Forest conservation for communities and carbon: the economics of community forest management in the Bale Mountains Eco-Region, Ethiopia

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Declaration of work

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Abstract

Forest conservation based on payments anchored to opportunity costs (OCs) is receiving increasing attention, including for international financial transfers for reduced emissions from deforestation and degradation (REDD+). REDD+ emerged as a payment for environmental service (PES) approach in which conditional payments are made for demonstrable greenhouse gas emission reductions against a business-as-usual baseline. Quantitative assessments of the OCs incurred by forest users of these reductions are lacking. Existing studies are coarse, obscure the heterogeneity of OCs and do not consider how OCs may change over time.

An integrated assessment of OCs and carbon benefits under a proposed community forest management (CFM) intervention linked to REDD+ is undertaken in Ethiopia. The OCs of land for the intervention are estimated through household survey and market valuation. Scenarios explore how OCs are likely to change over the intervention given qualitative conservation goals and available land-use change information. The feasibility of OCs payment as a tool for REDD+ is assessed by combining cost with emission reductions estimates generated from direct tree measurements. Households' environmental attitudes, perceptions and intention to cooperate with the intervention, estimated by a voluntary contribution to improve forest management, are then investigated.

Mean OCs of forest conservation are US$334/ha, but highly heterogeneous. Plausible futures of agricultural improvement, forest product commercialisation, and degradation of land uses suggest total OCs could approach US$441 million over a 20-year project. Applying carbon stock estimates of 231tC/ha±52 in moist and 132tC/ha±73 in dry forest, REDD+ revenues may not meet annual cumulative OCs, although more nuanced conservation planning could reduce OCs. Despite

OCs all households intend to cooperate in the intervention, with mean contribution of US$11±4/year/household. The expected incomes of households under the Bale REDD+ Project intervention however, were high and expectation management is necessary. Recommendations are made for REDD+ intervention design in Ethiopia.

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List of acronyms

BAU	Business-as-usual
BERSMP	Bale Eco-Region Sustainable Management Programme
BME	Bale Mountains Eco-Region
BMNP	Bale Mountains National Park
CBOs	Community based organisations
CCBA	Climate, Community and Biodiversity Alliance
CDM	Clean Development Mechanism
CFM	Community forest management
dbh	diameter at breast height
ETB	Ethiopian Birr
EPRDF	Ethiopian People's Revolutionary Democratic Front
ERPA	Emissions Reductions Purchase Agreement
FCFP	Forest Carbon Partnership Facility
GHG	Greenhouse gases
GIS	Geographical Information System
GTZ	German Technical Corporation
ha	hectares
hifp	high-impact forest products
HH	household
HWPs	harvested wood products
IPCC	Intergovernmental Panel for Climate Change
lifp	low impact forest products
LULUCF	Land use, land-use change and forestry
masl	metres above sea level
MRV	Monitoring, reporting and verification
NGO	Non-governmental organisation
NTFP	Non-timber forest products
OCs	Opportunity costs
OFWE	Oromia Forest and Wildlife Enterprise
OLS	Ordinary least squares
OTC	Over-the-counter
PES	Payments for environmental services
REDD+	Reduced emissions from deforestation and degradation, forest conservation, sustainable management of forests and the enhancement of forest carbon stocks
R-PP	Readiness Preparation Proposal
tCO_2e	tonnes of carbon dioxide equivalents
UNFCCC	United Nations Framework Convention on Climate Change
VCM	Voluntary carbon market
VCS	Voluntary Carbon Standard
WBISPP	Woody Biomass Inventory and Strategic Planning Project
WTP	Willingness-to-pay

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Chapter 1: Introduction

1.1. Background

Climate regulation is a non-material, non-extractive, environmental service that historically was non-marketed. Now recognised as a global public good, GHG emission reductions are now traded. Both regulated and unregulated carbon markets have grown substantially over the last five years and in 2010, carbon markets were worth US$142 billion (World Bank, 2011). Forests play an important role in climate change mitigation and deforestation is responsible for 17% of global greenhouse gas emissions annually (Stern, 2007, VCS, 2007). Reduced emissions from deforestation and degradation, forest conservation, sustainable management of forests and the enhancement of forest carbon stocks; henceforth referred to as REDD+, presents a substantial climate change mitigation opportunity (McCarl and Schneider, 2001, Sohngen and Mendelsohn, 2003, IPCC, 2006). International financial transfers for REDD+ are growing; forest carbon markets traded an estimated US$178 million in 2010 (Diaz et al., 2011). REDD+ is also operates outside of carbon market mechanisms, and substantial public money is going to support REDD+ activities (Watson and Nakhooda, 2012).

Where it operates at a local-level, REDD+ might be considered a payment for environmental service (PES) scheme whereby the environmental service of carbon dioxide emission reductions are sold, through a voluntary transaction, and payment is conditional upon the provision of that service (Wunder, 2005). Others ways to finance REDD+ exist, but a well-functioning PES can help deliver the environmental integrity, or effectiveness, of a REDD+ mechanism that relies on real, permanent and verifiable emission reductions (UNDP, 2009). Accounting for emission reductions from forest activities, however, involves substantial uncertainty (Brown and Lugo, 1992, Monni et al., 2007, Grainger, 2008, Larocque et al., 2008). This is particularly true for forest carbon stocks where uncertainty arises from complexity in forest ecosystems, sampling errors and from the choice of model parameters, and is compounded by uncertain carbon market variables such as carbon price, project preparation costs and transaction costs.

Despite the uncertainty it entails, the application of documented biome-average forest carbon stocks has become commonplace for emission reductions accounting (Brown and Gaston, 1995, Gibbs et al., 2007). Biome averages are simple and quick to apply without resource and logistical constraints, but biome averaged data rarely captures the full heterogeneity of the forest landscape (Houghton and et al., 2001, Bradford et al., 2010). Few studies have considered the discrepancy between the application of such simple default data and more complex forest carbon accounting methods. The discrepancy can be large, but there is no consensus on its direction (Smith, 2003, Brown et al., 2007). For REDD+ to be effective, policy-makers need to better understand the uncertainties of emission reductions accounting. Over-estimation of emission reductions can lead to large sums of finance being miss-directed for no climate benefit, while underestimation can result in lost opportunities for climate change mitigation and for the local realisation of economic incentives.

There is growing support for REDD+ to be delivered through community forest management (CFM) (Klooster and Masera, 2000, Murdiyarso and Skutsch, 2006, Agrawal and Angelsen, 2009, Hayes and Persha, 2010). In part this stems from findings that CFM can lead to emissions reductions where forest use becomes more sustainable (Chhatre and Agrawal, 2009, Skutsch and Ba, 2010). It has also been shown that community monitoring, reporting and verification of emission reductions can be less costly that equivalent costs of professionals or central forest departments (Somanathan et al., 2009, Palmer Fry, 2011). Under CFM, a common property regime is established where members of a well-defined group of people establish collective regulations for resource use, membership, monitoring, and sanctioning procedures (Arnold, 2001, Baland and Platteau, 2003). To deliver emission reductions those participating in REDD+ must have sufficient incentives to do so. It has been proposed that an estimate of the private opportunity costs (OCs) of forest conservation – the foregone benefits of alternative land uses – could be used to anchor the level of payment needed to achieve the desired level of forest conservation for REDD+ (Pirard, 2008, Wertz-Kanounnikoff, 2008, Pagiola and Bosquet, 2009, White and Minang, 2011). These OCs should be embodied within the payment that stakeholder, who become the providers of ecosystem services, are willing to accept (Ferraro, 2008).

Studies of the OCs of forest conservation are typically based on the OCs of foregone agricultural production (e.g. Chomitz et al., 2005, Naidoo and Adamowicz, 2006, Börner et al., 2009). However, under CFM wider restrictions on resource use experienced by households may mean that the forgone revenues from timber and non-timber forest products (NTFP) will also be relevant costs to consider (Karky and Skutsch, 2010, Fisher et al., 2011). Studies of the OCs of REDD+ have also largely been undertaken at broad-scales which do not translate well to on-the-ground design of payment incentives (e.g. Grieg-Gran, 2006, Grieg-Gran, 2008). In particular, these broad-scale studies of the OCs of REDD**+** do not sufficiently appreciate the heterogeneity in the OCs of forest users which result from differences in the economic reliance of households on forests (Vedeld et al., 2004).

REDD+ OC studies have also largely excluded forests under community management or that are common pool resources. This is largely attributed to the unclear rights to land, trees or carbon under such property rights regimes, or because forest use is illegal (e.g. Grieg-Gran, 2006, Börner and Wunder, 2008). Where the costs of REDD+ via CFM have been considered, the mechanism to deliver emission reductions becomes more 'PES-like'; where the incentives provided for service provision lack conditionality on the delivery of the service provision (Nepstad et al., 2007, Peskett et al., 2008, Skutsch et al., 2011, Goldman-Benner et al., 2012). This contrasts the REDD+ literature that talks of direct, output based payments that are strongly additional to the business-as-usual (BAU) baseline and conditional on continued service provision (see Santilli et al., 2005, Parker et al., 2008, Bond et al., 2009). More research into the form, magnitude and heterogeneity of OCs of REDD+ via CFM is necessary where communities become legal providers of the environmental service generated through REDD+ via CFM. These OC estimates can inform the design of a REDD+ via CFM conservation intervention to encourage enrolment in the local-level PES scheme, maintain conditionality and additionality and reduce the displacement of emission reductions due to inadequate incentives.

At the core of conservation interventions are also concerns for the persistence of a resource system into the future. Of the few studies of the OCs of conservation, however, most report OCs for a single year or assume OCs are constant over time subject only to discounting (Chomitz et al., 2005, Naidoo and Adamowicz, 2006, Börner et al., 2009). Where resource use is unsustainable this assumption of constant OCs are unlikely to hold (Pearce and Markandya, 1987, Ferraro, 2002). The OCs of REDD+ are a function of the drivers of land-use change and so will be influenced by changing income from direct human activities such as agricultural production. Conservation interventions also aim to alter economic incentives that will impact on OCs, for example, through the commercialisation of forest products (Brandon and Wells, 1992). The OCs of forest conservation over time are, however, hard to predict. There is often a lack of information on trends in the productivity of land uses and a lack of explicit goals and quantitative operational targets in conservation (Margules and Pressey, 2000). Being able to overcome the OCs of local forest stakeholders over time will be necessary for the longevity of the conservation intervention and the permanence of emission reductions from REDD+. Generating a better understanding of possible futures of OC is therefore necessary for more appropriate intervention and incentive design.

While PES uses a payment incentive to alter land use behaviours, CFM relies more on the overall impact on a household's payoffs that result from a change in the property rights regime, reputation, trust and reciprocity (Ostrom, 2000, Castillo and Saysel, 2005, Agrawal, 2003). Behaviours rely on the underlying values that individuals hold, themselves driven by motivation and belief systems (Kotchen and Reiling, 2000). An understanding of the attitudes and perceptions of participants of conservation interventions can go some way to explore these values and beliefs. A number of studies show that knowledge and perception of the resource base condition, of perceived environmental responsibilities, and of perceived legitimacy of the intervention are important for conservation success (Zanetell and Knuth, 2004, Davies and Hodge, 2006, Nkonya et al., 2008, Adams et al., 2003). However, none have explored how an ex-ante study of attitudes and perceptions can contribute to incentive design to encourage cooperation. The ongoing cooperation of local stakeholders in REDD+ activities will be critical for the longevity, or permanence of emission reductions (see Sedjo and Marland, 2003). On common property regimes, cooperation on a common pool resource can beget more cooperation and self-restraint in forest use brings more significant benefits when followed by sufficiently large number of users (Baland and Platteau, 1996, Castillo and Saysel, 2005). Greater cooperation, or conservation effort, under CFM can therefore increase REDD+ revenues. An understanding of local stakeholder's attitudes towards forest management and the use of the resource base will, therefore, allow better consideration of socio-cultural factors for cooperation that go beyond payment incentives that PES theory highlights.

A country associated with drought and poverty, forests do not immediately come to mind when images of Ethiopia are evoked. But the largely rural population is highly dependent on the forest resource base: across the country forest income is estimated to be around a third of total household income (Mamo et al., 2007, Babulo et al., 2009, Tesfaye et al., 2011). Sustainable forest management has been hindered by political instability and a focus on increasing food production and security (Teketay et al., 2010). Poor governance, uncertain land tenure, and a rapidly growing population means that Ethiopia is experiencing forest losses amounting to 140,000 hectares each year (WBISPP, 2005). With high levels of poverty characterising Ethiopia, forest conservation that also allows households to meet their livelihood needs is urgent (WDI, 2011).

CFM is being scaled up across the country with a view to meet livelihood needs and to conserve the remaining natural forest areas. In the Bale Mountains Eco-Region (BME) deforestation rates are more than four times the country-wide average (Dupuy, 2009, Teshome et al., 2011). The BME is not a WWF eco-region, however, it is referred to in this thesis as an eco-region so as to be consistent with the Bale REDD+ Project implementers at the case study site as well as the national use of the term to refer to this area. The Bale REDD+ Project has been proposed and initiated by the Government of Ethiopia (Oromia Regional Government, Bureau of Agriculture and Rural Development, and the Food Security and Disaster Prevention and Preparedness Commission) and NGOs FARM-Africa and SOS

Sahel Ethiopia: the Bale REDD+ Project implementers. The project area covers 900,000 ha including dry and moist tropical forest which is currently being lost at 4% annually. In order to reduce deforestation over a 20 year period, CFM will be implemented alongside promotion of fuel-efficient stoves and biomass briquettes and plans are underway to plant woodlots and manage fire outbreaks. Increasing agricultural production and the value of NTFP will also occur as part of the project. While CFM and REDD+ can both be undertaken as separate policy interventions, in the BME these are considered together: the Bale REDD+ Project undertakes REDD+ via CFM. Thus emission reductions do not have to be additional to that achieved through CFM, but rather are those generated by CFM.

Some do not consider Ethiopia to be a 'key country' for REDD+. Efforts to establish REDD+ projects and activities have focussed on countries where forest areas are more substantial and the carbon contained within the forests is very high. This includes Brazil, Indonesia and the Democratic Republic of the Congo where the majority of international finance to support REDD+ development has been channelled (Climate Funds Update, 2011). Establishing REDD+ in Ethiopia, therefore, may not contribute significantly to reducing emissions from deforestation assessed at an international scale. Ethiopia may not receive as substantial financial transfers as other tropical forested nations under an international REDD+ mechanism established by climate change negotiations. REDD+ does, however, contribute to internalising the externality of climate regulation. It could provide a source of finance that changes the economic incentives to make forest conservation more economically viable and it necessitates the discussion and review of property rights regimes in forested areas. It could also provide much needed finance that can help promote forest conservation in a country with limited public budgets for forest conservation. It is for these reasons, in addition to the potential climate benefits, that a discussion about REDD+ in

Ethiopia is justified. With CFM being pursued in a number of national REDD+ strategies in East Africa, including Ethiopia, the BME REDD+ project could prove exemplary for the how REDD+ via CFM might function on-the-ground (FCPF, 2011).

1.2. Research questions

In this thesis, I aim to increase the understanding of how REDD+ can be implemented through CFM as a local-level PES scheme in a developing country. This thesis addresses a number of identified gaps in the literature on the uncertainty of forest carbon stock accounting, the OCs of REDD+ via CFM, the OCs of forest conservation over time, and community-level PES. A proposed REDD+ via CFM forest conservation intervention in the Bale Mountains of Ethiopia provides an ideal case study to explore how information on the OCs of land, the uncertainty in OCs over time, and household's attitudes and perceptions can inform the on-the-ground design of a REDD+ via CFM intervention.

The specific research objectives of this thesis are:

- To estimate the forest carbon stock in the BME;
- To evaluate the discrepancy between simple and complex forest carbon accounting methods and the implications for the environmental integrity of a REDD+ mechanism;
- To estimate the OCs of a proposed shift from an open access forest management regime to forest conservation via CFM;
- To explore changes in the OCs of forest conservation over time in light of uncertainty in the conservation intervention objectives and paucity of data on future productivity of land uses;

- To establish if potential REDD+ revenues can overcome the OCs of forest conservation over time; and
- To investigate the attitudes and perceptions of forest management in the BME and consider households' intentions to cooperate in the proposed intervention through a voluntary contribution to the community group.

1.3. Thesis structure

This thesis is structured as follows and is also represented in Figure 1:

Chapter 2 places this research in context of the existing literature on PES and REDD+ via CFM. Identifying the gaps in the literature, it highlights the need and timeliness of research into the implementation of REDD+ via CFM.

Chapter 3 introduces the case-study site in the BME in Ethiopia and reviews forest policy and management both past and present. A detailed description of the proposed forest conservation intervention at the case-study site is also given including the likely rules of CFM.

Chapter 4 provides the conceptual framework and the methods employed in this analysis of forest conservation for carbon and communities.

