understood and analysed—but also in terms of policy and practice, that is, the way conservation and rural development interventions and policies are designed. Payments for ecosystem services and other so-called market-based mechanisms are above all political instruments and not only technical tools for getting the price of ecosystem services right or to correct economic externalities, as some authors have argued. McAfee and Shapiro (2010) have put it elegantly: "PES projects entail political choices about which classes of people, in which geographical locations, will have access to natural resources and their benefits now and in the future". PES projects, we concur with these authors, are political projects embedded in complex institutional and ecological contexts.

The literature on the commons has stressed the role of sanctions. New insights may nevertheless be derived from incorporating considerations about the role of monetary and non-monetary incentives in coordinating activities for the management of common resources (i.e. ecosystem services). Many scholars will undoubtedly spend the next few years trying to perfect the integration of the notion of incentives (rather than that of sanctions) within the institutional analysis of social dilemmas involving the management of natural ecosystems. We would nevertheless like to stress that there is already a large body of work on the management of common natural resources, which allows us to draw preliminary lessons that can be applied to the governance of the provision of ecosystem services. The literature on institutions for managing the commons stresses that these governance arrangements tend to be more effective in solving social dilemmas when they are built on local knowledge and trust (Ostrom 2011) and when they hold high levels of involvement of stakeholders in the design and enforcement of rules (Ostrom 2012), including monitoring and sanctioning (Coleman 2009). More than the general type of governance of property rights (government-, private- or community-based), what really matters is (1) how a particular arrangement fits the local ecological conditions, (2) how rules are developed and adapted across time and (3) how social actors perceive these arrangements in terms of legitimacy and equity (Cole and Ostrom 2011; Ostrom 2012).

To sum up, rules and rule-making autonomy and participation (i.e. how rules are designed and enforced and how they evolve over time) matter more than the property regime or the generic type of coordination between transacting parties (Banana et al. 2007; Chhatre and Agrawal 2009; Cox et al. 2010; Bastakoti and Shivakoti 2011; Ostrom and Basurto 2011; Persha et al. 2011). The insights gained through the broad range of case studies and conceptual chapters that compose this volume are in line with this conclusion. In short, the contemporary overemphasis on market-based instruments is misleading. It is true that these tools may, under specific circumstances, contribute to improving the governance regimes of natural ecosystems, but we must nevertheless put the necessary attention to their particular fit within specific socio-economic contexts and their capacity to modify rule-making structures. These two aspects are determinant when it comes to both their effectiveness and their social acceptability. At the end of the day, there is no escaping to the old concern for the suitability of rules, including by whom and for whom they are made. In other words, there is no escaping to the need to attend to political choices. As it is being demonstrated by current debates around the implementation of REDD+ policies, multilevel governance of ecosystem goods and services induces not only new ways of doing economics but also new forms of political theory and political action. Multilevel governance geared to protect the commons through enhanced cooperation holds the promise of redefining justice, recognition, redistribution, power, democracy, citizenship, the state and many other political categories. We hope that readers will find in the facts and analyses contained in this book matter to renew the political imagination and inspiration to act with and for the commons.

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