Chapter 5 estimates the carbon stock of forest at the case-study site and explores the uncertainty of forest carbon stock estimates and the resulting environmental integrity of emission reductions. It also estimates the potential REDD+ revenue that a project in the BME could generate. Chapter 5 adds to knowledge though the collection of primary data and estimation of forest carbon stocks in the BME. It builds on limited literature on the implications of forest carbon accounting method discrepancies for the environmental integrity of REDD+.

Chapter 6 investigates household income from the forest resource base and from cultivated land to allow the estimation of the OCs of avoided deforestation under a CFM regime. Chapter 6 adds to the limited literature on the OCs of forest conservation, in particular the OCs of REDD+ via CFM, as well as to the limited discussion of PES at the community-level.

Chapter 7 considers three futures through scenario modelling, exploring the OCs of land over the life-span of the proposed conservation intervention. It also assesses whether REDD+ revenues are sufficient to overcome the estimated OCs. Chapter 7 contributes to knowledge by applying scenario modelling in conservation planning. This Chapter also contributes to the limited literature on the OCs of conservation over time.

Chapter 8 examines the attitudes and perceptions of the local communities towards forest management. It also elicits their expectations of, and intention to cooperate in the proposed REDD+ via CFM forest conservation intervention. Chapter 8 adds to knowledge by eliciting environmental attitudes and perceptions of a CFM intervention ex-ante, and illustrating how this information can be used for intervention design.

Chapter 9 highlights the key findings of this thesis and how they may influence policy formulation at the case-study site, as well as making recommendations for future research.

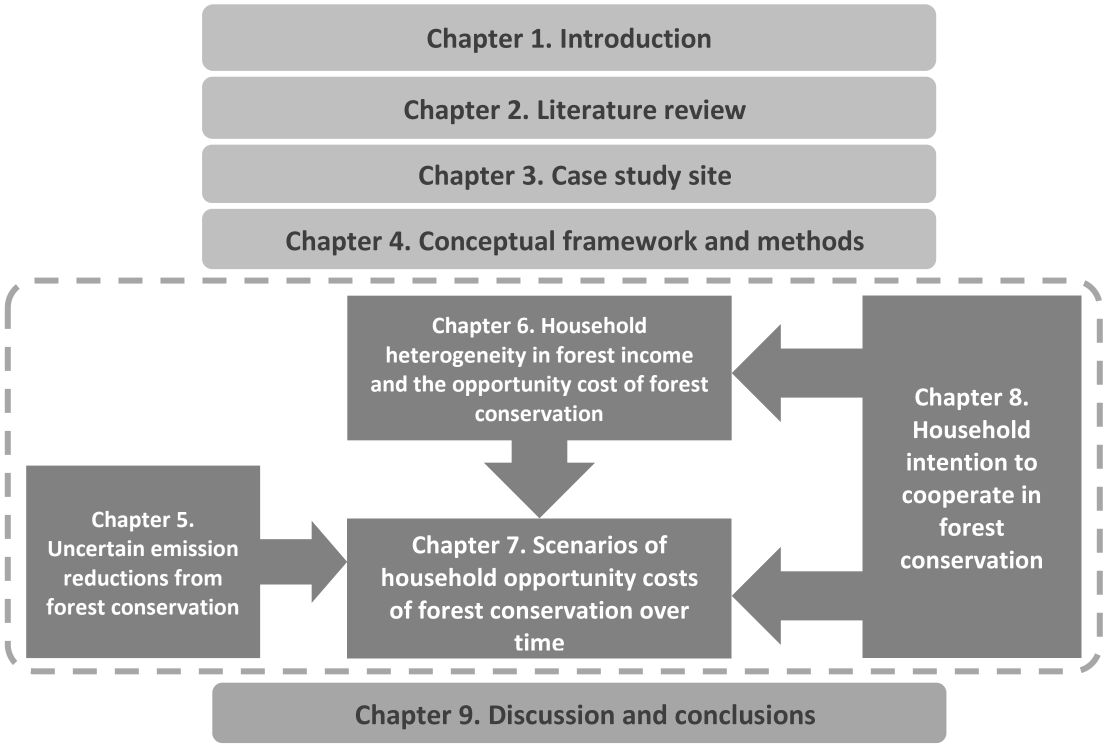

Figure 1. Schematic of thesis structure

Chapter 2: Literature Review

2.1. Introduction

Each year, 13 million hectares of forest are lost globally to expanding agriculture, infrastructure and wood extraction (Geist and Lambin, 2001, FAO, 2006). A store of carbon, this forest loss is responsible for around 12 to 20% global greenhouse gas (GHG) emissions (FAO, 2006, Stern, 2007). A mechanism that reduces emissions from deforestation and degradation, REDD+ presents a substantial climate change mitigation opportunity (McCarl and Schneider, 2001, Sohngen and Mendelsohn, 2003, IPCC, 2006). REDD+ goes some way to address market and policy failures in forest management that have historically undervalued or excluded from the market, forest products and services. As initially conceived, financial transfers to those who conserved and generated climate change mitigation potential through forestry activities, therefore, go some way to internalise positive environmental externalities such that non-marketed costs or benefits are reflected in the profits received by forest stakeholders (Richards, 1999).

Not requiring technological innovation of the scale required in many other sectors and with much of deforestation thought only marginally profitable (Boucher, 2008, Minang et al., 2008), REDD+ has been promoted as a cost-effective climate change mitigation option (Nabuurs et al., 2007, Bellassen and Gitz, 2008). The 2008 Eliasch review found that the inclusion of emissions abatement through the forest sector could greatly reduce the total estimated cost of halving global carbon emissions from 1990 levels, as compared to the forest sectors exclusion. These cost reductions were estimated as much as 50% in 2030 and 40% in 2050 (Eliasch, 2008).

REDD+ can be generated from more established ways to protect forests; protected areas, sustainable logging, integrated conservation and development projects, modifying plans for infrastructure such as road building, or through recognition of rights (Rudel et al., 2005, Chomitz, 2007, Nepstad et al., 2007, Boucher, 2008). REDD+ could also be funded in a number of different ways and not all of these would require payment to reach local forest stakeholders. Strassburg *et al.* (2009) illustrate that the costs of REDD+ in developing countries may be met by selling emission reductions in national, regional or global carbon markets that can raise substantial amounts of money; or from intermediate market-linked systems, not purchased as offsets or linked or market prices; or through official development assistance and other public funds. Such international public funds for REDD+ have been more flexible, allowing countries to prepare the enabling environment for the scaling up of REDD+ in addition to delivering actual emission reductions (Watson and Nakhooda, 2012). Advantages of each source of finance differs (Boucher 2008). With negotiations failing to make progress on aspects of climate finance as well as REDD+ finance, in the immediate future there will be a combination of sources of finance for REDD+. Similarly, most existing and planned REDD+ projects combine a number of policies, actions and measures (PAMs) to deliver REDD+. The proposed Bale REDD+ Project under scrutiny in this thesis, for example, combines CFM with, woodlots, fuel-efficient stoves, and biomass briquettes to reduce fuelwood needs, support for agricultural intensification to reduce needs for expanding agricultural land, and support for the development of NTFP and a 'Bale Wild' branding to increase the local value of products such as forest coffee and honey (see Chapter 3).

While an international REDD+ mechanism is likely to operate through national-level institutions in the future, the effectiveness at a country level will rely on successful local-level forest conservation (Hayes and Persha, 2010). Although it is acknowledge that some PAMs for REDD+ may not require finance to reach local forest stakeholders, for example, through national forest tenure reform and strengthening of enforcement against illegal logging practices, however, there has been much attention paid to ensuring that benefit-sharing does reach such levels (Costenbader, 2011, Peskett, 2011, Hoang et al., 2013). This thesis adopts an approach where, at local-level, REDD+ might be regarded as a PES scheme where a well-defined environmental service is bought by at least one buyer, from at least one provider, through a voluntary transaction and conditional upon the provision of that service (Wunder, 2005). Local-level REDD+ should, therefore, operate where the willingness-to-pay for a service exceeds a provider's opportunity costs (OCs) of alternative, or foregone, land uses and practices that generate emission reductions (Pirard, 2008, Wertz-Kanounnikoff, 2008, Pagiola and Bosquet, 2009, White and Minang, 2011). The ability to incentivise such forest conservation depends on REDD+ revenues to a project, themselves dependent on the emission reductions that are generated from forest conservation activities (Santilli et al., 2005, Parker et al., 2008, Bond et al., 2009).

Efforts have been made to ensure real, permanent and verifiable emission reductions are generated from REDD+ (eg. Brown et al., 2007, UNDP, 2009). A requirement of the payment is that emission reductions are additional; they would not have occurred in the absence of the intervention (Asquith et al., 2002, Rodríguez Zúñiga, 2003, Rojas and Aylward, 2003). The delivery of additional emission reductions requires that only those who threaten forest cover should be paid, despite any discontent this sparks for existing good forest stewards (Wunder, 2005). Cost-efficiency requires that those providing the environmental service should only be paid their costs, thus payments would ideally be differentiated between forest stakeholders. The longevity of emission reductions is also important. Where REDD+ gains are not permanent, no overall emission reductions will be made thus challenging the environmental integrity of a REDD+ mechanism (Marland et al., 2001, Sedjo and Marland, 2003).

There have been a number of calls for REDD+ to be delivered via community forest management (CFM) (Klooster and Masera, 2000, Murdiyarso and Skutsch, 2006, Agrawal and Angelsen, 2009, Hayes and Persha, 2010). Of course, this is one of many policies, actions and measures that can lead to REDD+. The establishment of protected areas, reduced impact logging and agro-forestry, for example, can also generate emission reductions (Watson, 2012). CFM inherently addresses the livelihood needs of communities, however, as it is implemented where centralised, state management is recognised as ineffective at sustainable forest management and/or where benefits are not distributed equitably (Agrawal and Angelsen, 2009). Necessarily understanding and addressing local livelihood needs such as biomass energy, CFM as a way to implement REDD+ may therefore reduce the risks and associated costs of dealing with the displacement of REDD+ outside of the project area as opposed to the generation of a protected area, or greater law enforcement as a main tool to reduce forest loss and decline. Where the agents of deforestation shift their activities or meet demands for the same products from other locations, such displacement is termed primary leakage (Aukland et al., 2002, Smith and Scherr, 2003).

The costs of monitoring, reporting and verifying (MRV) emission reductions and community enforcement of regulations required by REDD+ can also be lower where provided by communities than equivalent labour and administration provided by professionals and central forest departments (Somanathan et al., 2009, Skutsch and Ba, 2010). REDD+ via CFM, therefore, could be competitive, or more cost-effective, than alternative, mechanisms to deliver emission reductions

(Chhatre and Agrawal, 2009, Karky and Skutsch, 2010, Danielsen et al., 2011, Palmer Fry, 2011).

REDD+ via CFM may also increase participation in PES, particularly for the poor who are less likely to hold private titles to land to contract in individual-based PES schemes (Miranda et al., 2003, Zbinden and Lee, 2005, Kosoy et al., 2007). Such a community-level PES can also reduce barriers to participation such as high upfront investments, such as for tree seedlings for private land, or transaction costs, such as negotiating payments, that the poor may be less able to meet if they were acting individually (Gong et al., 2010). Revenues for REDD+ via CFM could strengthen the incentives to cooperate in forest conservation (Agrawal and Angelsen, 2009). Addressing a market failure that drives deforestation, the recognition of the international value of carbon storage in addition to the local values driving improved community-level forest management, would help local forest stakeholders international the previously non-market benefit of climate change mitigation. Klooster and Masera (2000) suggest that carbon mitigation could also leverage finance for the local investments needed to build local CFM capacity and knowledge.

There has been little consideration, however, of how REDD+ via CFM will operate on-the-ground. REDD+ necessitates a change in forest area, management regime and access for those local to the conservation intervention. Rural communities often depend heavily on natural resources for their livelihoods (Forsyth et al., 1998, Bishop, 1999). The changes in livelihood strategies and opportunities that this implies are unlikely to be identical between households local to conservation interventions. Cost information can contribute towards a better understanding; leading to more effective conservation interventions (Polasky et al., 2001, Polasky et al., 2005, Naidoo et al., 2006, Naidoo and Iwamura, 2007, Carwardine et al., 2008). Studies into the OCs of forest conservation, however, remain infrequent. Experience of PES in developing countries indicates that environmental service provision is often hard to attribute, payments are largely uniform and input-based with indirect and in-kind incentives, especially where PES operates at a community-level (Sommerville et al., 2009, Southgate et al., 2009, Skutsch et al., 2011).

PES in practice, therefore, largely does not appreciate the on-the-ground heterogeneity in the OCs of forest conservation. Furthermore, few studies have considered how the OCs of conservation change over time. This is despite the fact that conservation interventions are put in place largely due to concerns about the ecological and social sustainability of a resource system now and into the future (Pearce and Markandya, 1987, Ferraro, 2002). PES in practice appears less conditional on service delivery, less additional to the baseline, and payments may not fully overcome OCs and sufficiently incentivise ongoing service provision.

The PES literature has also afforded little attention to environmental services generated under a common property regime, as would be the case for REDD+ via CFM (Muradian et al., 2010). A considerable body of research has demonstrated that individuals can collaborate to manage a common pool resource more sustainably than in the absence of cooperation (Ostrom, 1990, Bromley, 1992, Bardhan, 1993, Baland and Platteau, 1996). The mechanisms and incentives for cooperation in common property regimes rely on positive incentives and penalties, as well as social norms and codes of conduct (Ostrom, 1990, Ostrom, 2000, Castillo and Saysel, 2005). Social and cultural norms include preferences for altruism, reciprocity, inequity aversion, reputation, trust and conformity with the wider community (Velez et al., 2009). These may influence the payoffs of a REDD+ via CFM conservation intervention, defined as the balance of costs and the benefits, both *perceived* and experienced, by the resource appropriators. PES, however, largely ignores the logic of collective action under which individuals may be willing to sacrifice private benefits or private consumption to raise public benefit provision so long as it is reciprocated (Vatn, 2010, Vicary, 2011).

Skutsch *et al*. (2011) address the core difference in incentives between REDD+ and CFM in detail. They consider output-based payments to communities not likely to be appropriate under CFM and suggest more manageable input-based incentives should be considered such as employing communities in MRV activities, or through alternative income generating activities. The incentives become more 'PES-like' and so do not fulfil all the criteria of the classic PES definition (Landell-Mills and Porras, 2002, Wunder, 2008). As is found in other studies that consider REDD+ via CFM, there is a move away from the conditionality of the payment on delivery of the emission reductions generated (Nepstad et al., 2007, Peskett et al., 2008). Thus REDD+ via CFM to date appears removed from the performance, or output-based REDD+ payments commonly discussed (see Santilli et al., 2005, Parker et al., 2008, Bond et al., 2009).

2.2. The carbon benefits of forest conservation: REDD+ revenues

Finance delivered through a REDD+ mechanism has the potential to bring a greater and more sustainable source of finance to conserve environmental services than often exists now (Landell-Mills, 2002, Pagiola et al., 2005a). It has garnered much attention as forest conservation in developing countries is often underfunded, and this is particularly true in Africa. Independent of whether REDD+ is financed through carbon markets or public funds (as discussed in Section 4.2.1), recognising and realising the value of climate regulation could attract more international and domestic finance over a longer period of time if carbon remains preserved within tree biomass.

The shift to such market-based instruments for conservation follows a shift in conservation towards meeting the basic needs, food and livelihood security of local stakeholders. Itself based on a growing understanding of the relationship between people and the environment-resource system (Arnold, 2001). Programmes for both conservation and development have emerged that attempt to diversify livelihoods to reduce pressure on forest resource systems, to commercialise and increase prices of forest products to increase the economic value of standing forest, and that pay stakeholders for the provision of an environmental service (Brandon and Wells, 1992). REDD+, however, goes further than integrated conservation and development projects (ICDPs) by attempting to address market and policy failures which undervalue or exclude from the market forest products and services, or that make other land uses more profitable (see also Wunder, 2012). REDD+ goes some way to internalise positive environmental externalities such that non-marketed costs or benefits are reflected in the profits received by forest stakeholders (Richards, 1999). For REDD+ to function as a local-level PES scheme, revenues from the sale of emission reductions must overcome the costs of forest conservation experienced by the local forest stakeholders.

In order to estimate the REDD+ revenues that can be generated by an intervention that avoids deforestation, information on forest carbon stocks, area change and market variables is necessary. Advances are being made in the technology and accessibility of remote sensing imagery for the measurement of forest area and forest area change and it is being increasingly used to infer forest biomass and so forest carbon stocks (Achard et al., 2004, Mayaux et al., 2005, DeFries et al., 2007, Ramankutty et al., 2007, Baccini et al., 2008, Goetz et al., 2009, Bucki et al., 2012).

Appropriate methods to establish the past and predicted rates of forest change in order to calculate the emission reductions resulting from an intervention also continue to be developed (Angelsen, 2008, Olander et al., 2008, Bond et al., 2009, Griscom et al., 2009, Huettner et al., 2009, Estrada, 2011). Although forest carbon stock estimation is being studied, relatively less attention has been paid to reducing uncertainty in this field with regard to REDD+ interventions.

The scale of forest ecosystems and complexity of interactions between environmental services within forest ecosystems means that there is more uncertainty in carbon accounting than in any other climate change mitigation sector (Peltoniemi et al., 2006, Larocque et al., 2008). Many developing countries also suffer from a lack of data on key forest variables and parameters, and/or resources or capacity to undertake forest carbon stock inventories (Brown et al., 1989, Smith and Heath, 2001, Andersson et al., 2009, Romijn et al., 2012).

The high uncertainty in carbon accounting is partly to blame for the absence of forestry in the European Emissions Trading Scheme (Fogel, 2005). It has also resulted in limited eligibility, strict definitions, accounting rules and caps for land use, land-use change and forestry (LULUCF) in national emissions accounting – required by developed country signatories to the 1997 Kyoto Protocol of the United Nations Framework Convention on Climate Change (Article 4 of the UNFCCC, Article 3 of Kyoto Protocol). There is also very little guidance for REDD+ implementation in the texts of the United Nations Convention on Climate Change; only paragraphs 72 to 74 really comment on activities that countries might need to consider when implementing REDD+.

The choice of method to estimate forest carbon stocks is often governed by financial, time, data and capacity constraints. Recognising these trade-offs, the

Intergovernmental Panel for Climate Change (IPCC) present three approaches for estimating carbon stocks and emissions (IPCC, 2006). Tier 1 is based on default assumptions and default values for carbon stocks. Tier 2 employs more country-specific carbon stock information and requires activity data disaggregated to smaller scales. Tier 3 uses advanced estimation approaches that involve complex models and highly disaggregated data (Böttcher et al., 2009).

The application of Tier 1 biome and regional forest carbon averages to estimate emission reductions has become widespread where data on forest carbon stock is not available locally (Brown and Gaston, 1995, Gibbs et al., 2007). These biome averaged data are able to capture broad ecological variables influencing carbon stocks, such as temperature and rainfall (Chave et al., 2004, GOFC-GOLD, 2008), but they obscure substantial forest heterogeneity (Houghton and et al., 2001, Bradford et al., 2010). Moving from Tier 1 to Tier 3 the costs and the accuracy of emission estimates increases; discrepancies between these Tiers can be large. Brown *et al*. (2007) found that Tier 1 accounting overestimated carbon density as much as 33% in Mexican temperate forest and underestimated density as much as 44% in African rainforest. Smith (2003) found a three-fold difference in a single hectare of Zambian wilderness.

Few studies have considered the impact of this uncertainty for the environmental integrity of REDD+ which relies on real, permanent and verifiable emission reductions (UNDP, 2009). Grassi *et al.* (2008) introduce uncertainty in accounting for emission reductions from REDD+ and its implications. They explore how concepts and methodological tools can help deal with these uncertainties and promote the adoption of the conservativeness principle whereby the risk of overestimation of emission reduction is minimised. They then link this back to discussions of emission reduction accounting under the UNFCCC. Kerr *et al.* (2004)

translate errors in estimating carbon stocks into environmental integrity of credits for avoided deforestation. In their estimation of carbon storage in Costa Rica, they show that errors in emission reductions could be large, but also vary by forest type. Pelletier *et al.* (2010) use more complex modelling over time, with five carbon stock estimates for Panamanian forests in land conversion and transition models, finding 144% difference in emission reductions resulted from highest to lowest.

It can be seen that the application of broad forest carbon assumptions over large spatial scales has substantial implications. Over-estimation can lead to large sums of finance being misdirected for no climate benefit, thus threatening the environmental effectiveness of a REDD+ mechanism. Underestimation can result in lost opportunities for climate change mitigation and for the local realisation of economic incentives.

2.3. The opportunity costs of forest conservation

2.3.1. Forest income and the opportunity cost of forest conservation

In developing countries, rural communities and households can depend heavily on natural resources for their livelihoods: the capabilities, assets and activities required for a means of living (Forsyth et al., 1998, Bishop, 1999). Standing forests, in particular, provide domestic material goods and energy, enable trade and economic activity, and are a source of both food and medicines (Vedeld et al., 2004). At a household level, research into the heterogeneity of forest reliance is primarily undertaken with a sustainable livelihoods focus which refers to the assets, the activities and the access to these that determine the living gained by a household (see reviews of Godoy and Lubowski, 1992, Lampietti and Dixon, 1995, Ellis, 2000, Vedeld et al., 2004). Market-based valuation of household production is employed to value non-timber forest product (NTFP) use and to determine the relative reliance of households on forests as a livelihood-generating resource (e.g. Dercon, 1998, Shackleton and Campbell, 2001, Dovie et al., 2005).

The reliance of households on forests for their livelihoods is influenced by access to and control over forest resources (Wollenberg et al., 2000, Angelsen and Wunder, 2003). Rural poverty and forests are also found to coincide, often with forest use varying according to changes in the resource base, prices and alternative income opportunities (Neumann and Hirsch, 2000, Angelsen and Wunder, 2003). Forest-poverty links, however, are complicated. There are cause-effect issues: forest reliance due to lack of resources and alternatives differs from opportunity-driven forest reliance for valuable cash products (Adhikari, 2005). The forests are also comprised of a multitude of goods that are utilised in different ways by different groups; high value timber extraction, for example, is likely to serve a different livelihood function than NTFPs. Therefore poverty does not necessarily lead to deforestation, although it may in some cases. Studies into forest-poverty links are on-going, for example by the Poverty Environment Network instigated by the Centre for International Forestry Research (PEN-CIFOR, 2011). What is clear from the literature is that households' forest income and resulting reliance on forests is highly heterogeneous even within a small geographical area (Godoy and Lubowski, 1992, Byron and Arnold, 1999, Cavendish, 2000, Coomes et al., 2004, Dovie et al., 2005). In a meta-study of 54 cases over 17 countries, Vedeld *et al.* (2004) find mean household forest income to be US$678 per year, but with a range from US$1.3 to US$3,460.

With differing incentives for deforestation, or degradation leading to deforestation, there will also be divergence in the OCs of the land for forest conservation; the foregone benefits of an alternative investment, activity or use of the resource. Although the assessment of conservation costs is increasingly being recognised as important strategically in conservation efforts (Polasky et al., 2005, Naidoo and Adamowicz, 2006, Börner et al., 2009), the literature on the OCs of conservation interventions is sparse. Existing conservation cost assessments have been dominated by management costs (Balmford et al., 2003, Frazee et al., 2003, Moore et al., 2004). They are also focused in developed countries (Ando et al., 1998, Polasky et al., 2001, Carwardine et al., 2008) as there it can be assumed that under perfect market conditions land prices will represent the discounted stream of income from the highest-value use (Bishop, 1999).

In many developing countries land tenure is uncertain, however, and land markets absent or data incomplete thus OCs cannot be established through land markets (Balmford et al., 2000, Balmford et al., 2003, Naidoo and Adamowicz, 2006, Waggoner, 2009). Although in Brazil, where land prices do exist, Chomitz *et al.* (2005) applied the hedonic method to estimate the OCs of maintaining forest cover in the Brazilian Atlantic Forest. They found forested land prices 70% below those of cleared land, clearly demonstrating the economic incentives opposing conservation.

Where hedonic methods cannot be applied, OCs can instead be calculated through the comparison of the productivity of alternative land uses. Norton-Griffiths and Southey (1995) estimated the OCs of biodiversity conservation in Kenya at US$203 million a year by comparing the potential net returns from agriculture and livestock production within parks, reserves and forests with net returns from tourism, forestry and other conservation activities. The net revenues of US$42 million from wildlife tourism and forestry were inadequate to overcome these OCs of land use. Public willingness-to-pay and external finance are therefore critical for Kenyan biodiversity conservation. Also in Kenya, Börner *et al*. (2009) estimated the OCs of forest conservation through household surveys, at US$129-201/ha annually (applying an exchange rate of US$0.804:€1 based on the 2005 year of data collection and reported foregone revenues of €160-250/ha). They go on to demonstrate that this OC information could be used to design appropriate extraction fees to restrict resource extraction with minimal negative welfare impacts. Fisher *et al.* (2011) include both OCs of agricultural production and charcoal production within 53 districts in Tanzania finding net present value of between US$663 and US$1456/ha for agricultural production, and US$358 and US$502/ha for charcoal production.

The finer the scale at which OCs studies are undertaken the better able they are to quantify heterogeneity. In addition to household demographics, heterogeneity in OCs depends on environmental endowments such as climate regime and soil fertility, which affect land uses (Merry et al., 2002, Smith and Scherr, 2003, Nepstad et al., 2007). In Paraguay, for example, Naidoo and Adamowicz (2006) disassociate land use types and find net economic benefits of US$257/ha associated with smallholder agriculture but much higher values of cattle ranching and soybean farming at US$375/ha and US$1347/ha, respectively.

Estimates of land productivity can be used to create maps illustrating where OCs will be greatest. These maps could then be applied in conservation planning (e.g. Chomitz et al., 2005, Naidoo and Ricketts, 2006, Carwardine et al., 2010). Small-scale OCs studies also identify distributional issues for conservation policy. Where variation in OCs is large, the integrity of an intervention could be eroded where it conflicts with local subsistence demands, or if it is not politically or socially acceptable due to exacerbation of existing inequalities in wealth, income or access to resources (Shyamsundar and Kramer, 1996). Assessing OCs quantitatively and using the results in intervention design can, therefore, bring greater acceptance, longevity and impact for forest conservation (Chomitz et al., 2005, Adams et al., 2010).

2.3.2. PES in practice

The full process of PES scheme design and benefit sharing is rarely documented and Engel *et al.* (2008) note that best practice is largely confined to the grey literature. However, the existing PES literature shows that while theory appreciates heterogeneity in OCs, PES in practice largely does not. Payments are typically uniform across local stakeholders rather than differentiated (Engel et al., 2008, Southgate et al., 2009, Gross-Camp et al., 2012). Such uniform payments across the providers of environmental services are more transparent, easy to implement and give an impression of fairness for local stakeholders (Alix-Garcia et al., 2005, Pagiola and Platais, 2007, Southgate et al., 2009). But while uniform payments can generate surplus to the land owners with OCs lower than payment levels, and so increase incentives for participation in PES, they also reduce the efficiency of the intervention as payments are made that do not lead to changes in land uses (Pascual et al., 2010). Alternative payment modes exist (see Engel et al., 2008, Ferraro, 2008, Wünscher et al., 2008). For example, where information is available on local stakeholders OCs, differentiated payments can be included in contract design or through auctions or bidding systems for PES contracts (Gong et al., 2010). The application of auctions and differentiated payment, however, is often prohibited by the high transaction costs of such payment methods through data and administrative needs.

Due to complex land use and environmental service linkages, PES are also typically input-based; where land-use change is assumed to produce the environmental service rather than actual service itself (Skutsch et al., 2011). Indirect payments, as opposed to cash, have also been made including goods or services, such as clinics, schools, public transport and infrastructure (Asquith et al., 2008, Bennett, 2008). Such in-kind payments lead to fewer questions on the reinvestment of payments; cash may well lead to short-term spending on intermediary needs, for example, alcohol and luxury goods rather than on longer-term investments (Wunder, 2005, Lee and Mahanty, 2009). Both input-based and in-kind payments, however, reduce conditionality of the payment on the environmental service provision. The upfront nature of some in-kind payments also introduces a risk of whether they will sufficiently incentivise on-going service provision and they are considered irreversible in many cases as they are hard to withdraw (Sommerville et al., 2010). It may also introduce ethical issues such as in withholding community-level in-kind benefits, for example clinics, for non-participants or if contract provisions are not met. It also becomes harder for households to judge whether benefits from in-kind payments sufficiently overcome their costs. Given that providers benefit differently from the use of the common good, those with high OCs are likely to perceive low net benefit (Gong et al., 2010, Pascual et al., 2010, Sommerville et al., 2010).

In the Wunder (2005) definition of PES, providers should voluntarily enter into environmental service contracts. Alternative definitions of PES have been proposed, but overall they agree that the decision to accept a payment at the level of the transaction of the stakeholder, should be voluntary (see e.g. Sommerville et al., 2009). Economic logic would therefore predict that when offered a payment below OCs the PES scheme would not be entered into (Engel et al., 2008). However, there is evidence that in some PES schemes OCs have not been met (Corbera et al., 2007b). This may be a result of local stakeholders lacking information on the market value of services they supply or the experience to truly evaluate the contracts they are offered (Peskett and Harkin, 2007, Kosoy et al., 2008). Non-use values of standing forest are also not often factored into OC estimates and may also play a role in decision making. Gardner *et al*. (2001) found in Cameroon that despite the low income generation potential of forests locals were highly motivated to manage the forest for conservation in light of non-marketed forest values.

Furthermore, in order to opt out, stakeholders must also be free from external pressure and coercion to enter a PES scheme which is not always the case (Grieg-Gran et al., 2005, Pagiola et al., 2005a, Robertson and Wunder, 2005, Bennett, 2008). The payment contract will also pay a role in overcoming OCs, defining details such as the timing of payments, length of contracts, upfront investments required and sellers' private risk and time preferences (Ferraro, 2008). A function of the drivers of land-use change, future OCs will be influenced by changing profits to direct human activities such as agriculture and wood extraction, and affected by infrastructure development. OCs will also be impacted by the underlying drivers of deforestation, including; demographic, economic, technological, policy and institutional, and cultural causes (Geist and Lambin, 2002).

Few studies quantify how OCs of conservation might change over time. Most report OCs for a single year or assume that OCs are constant over time subject only to discounting (Börner et al., 2009, Naidoo and Adamowicz, 2006, Chomitz et al., 2005). An exception, Ferraro (2002) considered the OCs of the establishment of a national park in Madagascar. Without establishment of the park, Ferraro predicted that the flow of benefits would first increase as locals extracted resources. As these resources became degraded, however, benefits would then decline. If in contrast, the national park was established, Ferraro predicted that the benefits of exploitation were zero but, in the zone surrounding the national park, benefit flows would decrease more rapidly by virtue of a more limited area of access. While sensitivity analysis of the parameter assumptions substantially changed the estimates of total OCs, this study was useful in highlighting the impacts of unsustainable resource use on the total OCs of conservation. Pearce and Markandya (1987) also make this point in their assessment of the social OCs of natural resource management. They identified externalities resulting from reduced tree cover which could, for example, result in soil erosion, reduction in soil fertility and sedimentation, all of which all are likely to reduce agricultural yields and hence OCs over time. They concluded that where the resource use is unsustainable and where complete exhaustion of the resource base is possible, the calculation of the OCs of conservation requires information on future patterns of exploitation as well as the future development and supply of substitutes for these resources.

2.4. REDD+ via CFM

2.4.1. Community forest management

Many forests are common pool resources which are resource systems that are sufficiently large as to make it costly, but not impossible, to exclude potential beneficiaries from obtaining subtractable benefits from their use (Ostrom, 1990). In the absence of well-defined property rights, an individual actor will appropriate resource units from a common pool resource without consideration of the social cost to others. Particularly in the tropics and developing countries, policy failures such as unclear land tenure, poor governance and lack of law enforcement often lead to *de facto* open access regimes on forested land (Davies and Richards, 1999, Richards, 2008). The tragedy of the commons is predicted to result from this extraction of non-excludable and rival goods by individuals and the negative externalities of their use (Hardin, 1968).

Such an outcome, however, is not inevitable and a considerable body of research has demonstrated that individuals can collaborate to manage a common pool resource more sustainably (Ostrom, 1990, Bromley, 1992, Bardhan, 1993, Baland

and Platteau, 1996). Where users of a common pool resource are able to collaborate, the forest resource becomes excludable. Under such a common property regime, a well-defined group of people establish collective regulations for resource use, membership, monitoring and sanctioning procedures (Arnold, 2001, Baland and Platteau, 2003). This is the theoretical underpinning of CFM which creates the mechanisms and incentives such that community institutions are able to conserve forests at the same time as meeting livelihood needs (see Ostrom, 1990, Bromley, 1992, Baland and Platteau, 1996, Arnold, 2001).

The success of CFM in practice is largely demonstrated through case-studies. Literature, however, focusses on differing aspects of what might be considered a successful CFM programme and case studies equally note instances where CFM has led to uncertain livelihood and forest management outcomes. Case-studies from Nepal, where community forestry has operated since the 1980s, indicate that forest product collection rates have increased over the course of a CFM interventions, although livestock ownership decreased; the poor receive lower forest benefits than the rich and were less likely to participate in decision-making; and benefit appropriation largely depended on wealth, education and household status (Adhikari et al., 2004, Adhikari and Lovett, 2006, Adhikari et al., 2007, Adhikari and Di Falco, 2009).

CFM is now widely adopted across East and Southern Africa (Wily, 2010). In East Africa, experiences in Tanzania dominate where CFM took off in the 1990s. Case studies show that CFM can deliver improved forest outcomes in Tanzania (Blomley et al., 2008, Lund and Treue, 2008), but there has also been criticism of a lack of integration of CFM into existing local institutions (Blomley and Ramadhani, 2006) and in the equity of benefit distribution (Meshack et al., 2006, Persha and Blomley, 2009). Experience in Ethiopia is also mixed, while studies note positive impact on forest condition (Gobeze et al., 2009), others point to low participation due to low returns for locals that has led to conflict (Getahun et al., 2007). Wily (2010) emphasises the strength of the CFM approach in Africa is the recognition and empowerment of local communities as resource owner-managers, despite the uncertain forest, livelihood and governance outcomes of CFM.

2.4.2. Implementing REDD+ via CFM

Property rights are a foremost issue in PES, where property rights can be defined as the bundle of entitlements defining the owner's rights, privileges and limitations for the use of a resource (Tietenberg and Lewis, 2009). Eligibility for PES schemes often depends on an individual's right to change land use, ability to protect the service from others, and right to transfer rights (Corbera et al., 2009). Where property rights are weak it is more complex to determine who to pay, to enforce contracts, elite capture is more likely and there is likely to be weak law enforcement (Wunder, 2007, Engel and Palmer, 2008, Clements et al., 2010). The literature on PES has, therefore, largely focussed on contracts between individuals with clear legal control over environmental service provision.

REDD+ via CFM, however, would not operate through private land owners. The devolution of rights and management responsibility provides forest communities with greater long-run incentives to become good stewards of the forest resource (Agrawal and Gibson, 1999, Petersen and Sandhövel, 2001). REDD+ OCs studies have often deliberately excluded community forests. On common pool resource where forest use and deforestation is forbidden by statutory law, it has been suggested that the OCs of land may be an inappropriate measure for assessment of the feasibility of REDD+ policy as either illegal behaviours would be rewarded or emission reductions may not be additional (Börner and Wunder, 2008). It is for this reason that Wünscher *et al.* (2008), in their calculation of OCs of forest conservation, assume natural forest produces no household income. It has been suggested, where forest use is illegal, that costs incurred by the government of improving laws and law enforcement may therefore be considered more relevant in planning interventions than the requirement to overcome the OCs of forest users (Börner and Wunder, 2008, Busch et al., 2009, Gregersen et al., 2010).

A CFM intervention, however, recognises communities as forest management agents. The communities, therefore, legally become environmental service providers. The assertion that OCs will set the level of payment for land use managers to avoid deforestation may, therefore, only hold as a result of a conservation intervention implementation. Where environmental services are generated under a common property regime, however, attributing service provision to an individual is complex. All members of CFM groups have legitimate claim to payment as forest use rights are given to the community. But, not all households would deforest in the BAU baseline. Furthermore, more than one household can contribute to forest conservation on a single hectare as forest use on overlaps. Not only is it unclear which household incurred the costs of environmental service provision, the lack of attribution can also introduce free-riding and moral hazard in community-level PES; where the actions of one person are unobservable and so cheating is a distinct possibility (Hanley et al., 2006). Elite capture of community-level payments is another possibility, and well documented in the community based natural resource management literature (e.g. Fritzen, 2007, Platteau, 2004).

Transaction costs incurred by local forest stakeholders should also be considered for REDD+ under CFM. Transaction costs may include; arranging, bargaining, monitoring and enforcing agreements (North, 1990). For CFM in particular, meetings to negotiate forest areas and byelaws, in addition to ongoing monitoring and enforcement of schemes can be high for some individuals. Meshack *et al*. (2006), found poorer households took on greater transaction costs relative to their forest product benefits in Tanzania. In Nepal, richer households bore almost twice as much absolute transaction costs as poorer households in terms of the number of days contributed to CFM meetings, although costs were still a higher percentage of resource appropriation in poorer households (Adhikari and Lovett, 2006). With varying definition, each individual experiencing different costs, and hard to separate from production decisions, transaction costs are hard to estimate, however (Benham and Benham, 2000).

In the few instances where REDD+ via CFM has been considered, incentives become more 'PES-like' (Wunder, 2008). Peskett *et al.* (2008) suggest that while direct payments for REDD+ might be provided where rights are clearly established, a mixture of these and indirect benefit distribution mechanisms are preferable for REDD+ under rights regimes such as CFM. They suggest broader development projects such as improving schools and social services might be employed as incentives for REDD+. Skutsch *et al.* (2011) consider three types of payment mode for REDD+ via CFM; output-based, input-based and OCs-based. They conclude that output- or OCs-based payments to communities are not likely to be appropriate under CFM due to high transaction costs of establishing and distributing such differentiated payments. They suggest more manageable input-based incentives should be considered such as employing communities in MRV activities, or through alternative income-generating activities. These are predicted to have greater predictability of benefits, a greater focus on co-benefits rather than economic efficiency and less collusion and strategic manipulation.

Nepstad *et al.* (2007) in their assessment of costs of REDD+ in the Brazilian Amazon propose a Public Forest Stewardship Fund for avoided deforestation on 'social' forest reserves comprising 26% of the forests, including; indigenous lands, extractive reserves, and sustainable development reserves. They suggest direct payments to households, although payments are uniform and set to half the minimum salary (equating to US$1200 per year) and not linked to the delivery of emission reductions. Suggestions for the delivery of incentives for REDD+ via CFM, therefore, are more input-based, indirect or uniform all of which decrease conditionality and, thus, the efficiency that PES was initially proposed to deliver (Simpson and Sedjo, 1996, Ferraro and Simpson, 2002, Ferraro and Kiss, 2002).

2.4.3. Cooperation on a common pool resource

Historically CFM implementation has not offered communities payments, but the change in the property rights regime, leading to increases empowerment and improved governance mechanisms, is assumed to shift incentives sufficiently to deliver desired resource management outcomes (Agrawal, 2003). Collective action on a common pool resource has been shown to be influenced not only by incomes from direct extraction, but also through the impact of reputation, trust and reciprocity on households' payoffs (Ostrom, 2000, Castillo and Saysel, 2005). Collaboration on a common pool resource largely means that resource appropriators extract less than private incentives would dictate, but are willing to incur these costs for longer term sustainability on the understanding of reciprocity and cooperation of others (Heckathorn, 1993, Seabright, 1993). Game theory and experiments also indicate that as individual effort increases, the total group effort increases and incentives for free-riding decline (Fischbacher et al., 2001, Castillo and Saysel, 2005).

Research from a large body of case studies on common pool resources has converged on a set of variables that enhance the likelihood of cooperation. These can be divided into the attributes of the resource and the attributes of the appropriators. In the first instance, collaboration is enabled by: the feasible improvement in the resource as a result of collaboration; reliable and valid indicators of condition; predictability of resource units; and, a sufficiently small spatial extent that knowledge of boundaries and microenvironments are known by the appropriators. In the second instance, collaboration of appropriators is enabled by: dependency on the resource base for a major portion of their livelihood; a common understanding of how use affects that of others; a discount rate that allows future benefits to be achieved from the resource; similarly affected interests of appropriators despite economic and political asset heterogeneity; trust and reciprocity; autonomy to determine access and harvest rules from external authorities; and, local leadership and organisational experience (see Baland and Platteau, 1996, Agrawal, 2001). The enabling factors of cooperation all impact upon the balance of costs and the benefits, or the payoffs, both perceived and experienced by the resource appropriators (Matta and Alavalapati, 2006). The payoff determine whether households will cooperate in collective resource management (Ostrom, 1990, Varughese and Ostrom, 2001).

PES so far has failed to consider the logic of collective action (Kosoy et al., 2008, Muradian et al., 2010). Kosoy *et al.* (2008) is a rare study of willingness to participate in PES that accounts for rules, institutions, values and interactions between actors and, it considers PES on a common-property regime. Analysing Mexican communities receiving payments for biodiversity and carbon under the government initiated Payments for Hydrological Environmental Services Programme, they find that procedural rules and management impact on participation, but also note that collective motivation can be distinctly different from individual preferences. Estimates of the OCs of REDD+ commonly assume households act as self-interested profit maximisers that act independently of their impacts on others (e.g. Busch et al., 2009). They may, therefore, overestimate the payment required to incentivise forest conservation.

Fisher *et al.* (2010) notes that PES in developing countries often operate under conditions much like a common pool resource; with unclear property rights, poor monitoring capacity and information asymmetry. The transfer of formal land tenure to local land managers has been used as a PES incentive in South-East Asia under the Rewarding the Upland Poor for Environmental Services (RUPES) project (van Noordwijk et al., 2004). The importance of social capital and social empowerment is also shown to be important in PES. Gong *et al.* (2010) show that areas of low uptake of a Clean Development Mechanism (CDM) forest project in China, are those in which social capital is also low due to its impact on the ability to enforce contracts through social structures. It is clear that the lessons and incentives for cooperation in common pool resource management are important for PES design and implementation.

2.5. Conclusion

Despite the growing support for REDD+ via CFM, it is unclear how REDD+ as a local-level PES could be implemented on-the-ground. PES in practice has so far failed to appreciate the heterogeneity of OCs of land and there has been little consideration of how OCs will change over time. There has also been limited consideration of how PES will operate on a common property regime, in particular how financial incentives of a PES scheme will be impacted by the non-financial incentives so far provided under CFM. Ongoing discussions to ensure that finance for emission reductions through REDD+ is direct, conditional, additional, and permanent, contrast the proposals for REDD+ via CFM implementation, and indeed PES in practice, which become more indirect and less conditional on emission reduction delivery. If current levels of public and private interest in forest conservation through REDD+, and REDD+ via CFM, are to be maintained and expectations are to be met, this divergence in discourse and practice needs to be addressed.

Chapter 3: Case study site

3.1. Introduction

Sustainable forest management in Ethiopia has been hindered by political instability and a country-wide focus on increasing food production and security. In 2005, the Woody Biomass Inventory and Strategic Planning Project (WBISPP) reported that 13 million ha of forest remained in Ethiopia covering 12% of Ethiopia's land mass. Country-wide forest losses of 140,000 hectares each year are driven by conversion to agriculture, and unsustainable forest management, underpinned by poor governance, uncertain land tenure and a rapidly growing population. High levels of poverty characterise the country, 78% of Ethiopia's population live on less than US$ 2 per day and GDP per capita was reported as US$221 in 2010 (WDI, 2011). Forest conservation that can also meet livelihood and development needs in Ethiopia is therefore necessary.

In the Bale Mountains Eco-Region (BME), deforestation rates are four times the country-wide average. A forest conservation intervention, referred to in this thesis as the Bale REDD+ Project, is underway to devolve management responsibilities to communities while also generating emission reductions through avoided deforestation. The Bale REDD+ Project that achieves REDD+ via CFM could be exemplary for the proposed scaling up of CFM across the country, as well as for a growing number of REDD+ projects in development in Ethiopia. This Chapter introduces the history of forest policy in Ethiopia and the proposed forest conservation intervention in the BME on which this thesis is based.

3.2. Forest Management in Ethiopia

3.2.1. Ethiopia in context

Ethiopia is divided into nine administrative regional states: Afar, Amhara, Benishangul-Gumuz, Gambela, Harari, Oromia, Somali, the Southern Nations Nationalities and Peoples Region, and Tigray. Ethiopia's administrative regional states are subdivided into *zones* of which there are a total of 68. The most recent census reports Ethiopia's population at 74 million across a land area of 1,221,900 km^2. Ethiopia is Sub-Saharan Africa's second most populous nation with 84% of the population living in rural areas (International Monetary Fund, 2007).

In 2010, a new economic plan for Ethiopia was proposed that focusses on infrastructure, industrialisation, large-scale commercial farming, boosting the role of small private enterprises, and improving economic governance (EIU, 2010). This presents possible risks of land grabs and big commercial farms may threaten the retention of forested areas. Since 2000, a loss of 140,000 ha of forest annually, or 1.1% has been reported (WBISPP, 2005). Recent exploration of the main drivers of deforestation and forest degradation in Ethiopia identify the small scale conversion to agriculture, large scale conversion to agriculture, and unsustainable forest management (R-PP, 2011).

Any forest conservation efforts in Ethiopia must be managed alongside development plans. The country ranks low, at 174 of 187 countries on the Human Development Index in 2011. With key links between human wellbeing and the maintenance of ecosystem goods and services being made in Ethiopia's recent environmental policy, the renewed positive attitude to natural resource conservation, and an emerging participatory approach to management, could prove profoundly helpful for meeting both development and poverty reduction goals.

3.2.2. *Forest management: past to present*

Ethiopia's forests were historically under traditional management practices throughout the 19th Century. The *Gada* system, for example, divided society into age classes, the peak of which males entered the *Gada* council for a period of eight years. These elders were responsible for day-to-day jurisdiction as well as reiteration and introduction of the locally agreed rules and norms of resource use (Wakijira et al., *in press*).

In the 20th Century, under Menelik resource management was centralised and in the 1940s, Emperor Haile Selassie privatised land. This limited people's access to forests and eroded traditional forest management practices as elders' functions were to promote central policies rather than maintain and adapt local informal institutions. To protect Ethiopia's biological diversity, however, the Ethiopian Wildlife Conservation Organisation was founded in 1964 to form a network of protected areas.

Overthrowing Haile Selassie in 1975, the Derg socialist military regime, or Provisional Military Administrative Council, came into power. Forest management was further centralised but land ownership was nationalised (Mekonnen, 2000). This made all forest use prohibited, further eroding local institutions for forest management (Wakijira et al., *in press*). Village organisations were formed that brought together the general assembly of household heads in the village and formed an executive committee and judicial tribunal. Again, these were in place to implement directives, decisions and orders that came from higher officials and central government, rather than to continue local resource management institutions. By this time the traditional management systems in Ethiopia had been eroded.

The central government did form state owned Forest Priority Areas, National Parks, Game Reserves, Sanctuaries and Controlled Hunting Areas. However, these were poorly implemented. Forest Priority Areas established by the government were largely nominal and forests were perceived to be for exploitation rather than protection. Of 58 Forest Priority Areas only 48 were demarcated, 5 inventoried, 4 had management plans, and none were legally constituted (or gazetted) (Teketay et al., 2010). Furthermore, only two of the nine National Parks and three wildlife sanctuaries have ever been gazetted.

Encroachment into forest areas for informal and uncoordinated resource use has been experienced across Ethiopia as a result (Macqueen, 2008). Rebel force occupation of the forests and the protracted civil war and political instability in Ethiopia also contributed to the degradation of many forest areas. Displaced communities were known to settle in a number of Ethiopia's neglected National Parks.

The defeat of the Derg in 1991 by the Ethiopian People's Revolutionary Democratic Front (EPRDF) ended political suppression and initiated extensive economic reform within Ethiopia. The economic reform was largely focused towards poverty alleviation through efforts to increase the productivity and efficiency of agriculture (Abrar et al., 2004). With countrywide issues of food security and land scarcity, there was, and still remains, clear justification for policies encouraging agricultural intensification in Ethiopia (Byerlee et al., 2007, Diao and Pratt, 2007). Agricultural output contributes 50% of GDP and 90% of output arises from smallholder farming on micro-holdings (Shiferaw and Holden, 1999, Shiferaw and Holden, 2000).

The success of agricultural intensification efforts so far, however, is questionable. The ongoing public investment and provision of technology for agricultural intensification has not led to higher or more sustainable cereal yields, reduced food aid dependency, improved food security or lower prices for staple crops (Byerlee et al., 2007, Spielman et al., 2010). Grain production in Ethiopia did grow by 74% between 1989/90 and 2003/04, but cultivated area increased by 51% (Gebreselassie, 2006). These productivity gains, therefore, have been attributed to the expansion of agricultural land rather than successful agricultural intensification (Byerlee et al., 2007, Diao and Pratt, 2007).

The continued investment in agricultural intensification may have come at a cost to natural forests, however. With no dedicated central government forest ministry, the Ministry of Agriculture and Rural Development is responsible for the formulation of forest resource relevant policies, laws and for the provision of technical support to the Bureaus of Agriculture and Rural Development in each of Ethiopia's regional states. At the local-level, Agricultural and Development Agents have focussed on their responsibilities for agricultural development activities and given less attention to natural forest conservation. The substantial annual forest losses and the unsustainable exploitation of Ethiopia's forests threaten the livelihood security of the rural population. The WBISPP indicated that 70% of *woredas* consume wood products faster than they can be replaced (WBISPP, 2005). Furthermore, Ethiopia's population is growing rapidly at 2.6% (FDRE, 2008).

Federal government's current attitude to forest conservation and natural resource management has been more promising since sever forest fires of 2000 (Wakijira et al., *in press*). Several environmental initiatives have been adopted. In 2005, a new Wildlife Development Conservation and Utilisation Policy and Strategy was accepted, uniting previously unrelated policies for wildlife, biodiversity and environmental protection. It also highlights key links between human wellbeing and the maintenance of ecosystem goods and services; it supports environmental valuation approaches as well as PES.

In 2007, the government issued a proclamation for Forest Development, Conservation and Utilisation (542/2007). This proclamation divides forest into state and private ownership, but under both, makes provision to engage communities in forest management (Moges et al., 2010). It is under the guidance of this federal policy and proclamation, in combination with the Environment Policy of Ethiopia and the Conservation Strategies of Ethiopia, that regional states then administer Ethiopia's forest resources.

3.2.3. Community forest management in Ethiopia

The deforestation and degradation of Ethiopia's forests is exacerbated by total government ownership of land: the common property of the state and the people, land shall not be subject to sale or exchange (Amente and Tadesse, 2004). This has prevented a mass rural-urban migration, where infrastructure is not sufficient to support an influx of people. But the uncertainty of tenure has generated disincentives for the rural population to maintain ecosystem quality or for farmers to invest in productivity improvements. The state forest authorities also lack resources to sustainably manage the forests (Amente and Tadesse, 2004). Thus while forests are legally owned by the government, they are utilised by local communities with a lack of law enforcement and many of Ethiopia's forests have characteristics of an open access regime on a common pool resource.

Although issues of property and use rights of land and forests remain, there is strong support for CFM across Ethiopia. CFM involves the legal transfer of forest use rights from the government to community-based organisations (CBOs) - the small groups of households that sign forest use agreements – enabled by and dependent upon a negotiated Forest Management Agreement outlining forest management plans and the implementation of sustainable forest management practices.

The policy and legal framework of CFM in Ethiopia is driven predominantly by the 2007 proclamation for Forest Development, Conservation and Utilisation (542/2007), the Environment Policy of Ethiopia and the Conservation Strategies of Ethiopia also play a role. Of course, CFM is not the only forest conservation measure that Ethiopia is pursuing. The protected area system is still in existence and the Ministry of Agriculture and Rural Development is currently implementing a national level Protected Area System Plan (PASP).

The CFM approach in Ethiopia has been employed for more than a decade in both Oromia and the Southern Nations Nationalities and Peoples Region. Efforts have been largely driven and supported by NGOs: FARM Africa with SOS Sahel, and the German Technical Cooperation (GIZ). CFM is now supported at the national level and a country-wide CFM programme is being scaled-up. This requires substantial finance, some of which is being provided by the European Development Fund (R-PP, 2011). In 2009, the Strengthening Sustainable Livelihoods and Forest Management Programme was commenced in four regional states of Ethiopia with a vision to see government authorities incorporating CFM in annual plans, budgets and management structures (SSLFM, 2010).

CFM is supported in Ethiopia despite weak evidence on its long-term effectiveness. In 2001, FARM-Africa worked to implement CFM in Bonga Priority State Forest of the Kafa zone of the SNNPR. A moist tropical forest, implementation of CFM appears to have positive impacts on the state of the forest and living conditions within the project lifetime, but continuation of CFM appears threatened by weak government support for the scheme after the NGO support was terminated (Gobeze et al., 2009).

In Oromia, three CFM areas exist in the forests of Chilimo, Borena and Adaba-Dodola. Chilimo, in the West Shewa zone of Oromia, is a highland montane forest where FARM Africa initiated a pilot CFM project in 1996, although it was not until 2004 that the first forest user group was established. It is believed that CFM has improved people-forest relationships with reduced deforestation, increased regeneration and the empowerment of locals. However, in a largely qualitative exploration of the intervention, Kassa *et al.* (2009) suggest that the technical, managerial and administrative capacity of the CBOs need to be strengthened and efforts to diversify livelihood options are still needed to reduce human pressures on the forest. In Borena, CFM implementation has proved more challenging. A lowland Juniper forest in the Borena and Guji zones of Oromia, where livelihoods are more pastoral, forest based enterprises are producing low returns for farmers and land conflicts have arisen (Getahun et al., 2007).

The Integrated Forest Management Project Adaba-Dodola, a project of both the government of Ethiopia and GIZ, was implemented by the Oromia Rural Land and Natural Resources Administration Authority in June 1995. Located within the BME, plans to scale up CFM across the region will build on the lessons learnt in Adaba-Dodola. The goal of the project was to establish Forest Dwellers Associations, or *Waldaa Jiraatotaa Bosonaa* (WAJIB) in Oromo, where members protect the forest and carry out management activities and restrict their expansion of farm plots in return for rights to live in the forest and generate forest-based benefits. Forest blocks constituted 300 to 500 ha and not more than 30 households, based on a forest carrying capacity of 12 ha per household established from previous CFM experience (Kubsa and Tadesse, 2002, SUN-Dodola, 2005). A functioning WAJIB consists of a general assembly, an executive committee and various other committees elected by members. Each WAJIB group has its own by-laws (internal regulations), that govern use, protection, rights and responsibilities of each household within the forest block. The forest administration is providing mostly technical advice on the development and sustainable utilisation of forests. Positive impacts of this CFM effort, to date, have been the improved forest condition and management. Rural livelihoods and social welfare are also reported to have improved, although not quantitatively (Kubsa and Tadesse, 2002, Tesfaye et al., 2011).

In spite of a lack of evidence in Ethiopia and more broadly in Africa, CFM approaches have been adopted across East and Southern Africa (Wily, 2000). In Tanzania, for example, the 1998 Forest Policy made a commitment to bring more forest and woodlands into village forest reserves. In 2010 it was reported that since 2005 more than 500 village forest reserves were declared by communities from communal lands (Wily, 2010). Also in her 2010 review, Wily notes that such management approaches are sufficiently widespread in Africa to be recognised as a route to securing and sustaining forests. The review also indicates how the concept has evolved to recognise that forest management is a matter of governance and, increasingly targeted at the grassroots level, the empowerment of local communities as owner-managers through devolution of responsibilities has been important.

3.2.4. Forestry Carbon in Ethiopia

Efforts to establish REDD+ projects and activities have often focussed on countries where forest areas are more substantial and the carbon contained within the forests is very high. This includes Brazil, Indonesia and the Democratic Republic of the Congo where the majority of international finance to support REDD+ development has been channelled (Climate Funds Update 2012). Establishing REDD+ in Ethiopia, therefore, may not contribute significantly to reducing emissions from deforestation assessed at an international scale. Ethiopia may not receive as substantial financial transfers as other tropical forested nations under an international REDD+ mechanism established by climate change negotiations. REDD+ does, however, contribute to internalising the externality of climate regulation. It could provide a source of finance that changes the economic incentives to make forest conservation more economically viable and it necessitates the discussion and review of property rights regimes in forested areas.

Signatory to the United Nations Framework Convention on Climate Change (UNFCCC), and the Kyoto Protocol, political and public awareness of climate change issues is increasing rapidly in Ethiopia. This can be partly attributed to the presence of Prime Minister Meles Zenawi at the United Nations Climate talks and national media campaigns up until his death in 2012.

Ethiopia's growing interest in REDD+ also stems from a number of organisations, NGOs in particular, which have begun to explore the potential for such forest carbon projects. The Humbo Community-Based Natural Regeneration Project, developed by World Vision Ethiopia and Australia, was the first forest carbon project in Ethiopia. An afforestation/reforestation project covering 2,728 ha in the southwest of Ethiopia, the project aim was to restore indigenous forest species to the land. In 2009, the Humbo project was registered under the CDM of the Kyoto Protocol and the World Bank Bio Carbon Fund has purchased the emission reductions generated by the project (FCPF, 2011). Following the success of this project, four further CDM projects are under development (R-PP, 2011). The development of avoided deforestation and degradation activities in Ethiopia has also taken off, although no REDD+ projects are yet certified and generating emission reductions for sale. NGOs instrumental in driving REDD+ in Ethiopia include Farm Africa, SOS-Sahel, World Vision Australia, and Save the Children US.

Ethiopia is also a member country of the World Bank's Forest Carbon Partnership Facility (FCPF). A multilateral REDD+ initiative, the FCPF builds capacity for REDD+ and tests a programme of incentive based payments through grants to its 37 member countries (FCPF, 2011). In 2011, a revised Readiness Preparation Proposal (R-PP) outlining a national REDD+ strategy for Ethiopia was formulated. Financing to implement the R-PP was estimated at US$12,495,000 with a timeline of completion in 2014. During the R-PP preparation a number of workshops and consultations were carried out. In-country capacity is building for REDD+ and activities of the RPP are already in progress. In November 2012, US$ 3,400,000 was approved for the R-PP.

With REDD+ activities in their infancy, the legal and institutional setting in Ethiopia is uncertain. The Environmental Protection Authority of Ethiopia is currently chairing the REDD+ process in Ethiopia with a REDD+ steering committee and REDD+ technical working group also established. The Environmental Protection Authority will hand over to a federal agency dedicated to forestry once it is created. Plans exist to develop regional steering committees and technical at REDD+ sites. More on the legal and institutional setting of REDD+

in Ethiopia is expected as the R-PP grant progresses through its three phases, with the preparatory phases spanning the next four years.

Ethiopia can stand to learn from other countries in the region and their experiences with REDD+. The drivers of deforestation in Ethiopia are similar to those in other East African countries such as Kenya, Tanzania and Uganda. In all of these countries efforts are underway to build national REDD+ capacity and REDD+ projects. Tanzania in particular, with 40% forest cover, has commanded a lot of attention and US$ 131 million has been approved for REDD+ activities through dedicated public climate funds (Climate Funds Update, 2012).

In Ethiopia's national REDD+ strategy, it is acknowledged that substantial work is to be done. In particular, a national forest inventory with a view to determine carbon stocks and a deforestation baseline is required. To date detailed measurement on Ethiopia's vegetation coverage, and changes in this cover over time, are largely inadequate with conflicting information and no regular inventories (Teketay et al., 2010).

With 100% publically owned forest, REDD+ in Ethiopia will require clarification of forest use and carbon rights and substantial engagement and participation of the 84% of the population that resides in rural areas. Governance is also important for investors and Ethiopia ranks low in the World Bank Governance Indicators. For political stability and absence of violence Ethiopia has a score of -1.71 in 2010, where country scores range between -2.5 to 2.5 and higher values correspond to better governance. For government effectiveness Ethiopia ranks -0.35, for rule of law -0.76, and for control of corruption -0.70 (WGI, 2010).

Without a national level forestry institution, designing and managing REDD+ financial structures and benefit sharing mechanisms may also prove challenging. Although the R-PP does mention that a body that bypasses '*normal administrative budgetary functions'* will be established that can ensure dispersal of REDD+ funds to the local-level (R-PP, 2011). Ethiopia's R-PP also highlights the excessive expectations that exist for REDD+ activities to address issues of deforestation and forest degradation as well as reducing poverty in the country. Ethiopia's R-PP, however, is highly supportive of pursuing REDD+ through community forestry. It is integral as a source of funding for community forestry as well as community forestry as a way to reduce deforestation.

3.3. The Bale Mountains Eco-Region

3.3.1. The south eastern Ethiopian highlands

The BME forms part of the Bale-Arsi massif in the south eastern Ethiopian Highlands (Figure 2). Although it is named an eco-region by local implementers, the BME is not a WWF eco-region, which is defined as a large unit of land or water containing a geographically distinct assemblage of species, natural communities and environmental conditions. It is referred to in this thesis as an eco-region, however, to be consistent with the Bale REDD+ Project implementers at the case study site as well as the national use of the term to refer to this area.

The BME falls within the Oromia regional state, the most populous province in Ethiopia with a population of 27,029,760 in 2007 (FDRE, 2008). 70% of Ethiopia's remaining forest is in Oromia (Macqueen, 2008). The Bale zone is found between 50˚22'-80˚08'N and 38˚41-40˚44'E. Zones are further divided into *woredas*, or districts, that are managed by a local government of which there are around 550. The BME within the Bale zone, covers 2,217,600 ha over fourteen woredas: Adaba, Agarfa, Berbere, Dinsho, Dodola, Gasera, Goba, Gololcha, Goro, Harenna Bulluk, Kokosa, Mena, Nensebo and Sinana. These woredas are composed of *kebeles*, or villages, which are the smallest local government unit (Figure 3).

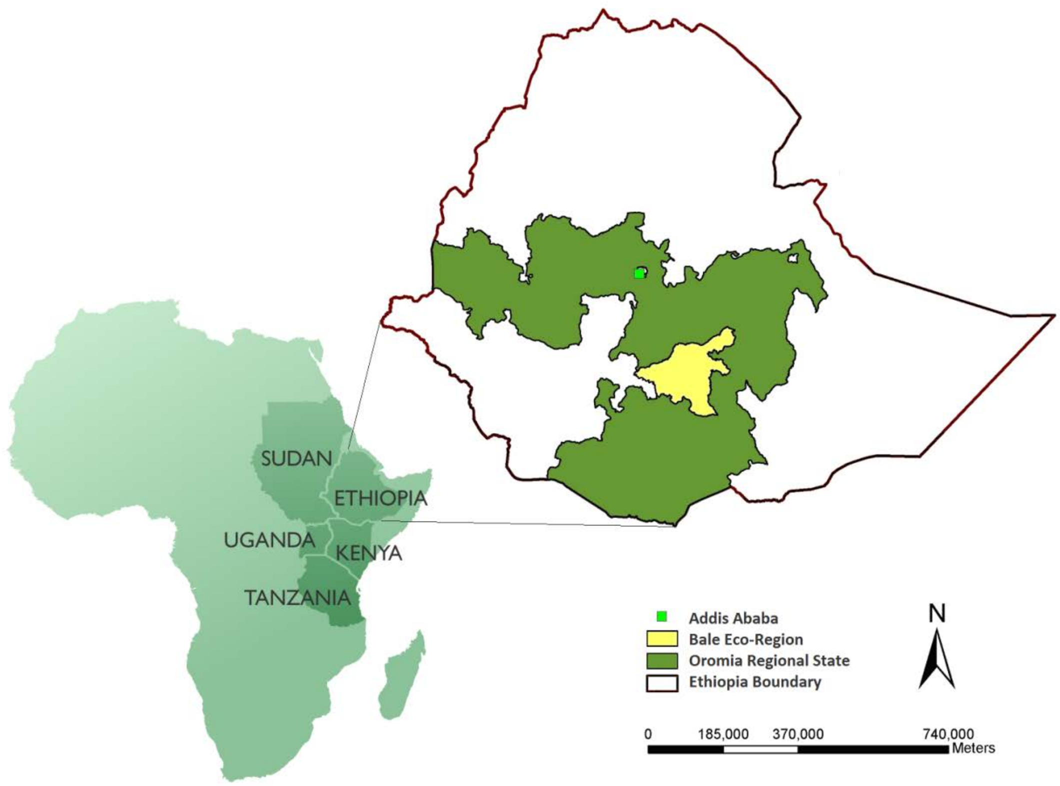

Figure 2. Map of Ethiopia and the Bale Mountains Eco-Region.
Located in Oromia regional state, the Bale Mountains Eco-Region (BME) lies 400km south east of Addis Ababa, the capital of the Federal Democratic Republic of Ethiopia a land-locked nation in the horn of Africa bordered by Eritrea to the north, Somalia and Kenya to the south and Sudan to the west. *Source: author generated*

3.3.2. Ecological context

The annual temperature of the Bale zone is 17.5˚C ranging from 10˚C to 25˚C, with annual rainfall of 875mm experienced in one long season between June and October, and one short rainy season between March and May (Yimer et al., 2006). This range obscures the substantial topographic variation which characterises the vegetation in the BME (Figure 3). Distinctive endemic flora and fauna of the Bale

Mountains result from its isolation from the bulk of the Ethiopian highlands and its topography and climatic history (Hillman, 1986, Yalden and Largen, 1992).

The afro-alpine plateau of the central area of the BME reaches more than 4000 metres above sea level (masl). Containing *Erica*, Giant lobelia (*Lobelia rinchopatelum*) and *Helichrysum*, this is the largest remaining area of Afroalpine habitat on the African continent (BMNP, 2007). South of the plateau the altitude falls rapidly with moist tropical forest between 2600 masl and 1500 masl. The moist forest is characterised by *Hagenia abyssinica* and wild coffee (*Coffea arabica*). Lions and African wild dogs are also still found in this forest which is the second largest stand of moist tropical forest in Ethiopia. North of the plateau habitats comprise of dry forest, woodlands, grasslands and wetlands, largely between 2500 masl and 3500 masl. The dry forests contain high-value commercial species such as *Juniperus procera* and *Podocarpus falcatus* as well as *Prunus africanus,* a threatened species. The lower altitude land of the south east of the BME, below 1500 masl, is dominated by acacia woodland (Teshome et al., 2011, UNIQUE, 2008).

The BME is part of one of 34 global biodiversity hotspots which contain more than 1,500 species of vascular plants as endemics and it has to have lost at least 70% of its original habitat; it falls within the Eastern Afro-Montane biodiversity hotspot (Myers et al., 2000, Conservation International, 2012). This ranges from Saudi Arabia and Yemen to Zimbabwe, taking in a number of mountain ranges. The habitats of the BME host a rare and endemic species including the Ethiopian wolf (*Canis simensis*), Mountain Nyala (*Tragelaphus buxtoni*), and the Giant mole rat (*Tachyoryctes macrocephalus*). This ecological importance was acknowledged by the establishment of the Bale Mountains National Park (BMNP) in 1971, which lies at the heart of the BME. The (proposed) BMNP is stated to be one of the most important conservation areas in Ethiopia (FDRE, 2005). The 220,000 ha park was actively managed until 1991 but resources within the park boundary, particularly forests, are currently being used unsustainably.

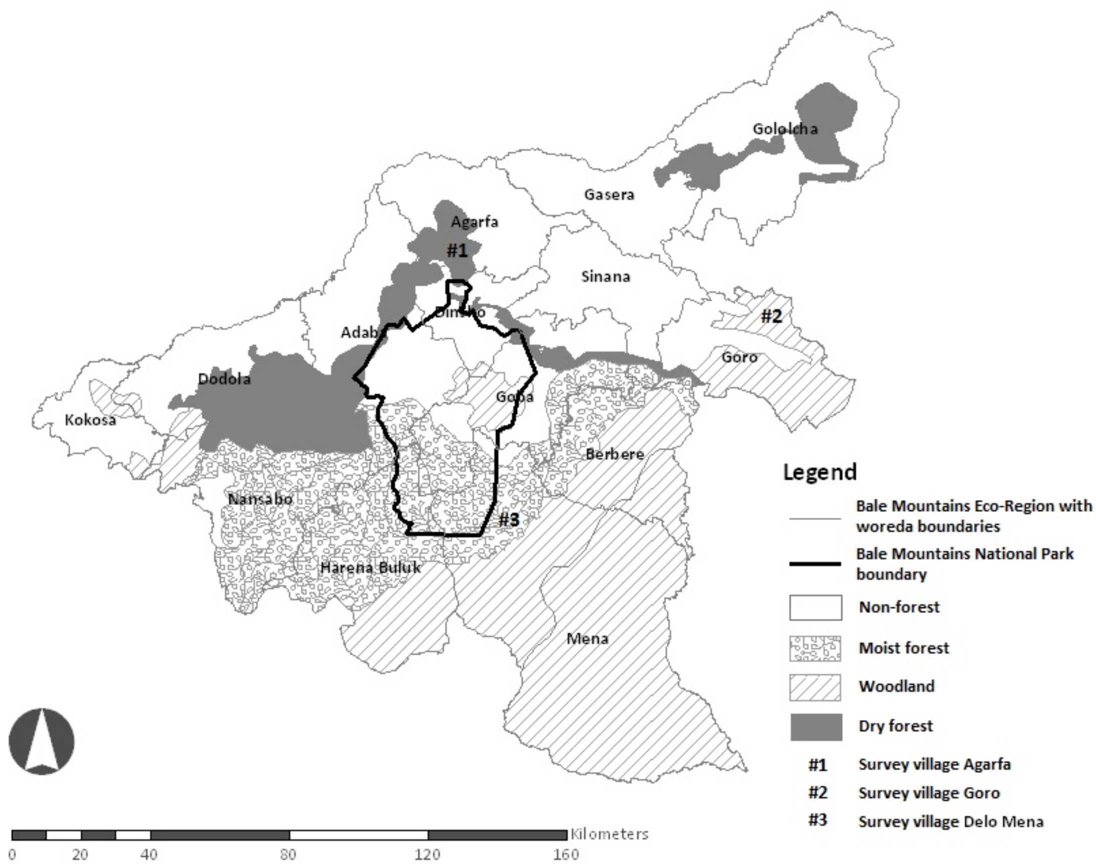

Figure 3. Forests of the Bale Mountains Eco-Region.
The *woredas*, or districts, of the Bale Mountains Eco-Region (BME) vary widely in their forest cover, with forest divided into broad categories of moist forest, woodland and dry forest. The (proposed) Bale Mountains National Park lies at the centre of the BME, and the three survey locations are distributed across the BME. *Source: author generated*

3.3.3. Forest use in the Bale Mountains

The dominant livelihood strategy in the BME, as in wider Ethiopia, is small-scale farming using traditional technologies for low input, low output rain-fed mixed farming (World Bank, 2007, Rosell, 2011). Households cultivate crops on distinct land plots. Most commonly cultivated are cereal crops including Maize, Teff, Wheat, Barley, and Sorghum. Households also engage in livestock rearing for meat and milk products, manure, draught power, transport and skins. Livestock also play a role in marriage, dispute settlement and ritual performances (BMDC, 2003). Rural households gather many products from the forest and where valued can make up a significant portions of their income.

Under a total environmental value framework, the forest produces a variety of direct, indirect, option and non-use values (Pearce and Warford, 1993). Direct use values that more tangibly contribute to household income include: NTFP such as honey, coffee, medicinal plants and fuelwood; timber and construction products; recreation; and livestock grazing lands. Tesfaye *et al.* (2011) estimated such forest incomes contribute to 34% of per capita income in the BME. This aligns with other research on forest income reliance such as Babulo *et al.* (2009) who find households derive 27% of income from forests in northern Ethiopia, and Mamo *et al.* (2007) who find 39% of incomes are derived from forest in central Ethiopia. A lack of employment opportunities restricts the diversification of livelihoods in the BME, thus crops, forest and livestock are the three main livelihood sources.

Indirect use values accruing to households include carbon sequestration and watershed protection. The Bale Mountains have been described as a water tower and the hydrological system supplies water to an estimated 12 million people in the lowlands of south eastern Ethiopia, northern Kenya and Somalia (BMNP, 2007). Option values include pharmaceuticals and the genetic library of biodiversity. Arabica coffee, for example, has its origin in Ethiopia where it occurs naturally and so the diverse gene pools of wild coffee populations have potential options for new coffee varieties (Schmitt et al., 2009). Non-use values include cultural values placed on forests, values held for endemic species, and landscape beauty. Burial sites, for example, exist in the forest and deforestation is regulated in these small areas by local communities.

The forest use in the BME, however, is unsustainable as across wider Ethiopia. There is rapid deforestation to procure land for crops and livestock grazing and to meet livelihood needs through timber and firewood extraction (BERSMP, 2006, BMNP, 2007). The lack of human and financial resources, political interest and technical knowledge, combined with population growth and immigration to the area also contribute to forest losses (BMNP, 2007). Between 2001 and 2009 the average annual deforestation rate in the BME exceeded the countrywide rate of forest loss. Average deforestation rates in the BME were 3.44%, ranging from 1 to 8% (Dupuy, 2009). There is evidence that this rate is accelerating, particularly in the moist forest of the (s) BMNP where deforestation rates have increased from 1.64% in 1973-2000 to 15.0% between 2000 and 2006 (Teshome et al., 2011).

3.4. The 'Bale REDD+ Project': REDD+ via Community Forest Management in the Bale Mountains

3.4.1. Project outline

To address the decline in forest area, the Oromia Forest and Wildlife Enterprise (OFWE) are implementing CFM across all forests of the BME. The intention is to generate REDD+ as a result of CFM implementation. While CFM and REDD+ can both be undertaken as separate policy interventions, in the BME these are therefore considered together: the Bale REDD+ Project undertakes REDD+ via CFM. Thus emission reductions do not have to be additional to that achieved through CFM, but rather are those generated by CFM.

Covering more than 900,000 ha, the proposed Bale REDD+ Project area consists of the dry and moist tropical forest as well as the southern woodlands of the BME. In 2008, a report on carbon finance in the BME was undertaken by external forestry consultants identifying good opportunities for REDD+. The Bale REDD+ Project builds on this pre-feasibility study and aims to gradually reduce deforestation below the BAU baseline of 4% per annum to 1% by project-year 20. In order to achieve these emission reductions, CFM will create a common property regime in the BME.

Under the Bale REDD+ Project, households in the BME will experience a change in forest access from a *de facto* open access regime to a *de jure* common property regime. To do so, a set of identifiable forest users who hold the resource and that can exclude others and regulate use will be formed as a CBO group. It is proposed, that forest blocks of 300 to 500 hectares are allocated to not more than 30 member households. Entry into the CBO groups will be controlled. Eligibility for membership relies only on the fact that you live in the Kebele, and entry is voluntary. In order for user groups to be a legal entity under Ethiopian law there is a nominal registration fee in the region of ETB5. These groups will be created without assessment of the carrying capacity of the forest, but will rely on adaptive management to revise the management plan every three years to ensure forest use becomes more sustainable over time.

The rights and duties of households under CFM will be formalised in contracts signed between CBOs and the forest agency. Rights of the CBO include settlement and grazing, maintaining existing farm plots and using forest products for consumption and sale. Thus while they will be given use rights – in contrast to the status quo where forest use is not allowed – they will not be given land rights.

Aside from the secured use rights, the exact conditions and managerial responsibilities of communities will be defined in a participatory manner with the communities in question. These byelaws agreed by the communities are likely to include restrictions on further settlement and agricultural expansion and for initial forest cover to be maintained. Fuelwood use will also be determined in the byelaws; for example, where CFM has progressed in the region the number of days per week that dead wood can be collected for fuel is now limited. Periodic forest cover assessments and settlement censuses will therefore be agreed by the CBO and Forest Agency. The Forest Agency is expected to safeguard CBO groups against free-riders and enforce sanctions in the case of non-compliance.

The specific roles of CBO members will also be determined under the byelaws, but members will be required to work free of charge. This will mean that households incur transaction costs of CFM. Transaction costs of CFM include through meetings, such as for the arrangement and negotiation of forest areas and byelaws, as well as monitoring and enforcement. This has and is occurring in the community to manage other communal resources. For example, Oromo pastoralists use mineral springs (horas) for their livestock (cattle, sheep and goats) as they are perceived to enhance fat, fertility and resistance to diseases of livestock. Horas are maintained by the communities that use them most frequently for free, this includes establishing and maintaining fencing as well as cleaning of excess mud (Chiodi and Pinard, 2011).

The core CBO committee will meet regularly and will be required to patrol the forest in crucial times, such as harvesting season for forest coffee. Where byelaws are broken, individuals must appear in front of the elders committee to be sanctioned. Only repeat and serious offenders will be sent to Woreda level for sanctioning. Pro-poor provision can also be designed by the CBO group themselves. There is past experience of such provisions in the traditional forest management systems in the Bale Region. For example, under the *Gada* system contributions of forest coffee beans from the moist forests were collected and redistributed to families unable to collect their own as a result of illness, physical disability or old age (Wakijira et al., *in press*).

BERSMP are also undertaking efforts to sustainably increase agricultural production, establish woodlots, promote fuel-efficient stoves and biomass briquettes, improve forest fire management, and add value to forest products (BERSMP, 2006). Measures that substitute for fuelwood demand are critical in order to address the drivers of deforestation in the BME. Progress towards the establishment of woodlots led by OFWE has been progressing slowly, however. Such woodlots are unlikely to take less than 3 years to be established, and there is uncertainty over community contributions, such as labour, will be required. There has been more success with energy efficiency measures; with fuel-efficient stove distribution widespread. BERSMP is also supporting home planting in backyards and group woodlots to try to meet needs and buffer plantations are under consideration.

The Bale REDD+ Project is still in early stages with regards to REDD+ development; a Project Design Document is underway. As a result, no further decisions have been taken on the shares of carbon revenues to stakeholders, including communities. To date, the costs of REDD+ project development and capacity building for REDD+ have been absorbed by BERSMP. A trust fund handling monetary aspects of the ERPA supervised by a board including NGO, CBO and state institution members has been proposed (UNIQUE, 2008).

3.4.2. Bale REDD+ Project implementers

The roll-out and scaling up of CFM across the BME is supported by the Bale Eco-Region Sustainable Management Programme (BERSMP). Initiated in 2007, BERSMP is an operational partnership between the Government of Ethiopia (Oromia Regional Government, Bureau of Agriculture and Rural Development, and the Food Security and Disaster Prevention and Preparedness Commission) and NGOs FARM-Africa and SOS Sahel Ethiopia. It is these organisations that are referred to as the Bale REDD+ *Project implementers* in this thesis.

BERSMP has a distinct goal to mutually and sustainably enhance the unique biodiversity and ecological processes of the BME and the social and economic wellbeing of the communities dependent on the natural resources. Six programme outputs to achieve this are: an Eco-Region plan, building government and community capacity for sustainable natural resource management; functional and sustainable natural resource management and conservation systems, incorporating environment and community needs; diversification of community natural resource based livelihoods; sustainable financing mechanisms that benefit government and communities; and, improved legal, policy and regulatory frameworks (BERSMP, 2006).

OFWE, a semi-autonomous agency of the Oromia government, was created in 2007 under the decentralisation of forest management to the regions of Ethiopia. Its function is to coordinate the eight forest enterprises of Oromia. The BME falls under the jurisdiction of two forest enterprises; the Bale Forest Enterprise and the Arsi Forest Enterprise. Although they remain government agencies, the forest enterprises are run and organised like private sector businesses. Revenues and profits, largely from plantations, are earmarked for reinvestment into local-level development (Macqueen, 2008).

3.4.3. Project legal and institutional framework

The regional states of Ethiopia develop their own forest regulations under the guidance of federal proclamations (see Section 3.2.2). Oromia, where the BME is located, has become a leader for forest policy and conservation strategy (R-PP, 2011). Oromia's Forestry Proclamation (72/2003) was the first to legally recognise the ownership and participation of communities in forest management. It therefore, goes beyond the federal Forest Development, Conservation and Utilisation Proclamation (542/2007) which recognises both private and state ownership of forests, by separating out community forestry as a distinct form of ownership. The legal basis for REDD+ at the case study site, as in wider Ethiopia is yet to be determined.

Under the CFM arrangements CBOs will be given forest use rights, but not land rights which remain in the ownership of the state. OFWE will likely remain the legal owner of the emission reductions generated from REDD+ and therefore will act as the lead contractor in Emission Reductions Purchase Agreements (ERPAs). OFWE would then sign contracts with CBOs under the proposed carbon finance scheme, and the Forest Enterprises (Bale and Arsi) would act as executive entities for implementation and monitoring of REDD+ implementation.

CFM necessitates interplay between formal institutions and traditional, customary rules. The tradition Oromo cultural and political system, the *Gada*, is an age-set democratic political institution. Oldest rules refer to the limited time periods when grazing was allowed in the forest (determined annually according to rainfall patterns). The forest coffee harvest period also had strict limits and sanctions imposed for breaking these included social exclusion and deprivation from social support (Wakijira et al., *in press*). Although traditional forest management has been in decline (see Section 3.2.2), there is strong institutional memory in the Bale Mountains and many remember the *Gada*'s management of the forest as it relates to livestock grazing, beekeeping and forest coffee harvesting. It remains to be seen, however, if such institutional memory will aid the implementation of CFM in the BME.

3.5. Survey locations

Three survey locations were selected within the BME for household surveys and forest carbon stock assessments. The survey locations fall in three woredas and are henceforth referred to as: Agarfa, Goro and Delo Mena (Figure 3; Figure 4). Travelling by truck, public bus, horse and foot, survey locations were chosen on the basis of logistical feasibility, but also to represent the three major forest types found in the BME: dry forest, moist forest and woodland.

Initial fieldwork plans had proposed multiple survey locations in each forest type. Delays in research permissions and transport difficulties, however, restricted surveys to only three locations and reduced the sample size. The presence of three survey locations in three forest types means that the effects cannot be separated from other location differences for example in demography or infrastructure. Secondary data were also gathered at each location to provide contextual information to aid the interpretation of the findings (Table 1). These were sourced from village officials, key informants, focus groups as well as Bale REDD+ Project implementers at the case study site.

The Agarfa woreda borders with the Arsi zone of Ethiopia. The woreda is bounded by the Genale river and the Wabe Shabele river with numerous tributaries of these river basins flowing through the region. Within the Agarfa woreda the dominant forest type is dry forest with more than 35,000 ha. Altitude varies between 1000 and 3000 masl and mean annual temperatures are 17.5 degrees Celsius. Tree species found in the dry forest include *Juniperus procera* and *Podocarpus falcatus*. The Agricultural and Rural Development Office estimates 11.5% of Agarfa's land is covered by natural forest and less than 1% with manmade forest, or plantations.

Table 1. Survey location general characteristics

Characteristic	Description	Survey location			All BME
		#1	#2	#3	
Woreda	*The name of the district in which the survey village is found*	Agarfa	Goro	Delo Mena	-
Kebele	*The name of the surveyed village(s)*	Dera Honsho/ Galema Hebano	Walta'i Mana	Irba	-
Population	*The total population*	7703	1529	4465	1,307,078
Households	*The number of households*	1149	255	1170	217,846
HH surveyed	*The number of household surveys undertaken*	87	50	98	235
Proportion of HH surveyed	*The proportion of total village households surveyed*	8%	20%	8%	0.1%
Forest type	*Forest category*	Dry forest	Woodland	Moist forest	(all)
Forest area	*The area of forest*	35,107	5,938	10,673	923,593

The Agricultural and Rural Development Office indicates that 86% of Agarfa's population is rural, with a high proportion of young and few old people resulting in high population growth. The BERSMP estimate population density of 65 to 83 people per km^2. The economic base is rain-fed agriculture including traditional and small-scale cattle rearing. Close to 30% of the total land area of the woreda is agricultural land, and a further 30% is grazing land. There is a dirt road that connects Agarfa to the main all weather road that runs from Robe, the administrative centre of the BME with Awassa, in turn linking to Addis Ababa, the capital. Agarfa is approximately 30 km from Robe, the district centre. However, the form of transport within Agarfa is mainly traditional use of pack animals and humans, for fuelwood loads for example. The total population of the two survey kebeles in Agarfa – Dera Honsho and Galema Hebano – is an estimated 7703, consisting of 1149 households.

Goro woreda has mean annual temperatures of 27 degrees Celsius, but reaching up to 35 degrees Celsius, with annual rainfall of 1900mm. Woodland covers 5,938 hectares and is dominated by acacia. BERSMP estimate that this covers 23% of the total woreda area. Land use is largely cultivated and dominant livelihood is agriculture, including livestock rearing; 39% of the woreda's area is under agricultural production and 3% is grazing land. An estimated 93% of the population of Goro is rural. Population density is estimated by the Agricultural and Rural Development Office at between 24 and 49 people per km^2. Goro lies about 60km from Robe, the administrative centre of the BME. As in other survey sites, the predominant transport form is pack animal. The population of Walta'i Mana is an estimated 1529 and 255 households.

Mean annual temperatures in the Delo Mena woreda are 29.5 degrees Celsius and mean annual rainfall is 700mm. Moist forest covering 10,673ha characterises Delo Mena with Hagenia abyssinica and Coffea arabica characterising the forest: the name of the district comes from the combination of Oromo words Dalaa and Buna which mean "a core place of coffee". It is estimated by the Agriculture and Rural Development Office that 65% of the woreda's area is under forest cover.

The Agriculture and Rural Development office estimates Delo Mena's population density of between 11 and 19 people per km2. Most inhabitants are engaged in agriculture, with 86% of the population rural. This is despite domination of forest cover in the woreda; only 3% is under crop production and 21% is grazing land. As the elevation declines in the woreda, the livestock populations increase with the rising temperatures. Delo Mena is 125km from Robe, the district centre of the BME. An all-weather road connects Delo Mena to Robe, however, it can be a very long journey despite the short distance in the wet season. The Irba kebele in which surveys were undertaken in Delo Mena has an estimated 4465 people across 1170 households.

Across all sites there is limited access to modern energy sources. In the urban parts of BME fire-wood, charcoal, kerosene and electricity are major sources of energy, while in rural areas fire-wood, dung, crop residue, charcoal and occasionally kerosene are used. Each survey site has its own market days in which produce is traded informally for cash. Sometimes goods are also taken to regional markets by pack animal to be traded in the larger towns of Robe and Goba.

3.6. Conclusion

A history of political instability and a drive towards agricultural intensification has side-lined forest conservation in Ethiopia. With rising acknowledgement that forest conservation is necessary to sustain the livelihoods of the population, Oromia regional state is advancing CFM and REDD+. REDD+ revenues resulting from the Bale REDD+ Project could help fund these activities and provide a pilot project for Ethiopia. This research into the economics of REDD+ via CFM is timely at the case study site; it adds to limited data on forest carbon stocks and socio-economic household characteristics. Gathering primary data, this integrated, ex-ante study of the proposed REDD+ via CFM intervention could also inform the intervention design. The forests of the BME are typical of many forests in developing countries that present a *de facto* open access regime on a common pool resource. With REDD+ via CFM being supported more widely in East Africa and beyond, this research also adds to the limited literature on PES, such as REDD+, on common property regimes.

Figure 4. Four major forest and habitat types of the Bale Mountains Eco-Region
(a) Afro-alpine habitat, (b) Dry forest, (c) Moist forest, and (d) Woodland *Source: author's photos*

Chapter 4: Conceptual framework and methods

4.1. Introduction

Drawing on both ecological sciences and social sciences, I undertake an integrated study of the economics of CFM at a case-study site in Ethiopia. This Chapter presents the conceptual framework of this research. An ecological approach is followed to understand forest carbon stocks, emission reductions and REDD+ revenues at the case study site. A more social sciences approach is then adopted to estimate household opportunity costs (OCs) of forest conservation. The approaches are then combined to explore how REDD+ via CFM might be implemented as a local-level PES scheme. The conceptual framework is followed by an overview of research methods applied for forest carbon accounting, and for the estimation of the OCs of REDD+ through household survey, market price valuation, and scenario modelling. These quantitative methods are complemented by more qualitative attitudinal data on the proposed forest conservation intervention which gives context to the empirical findings. A description of the data collection and analysis is also presented.

4.2. Conceptual framework

4.2.1. REDD+ as a PES

A REDD+ mechanism recognises and rewards the positive externalities of climate regulation provided by forest users. It involves an economic incentive that turns standing forest into a valuable asset. It can, therefore, be regarded as a PES scheme (Angelsen, 2008, Campbell, 2009, Fisher et al., 2011). Establishing a price and a market, PES inherently requires the commoditisation of an environmental 'product'. In the case of REDD+, this is the carbon stored in the biomass of trees and forest vegetation. Forests absorb atmospheric carbon through growth and release carbon dioxide (CO_2) through decay, combustion and respiration. The avoidance of deforestation slows the build-up of CO_2, a major GHG, in the atmosphere thus mitigating the impacts of climate change (Bonan, 2008). Forest losses also result in emissions of other GHGs, particularly, methane and nitrous oxide. Emission reductions are therefore, reported as tonnes of carbon dioxide equivalents (tCO_2e) which includes other major GHGs standardised according their global warming potential. Following the Wunder (2005) definition of PES, the well-defined product – a tonne of carbon emission reduction equivalents – is then voluntarily 'bought' from a 'provider' who continually secures the supply of the environmental service.

Based on the underlying logic that voluntary contracts can overcome the market failures of environmental externalities, PES schemes are theoretically grounded in the work of Coase (1960). Coase proposed that if property rights are defined and transaction costs minimal, a socially efficient resource allocation can result from bargaining between those willing-to-pay for an environmental externality and those willing-to-accept compensation for its provision. Although these conditions are unlikely to hold in real life, PES can operate where the willingness-to-pay (WTP) for a service exceeds a provider's OCs of alternative, or foregone, land uses and practices, as well as their participation and transaction costs (Wünscher et al., 2008). Considered by some to be the largest cost in studies of REDD+ (Karky and Skutsch, 2010), in overcoming the OCs of forest conservation the payment should be sufficient to make forest conservation more economically attractive than land use alternatives (Pagiola and Platais, 2007, Engel et al., 2008).

It is acknowledged that PES can exist at many levels. Public schemes, for example in Costa Rica, Mexico and China exist where the state is the buyer of environmental services. Private schemes are often smaller-scale and more local to the point of environmental service provision, with buyers often paying stakeholders directly rather than through intermediaries (Wunder, 2005). The differing scales inherently imply implementation and transaction costs of payments for the variety of stakeholders depending on how it is structured. The level at which REDD+ will operate in the future is not clear. At present, the REDD+ discourse is moving towards national-level REDD+ whereby international financial transfers under a national REDD+ scheme will be based on national-level carbon accounting systems with country governments, or intermediaries, then paying subnational governments or local land owners for emission reductions.

Under some national-level proposals, REDD+ may not operate as a PES. REDD+ can be implemented through a number of policies, actions and measures and these may include strengthening of law enforcement or reductions in logging, rather than payments to communities local to forests (see also Section 2.1). Fisher *et al.* (2011), for example, note that REDD+ in Tanzania could be implemented through alleviating the demand for deforestation by raising agricultural yields on existing cropland and increasing charcoal fuel-use efficiency rather than the OCs of rents from agricultural and charcoal production. It is therefore recognised that even if financial transfers where to be conditional and voluntary at the national level, it may not be at the local-level, for example if national level tenure reforms and law enforcement is put in place to reduce deforestation (see Angelsen, 2008, Olander, 2011 for reviews).

Sub-national, or project-level REDD+ experiences continue to generate most lessons for future REDD+ implementation (Caplow et al., 2011). Alongside these project experiences, a number of initiatives are currently building national REDD+ readiness for example the World Bank's Forest Carbon Partnership Facility or the UN-REDD Programme, but discussion on how national-level REDD+ would operate in international climate change negotiations is ongoing (see the COP17 outcomes of the UNFCCC, 2011). I therefore consider REDD+ as a local-level PES and assume that whether project or national-level REDD+ is pursued, or a combination of both, incentives will still be required on-the-ground to change land use behaviours.

4.2.2. The opportunity costs of REDD+

The foregone benefits of an alternative investment, activity or use of a resource, private OCs of land are limited to those people directly affected by the conservation intervention (Pirard, 2008). The OCs of forest conservation land will be dependent on the underlying drivers of the forest loss. Broad-scale drivers of deforestation are variable; extensive cattle ranching and large-scale soybean production drives losses in South America and large-scale oil palm and wood product plantations in Asia. In Africa, deforestation for small-scale staple crops and fuelwood collection is the primary driver (FAO, 2009).

At a finer scale, the drivers of deforestation depend on returns from non-forest land uses and are affected by accessibility to markets, climate regime, soil fertility, as well as socio-economic variables such as commodity prices, GDP, population growth and density (Geist and Lambin, 2001, Tomich et al., 2005, Chomitz, 2007). A substantial body of literature on household income from forests show that forest reliance is highly heterogeneous (Godoy and Lubowski, 1992, Byron and Arnold, 1999, Cavendish, 2000, Coomes et al., 2004, Dovie et al., 2005). The OCs of forest conservation interventions that alter forest access and extent are, therefore, unlikely to be identical between households.

The drivers of land-use change and thus OCs of land also change over time. The future OCs incurred by local forest stakeholders will be affected by changes in income from direct activities such as agriculture and forest product extraction. The underlying drivers of deforestation will also play a role in influencing future OCs, including changes in demographic, economic, technological, policy and institutional, and cultural factors (Geist and Lambin, 2002). Where resource use is currently unsustainable, OCs may well decline through degradation of the resource base (Pearce and Markandya, 1987). OCs may also be altered through direct actions of conservation interventions, for example, where attempts are made to commercialise and increase the prices of forest products or to diversify livelihoods (Brandon and Wells, 1992, Arnold, 2001). Conservation interventions, however, often suffer from a lack of explicit goals and quantitative operational targets (Margules and Pressey, 2000). This linguistic uncertainty arises from the underspecificity, or generality of most conservation objectives (Regan et al., 2002). With uncertainty about the future drivers of land-use change and uncertainty in the impacts of conservation interventions on households, the OCs of forest conservation are difficult to predict.

The utility of measuring the OCs of forest conservation for a household in the context of REDD+ is threefold. In the first instance, the OCs of forest conservation can be used to estimate the costs of a REDD+ intervention (Fisher et al., 2011). For REDD+ as a local-level PES, information of the magnitude of the OCs of forest conservation can provide information on the private incentives that must be overcome to generate the desired level of forest conservation (Polasky et al., 2005). This therefore helps to estimate payment levels if local communities must forgo certain land uses, but also establishes the feasibility to the project; if OCs of forest conservation are higher than the value of the emission reductions generated through avoided deforestation or degradation then the project may not be financially feasible for the investors to engage in.

Secondly, understanding and incorporating the heterogeneity of the OCs of REDD+ between households into a conservation intervention design can also serve to reduce the risks of negative social impacts. For example, by providing an understanding of whether particular social groups are more likely to experience higher OCs of changes in forest access than others. This is important given growing obligations to ensure that REDD+ projects 'do no harm' to forest communities (e.g. Griffiths, 2007, CCBA, 2008, Griffiths, 2009).

Finally, an understanding of how the OCs of forest conservation change through time will also help meet these costs over time. This will better allow emission reductions to persist into the future and increase the change of REDD+ delivering permanent climate change mitigation benefits as the mechanism was intended. By necessitating and understanding of the drivers of deforestation over time, assessment of the OCs of forest conservation may also reduce the possibility of leakage – the displacement of emission reductions – by ensuring that livelihood needs are considered in policy making.

4.2.3. REDD+ via CFM

Although support for REDD+ via CFM is growing (Klooster and Masera, 2000, Murdiyarso and Skutsch, 2006, Agrawal and Angelsen, 2009, Hayes and Persha, 2010), there has been little consideration of the divergence in incentive design between PES and CFM (Skutsch et al., 2011). The literature on PES has focussed on contracts with individual stakeholders and rarely considers PES on common pool resource or under common property regimes, as is established through CFM (Muradian et al., 2010).

Cooperation on a common pool resource largely means that resource appropriators extract less than private incentives might dictate on an open access resource. However, individuals are willing to incur these costs as well as those that might be incurred for monitoring of extraction, for example, for the longer term sustainability of the resource. This is largely on the understanding of reciprocity and cooperation of others (Heckathorn, 1993, Seabright, 1993). Under CFM, social and cultural norms will act as sanctions and as disincentives for resource appropriators to free-ride, in addition to fines, loss of rights and/or incarceration (Ostrom, 1990). These social and cultural norms have a strong influence on a household's payoffs of cooperation in CFM; their costs and benefits.

PES and OCs estimates of REDD+ omit this logic of collective action on which CFM has historically operated. Estimates of the OCs of REDD+ also omit changes in non-market environmental values generated through forest conservation such as watershed protection, biodiversity protection and the conservation of landscape beauty (Pearce and Warford, 1993, Davies and Richards, 1999). An understanding of non-market values and influence of collective action logic on payoffs could allow more appropriate incentive design.

More cooperation and self-restraint in forest use can bring more significant benefits when followed a greater proportion of users cooperate (Baland and Platteau, 1996, Castillo and Saysel, 2005). Although debate in the literature still remains if bigger groups sizes, and so larger number of cooperating individuals, bring greater benefits. An understanding the characteristics and determinants of households' supply of cooperative effort for REDD+ via CFM ex-ante, could therefore also encourage cooperation of households thus generating greater climate change mitigation benefits overall.

It is increasingly recognised that an understanding of households' environmental attitudes and perceptions of the resource base condition, of perceived environmental responsibilities, and of perceived legitimacy of the intervention are important for conservation success (Zanetell and Knuth, 2004, Davies and Hodge, 2006, Nkonya et al., 2008). Environmental attitudes are underpinned by motivation and belief systems that give rise to values and thus behaviours (Kotchen and Reiling, 2000). In community-based conservation such as CFM, the engagement and participation of the community is by definition central to the interventions success. An understanding of stakeholders' attitudes towards forest management and the use of the resource base will, therefore, allow better consideration of socio-cultural factors for cooperation that go beyond payment incentives that PES theory highlights. For REDD+ via CFM undertaken together, therefore, rather than as separate interventions, sustained cooperation of households in the intervention can deliver more permanent emission reductions. An ex-ante understanding of the perceptions and household's intention to cooperate in a REDD+ via CFM forest conservation intervention can aid in appropriate intervention design and necessary longevity for real climate change mitigation benefits.

4.3. Overview of methods

4.3.1. Forest carbon accounting

The assessment of revenues from the proposed REDD+ via CFM intervention requires knowledge of the amount of carbon stored in forests and the rate of forest loss. This will allow an understanding of deforestation; the complete removal of forest as a result of anthropogenic activities. Forest degradation, which reduces biomass without necessarily losing forest cover, is not assessed here. The rate of forest loss in the BME is being assessed by both Frankfurt Zoological Society and BERSMP – both NGOs involved in natural resource management in the area – through remote sensing, which uses space or air-based platforms to measure spectral indices of forests to which field-based forest carbon measurements are correlated (DeFries et al., 2006). Data on forest carbon stocks in Ethiopia, however, is largely lacking. Ethiopia's national average forest carbon stocks have been reported at 37tC/ha and 47tC/ha (FAO, 2000, Brown, 1997). The national forest inventory of Ethiopia, however, is criticised for conflicting data (Teketay et al., 2010) and country wide estimates are likely to underestimate the forest carbon stocks in the BME for which no estimates are known by the author.

Documented biome averaged carbon stocks are quick to apply and very low cost. These biome averages capture broad ecological variables that determine carbon stocks such as climatic zones which are based on temperature and rainfall regimes (IPCC, 2003, IPCC, 2006). The simple application of biome averages of carbon stock, however, obscures the substantial heterogeneity of forests. The biomass and so carbon content and rate of accumulation, also varies with factors such as soil type, topography, elevation, species composition, age and land use history (UNDP, 2009). Human activities in a given year such as logging intensity, distance to settlements, transport networks, and forest edge, will also impact on carbon stocks (Larocque et al., 2008).

More complex forest carbon stock accounting uses forest inventory to statistically relate tree diameters, or biomass volumes, to carbon stock using documented allometric relationships established through destructive tree measurements (e.g. Brown, 1997, Chave et al., 2005). Tree diameters and volumes can be sourced from field measurements or existing forest inventories which record forest stand structure, age, growth rate, biomass accumulation, and wood density (see FAO, 2006). Criticism of Ethiopia's national forest inventories, however, highlight the conflicting data that has been produced and that no regular or consistent inventory exists (Teketay et al., 2010). The above-ground biomass carbon pool at the case study site was, therefore, estimated by gathering direct tree measurements from 108 forest plots of 20m by 20m (see Section 4.3.3 on data collection).

Direct tree measurements and sampling protocol followed best practice methodologies and guidance (e.g. Brown, 1997, MacDicken, 1997, Pearson et al., 2005). Carbon is present in above-ground biomass, below-ground biomass, dead organic wood and litter, soil organic matter and harvested wood products. Although resulting in an underestimate of carbon stocks, only the above-ground tree biomass carbon pool was considered here as it contains the greatest fraction of total living biomass in a forest and this pool is most immediately impacted by deforestation and degradation (Brown, 1997, FAO, 2003).

Pan-tropical allometric equations were applied to estimate biomass from Brown (1997). These allometric equations were applied as few exist for Sub-Saharan African trees and woodland (Henry et al., 2011, Shackleton and Scholes, 2011). However, it is acknowledge that site and species specific allometric equations would allow better biomass estimation as they capture heterogeneity in forest characteristics. Thus, while few datasets from Africa exist to validate the allometric equations applied to the direct tree measurements at the case study site (Gibbs et al., 2007), resources to undertake destructive sampling to verify allometric equations were not available. Height measurements were also impractical and wood density estimates did not exist for the study area (see also Chapter 5, Section 5.2.1 for a longer discussion on allometric equations). Tree biomass was converted to carbon using the IPCC carbon fraction guidance of 0.47 (IPCC, 2006) and converted to a per hectare value.

To explore the discrepancy between simple and complex forest carbon accounting, Chapter 5 applies biome averaged and primary data to estimate the emission reductions and REDD+ revenues that could be generated in the BME. Biome averages are sourced from the Intergovernmental Panel on Climate Change (IPCC). Emission reductions are evaluated as the difference between a business-as-usual (BAU) deforestation baselines and an avoided deforestation project scenario. Methods to establish this deforestation baseline are controversial and have been discussed at length in the literature (see Olander et al., 2008, Huettner et al., 2009).

Approaches range from simple extrapolation of historical deforestation rates to complex and dynamic models of future land use (see Parker et al., 2008). Extrapolating from trends in forest cover change generated from GIS imagery analysed by the BERSMP, a historical emissions approach was adopted to establish the BAU deforestation scenario for the BME. A linear deforestation rate of 4% in all forest types was used to estimate emission reductions and subsequent REDD+ revenues. More complex models that predict deforestation rates and incorporate, for example, demographic, economic and technological variables which lead to infrastructure, energy and food demands that drive land-use change can also be used to establish BAU deforestation baselines (Huettner et al., 2009). These complex models are more politically acceptable and better predict deforestation rates (Böttcher et al., 2009), but the substantial data sets and technical capacity meant that this approach was not possible at the case study site.

Emission reductions were adjusted to account for possible project leakage and non-permanence. Leakage is the relocation of emission generating activities away from a REDD+ project boundary. It is termed primary leakage where the project fails to address the drivers of the original deforestation baseline; activities are shifted elsewhere or there is outsourcing for the products used as the baseline scenario. It is termed secondary leakage where third parties are incentivised to increase emission reductions activities as a result of the project; market effects on product supply and demand for example (Aukland et al., 2002). Both categories of leakage need to be accounted for so that emission reductions are not overestimated. Project permanence is the persistence of emission reductions over time (Sedjo and Marland, 2003). Permanence can be threatened by financial or management failure; economic risks, rising OC; political and social instability; and natural disturbances (fires, pests, disease and extreme climatic events) (VCS, 2007).

The dominant project-based method to deal with leakage and non-permanence are buffers of emission reductions, with other options suggested to be repayments of revenues/fines, expiring emission reductions, ex-post payments, portfolio approaches and insurance (Peskett and Harkin, 2007). A non-tradable reserve of emission reductions, the buffer acts as insurance for any emission reductions targets that are not achieved. With a history of forest fire, potential land disputes and imminent infrastructure development at the case study site, a buffer of 65% of emission reductions are set aside in Chapter 5.

To remaining emission reductions, market variables are applied to estimate possible REDD+ revenues. REDD+ revenue will depend on the price of a tonne of emission reductions and the costs of getting the emission reductions to market. Although social costing of carbon would value emission reductions more highly at US$23/t$CO_2$e (Tol, 2008), the voluntary carbon market is currently the only trading platform from which value can be realised from avoided deforestation. In 2007, the average price for emission reductions on the voluntary market was US$6.1/t$CO_2$e

(Hamilton et al., 2008). With greater flexibility and less stringent validation processes, the voluntary market price is lower than that in compliance markets. Prices for emission reductions through the CDM on the compliance market fetched an average of US$13.6/t$CO_2$e in 2007 (Capoor and Ambrosi, 2008). The VCM also allows price variation according to the source and integrity of the offset. In 2007, emission reductions on the VCM were sold for between US$1.8 to US$300 per tCO_2e (Hamilton et al., 2008). The highest prices went to projects with easily verifiable attributes and those that were more publicly appealing. Lower prices were realised by projects with low social or environmental co-benefits and high economic and project delivery risks. Where social co-benefits refer to additional positive impacts beyond climate regulation and may include improvement in long-term livelihood security or employment opportunities, for example. Environmental co-benefits may refer to REDD+ activities that operate in areas of high biodiversity, or those that contribute to watershed and soil regulation for example. In 2006-2007, emission reductions from avoided deforestation averaged US$4.8/t$CO_2$e (Hamilton et al., 2008). Two prices were used in Chapter 5, US$3/t$CO_2$e and US$6/tCO_2e to illustrate the sensitivity of emission reductions to market price.

Estimated REDD+ revenues were further adjusted for the implementation, transaction and capacity building costs incurred when bringing emission reductions to market. Implementation costs are either one-off or ongoing, but are incurred through actions directly generating emission reductions. They include; guards, intensification of agriculture, and re-routing of road projects. Transaction costs are those experienced when identifying the programme, negotiating transactions, and for MRV of emission reductions (Pagiola and Bosquet, 2009). Capacity building costs include those for the development of research capacity, technology transfer and legal support to establish REDD+ projects (see Hoare et al., 2008).

The bulk of these additional costs are experienced upfront and have, to date, been absorbed by stakeholders other than the ultimate forest users. Particularly where forest users are rural communities they have been absorbed by NGOs such as; The Nature Conservancy in Bolivia, and Conservation International and Wildlife Conservation Society in Madagascar (Asquith et al., 2002, WCS, 2009). With REDD+ an emerging policy instrument, very little has been documented about costs. The few estimates that do exist, however, show that these costs can be substantial (Cacho et al., 2005). Implementation costs were predicted by Nepstad *et al*. (2007) to be US\$0.58/tCO_2e. Antinori and Sathaye (2007) found average transaction costs of US\$0.38/tCO_2e from a sample of eleven project reports. Based on their experiences in Madagascar, the Wildlife Conservation Society estimate the costs of REDD+ project development at between US\$220-450 million, excluding implementation costs and brokerage of emission reductions (WCS, 2009). REDD+ revenues estimated in Chapter 5 were adjusted for costs of REDD+ project implementation estimated using a feasibility assessment undertaken by forestry consultants in the BME (UNIQUE, 2008).

4.3.2. The opportunity costs of REDD+

4.3.2.1. Estimates of the OC of REDD+

Estimates of the OCs of REDD+ can be broadly split into top-down and bottom-up assessments. Top-down assessments are coarse, aggregating forests into large blocks for example by country, continent or biome. They commonly make use of commercial agricultural returns on a hectare of land and estimate the highest potential OCs. These estimates differ in choice of the time frame considered, the costs included, market feedbacks, drivers of deforestation, land conversion benefits, elasticity of transformation, carbon density, and the benefits derived from retention of forest (see Nabuurs et al., 2007). At large spatial scales they make broad assumptions for agricultural returns, ignoring the substantial heterogeneity of both ecological and socio-economic factors.

Such top-down OCs analyses are too coarse to feed into on-the-ground REDD+ project design. Instead they have utility as components of global partial equilibrium models and global assessments of REDD+ supply (e.g. Grieg-Gran, 2006, Kindermann et al., 2008). Supply curves express OCs by quantity of emission reductions rather than by area. The OCs estimates, typically in US$ per hectare, are converted into US$ per tonne of emission reductions. The comparison of OCs estimates is made complex by the type of OCs reported. Average OCs in Indonesia, for example, ranged from US$-0.26 to US$5.22/tCO_2 where forest was razed for agricultural use and US$13.34/t$CO_2$ where it was commercially logged (Tomich et al., 2005). The 'choke' price to reduce all deforestation in the Brazilian Amazon was found to US$1.49/t$CO_2$e (Nepstad et al., 2007). Although coarse, top-down model estimates broadly indicate where emission reductions will be most cost-effective, and allow a comparison of abatement costs through forestry compared to other mitigation sectors.

Bottom-up studies are more specific to a particular locale, but still make use of agricultural returns, production models or land prices and, therefore, also consider the OCs of land. Fisher *et al.* (2011), for example, include both OCs of agricultural production and charcoal production within 53 districts in Tanzania finding net present value of between US$663 and US$1456/ha for agricultural production, and US$358 and US$502/ha for charcoal production. Bottom-up models are better able to include local factors including soil type, climate, technological inputs, and market access which enhance OC estimates (e.g. Merry et al., 2002, Bellassen and Gitz, 2008). Where substantial data sets and technical capacity exists, more complex production functions can be used to model agricultural returns. The production function approach incorporates variables such as yields, inputs, commodity prices and other spatial details. Alternatively, land values can be used to infer OCs as the market price of land, under perfect markets, should reflect its highest-value use (Bishop, 1999). This method, however, requires data to be available on land title costs. In developing countries, this data is limited and clear ownership and land markets often do not exist (Waggoner, 2009).

Few studies have considered the OCs of REDD+ via CFM. In Nepstad *et al.* (2007) the costs of REDD+ are assessed in the Brazilian Amazon. They establish the OCs of land for private forest stewards and for the government, and also suggest a payment level that can incentivise forest stewardship and conservation on 'social' forest reserves. These social forests comprise 26% of the Amazon's forest and include indigenous lands, extractive reserves, and sustainable development reserves. Nepstad *et al.* propose a Public Forest Stewardship Fund on these forest areas from which direct payments can be made to households. The payment is delivered per household, not by area, and payments are uniform and anchored to half a minimum salary (amounting to US$1200 per year). These payments are lacking conditionality on service provision and it is noted that more research is required to make these payments performance based.

In Karky and Skutsch (2010), the costs of carbon abatement through community forestry are calculated in Nepal. Establishing the break-even price that would be required for emission reductions to make REDD+ via CFM feasible, they call for the analysis of the OCs of land that encompasses more than agricultural returns and note the numerous other drivers of deforestation such as the harvest of fuelwood, fodder, timber and NTFP. Studies based on marginal analysis of the OCs of agriculture may be inadequate to anchor payments for REDD+ via CFM where they do not consider other inputs for subsistence livelihoods. In Chapter 6, I therefore, estimate both the OCs of agricultural production but also the OCs of fuelwood and timber on a hectare of land. These bottom-up estimates are based on household returns to land uses which are established through household survey and market price valuation.

4.3.2.2. Estimating OCs at the case study site

In the BME, under the proposed Bale REDD+ Project of REDD+ via CFM households will experience a change from open access to the forest resource to a common property regime. Despite the illegality of the expansion of agricultural land and the harvesting of fuelwood from live trees, in the status-quo anyone is able to use resources from the forest to the level they desire. This *de facto* open access situation is due to a lack of law enforcement and political interest in conserving the forest resource base (see the full Bale REDD+ Project description in Chapter 3).

Under the CFM regime, clearly defined use rights to the forest will make forest use excludable from those not participating in CFM and also to regulate forest use. The forest management agreement signed by the community groups will prohibit household expansion of agricultural land and engagement in timber and fuelwood extraction. Timber and fuelwood harvest reduces the biomass content of the forest where they are in excess of annual biomass growth. These are therefore termed, high-impact forest products here. The extraction of bamboo, honey, coffee, and climbers from the forest, will still be allowed under the intervention. These products can be managed such that they are harvested without the reduction in the biomass of the forest (Naidoo and Adamowicz, 2006). They are, therefore, termed here, low-impact forest products. While agricultural expansion and harvest of high-impact forest products will be prohibited by the Bale REDD+ Project, low-impact forest products can still be harvested, providing this extraction remains at sustainable levels.

The OCs of forest conservation will therefore be those of agriculture or high-impact forest products on a given hectare. As no model of land use change exists for the BME, so it cannot be predicted whether the next hectare will be converted to agriculture or deforested through harvest of high-impact forest products. Both OCs of land are therefore estimated in Chapter 6 and explored in regard to the implementation of the Bale REDD+ Project.

The OCs for a hectare of forest conservation (US$/ha) is first modelled as the foregone income from crop production. A second calculation is made of the OCs for a hectare of forest conservation net of low-impact forest product income that can instead be derived from the hectare of forest conserved. The inclusion of these market benefits of low-impact forest products better capture the household trade-offs on this hectare of land.

The OCs of high-impact forest product (US$/ha) is then estimated by aggregating the village forest income from timber and firewood, through household survey and market price valuation, and then dividing over the total forest area. While other studies have estimated by biomass per hectare and converted by market survey to estimate land use values (Fisher et al., 2011), it was not possible to do so at the case study site as estimates of a donkey load of biomass for fuelwood were unavailable (see Section 4.4.4). This assessment of OCs of high-impact forest products assumes that all household use of high-impact forest products must stop under the Bale REDD+ Project. It is recognised, however, that households need fuelwood. The Bale REDD+ Project implementers are already undertaking activities to meet these energy needs through woodlot establishment, fuel-efficient stove promotion and biomass briquettes. However, the measures to reduce the need for households to gather these products from natural forest will take time to be implemented. Woodlots, for example, will take time to be planted and mature. These OCs of the intervention may therefore be overcome as the intervention matures. As with agriculture, a second calculation of the OCs of high-impact forest products is calculated net of low-impact forest products.

Having estimated the OCs of forest conservation, the implications of the different OCs measures, with regard to any payment design of REDD+ via CFM as a local-level PES, are discussed in Chapter 6. The OCs of land generated by the REDD+ via CFM intervention are directly compared with the carbon revenues per hectare of conserved forest in Chapter 7.

Households also derive value from non-market benefits of the forest. These include other direct use values such as shade, recreation and cultural values; indirect use values that support and protect production such as soil fertility and the micro-climate; option value for future direct and indirect value; and, non-use values which capture the value of the forest's existence and bequest for future generations (Davies and Richards, 1999). The values that households derive from forests in the status-quo are, however, net of the negative externality that households exert on each other due to the non-excludable, rival nature of the forest. Inherent in the definition of an externality is that households do not take into account the effect on others when deciding how much of this externality to produce (Kolstad, 2000). Under CFM, households will experience benefits from the removal of the negative externality of a common pool resource, thus it is likely that non-market environmental benefits will be greater than in the status-quo.

Households will also benefit from the scheme according to the value placed on the use rights which they are awarded, any increase in social capital and empowerment as a result of CFM, as well as any payments for carbon under a REDD+ project. These benefits of the conservation intervention may serve to offset some of a household's OCs, but transaction costs will also be incurred by household participating in the scheme including negotiation, monitoring and enforcement costs that are not measured here. These transaction costs include; arranging, bargaining, monitoring and enforcing agreements (North, 1990).

An understanding of transaction costs can help in intervention design to reduce negative social impacts. Meshack *et al*. (2006), for example, assessed the transaction costs of CFM in Tanzania including for forest monitoring and meetings, against the benefits including the forest products consumed at the household level. Poorer households were found to benefit more than medium and rich households, although richer households had greater net benefits; poor taking on more of the transaction costs of CFM. Although it is noted that forest condition also plays a role in determining the transaction costs of CFM. Similarly, in Nepal it was found that while richer household bore almost twice as much as poorer households, 2312 versus 1265 Nepalese rupees per year, costs are higher as a percentage of resource appropriation costs for poorer households; with all households investing a mean of between 20 and 30 days per year (Adhikari and Lovett, 2006).

Transaction costs of CFM, however, are complex to measure. With varying definition, they are also difficult to separate from production decisions in addition to which each individual will experience different transaction costs (Benham and

Benham, 2000). Estimates of transaction costs in CFM are, therefore, few (Richards et al., 1999, Adhikari and Lovett, 2006, Meshack et al., 2006). While this thesis does not assess transaction costs of CFM, or the non-market benefits of the forest that result from conservation are omitted in the OC calculations in Chapters 6 and 7, they are discussed further in Chapter 8 and in Chapter 9.

In order to estimate the three OC measures, a number of simplifying assumptions are made about the household and about the costs and benefits of the intervention (see Table 2). One major assumption is that a household is a pure profit maximiser; thus profit affects consumption with no feedback on production decisions. However, it is well recognised that rural households in developing countries face a number of market imperfections and constraints. This includes variable transaction costs for households of accessing markets, inexistence of land markets and constraints on market participation (Sadoulet and de Janvry, 1995). In such situations, there is a link between production and consumption behaviour; where production is the inputs, choice of activities and desired production levels, while consumption is affected by consumption preferences, and demographic composition of the household, for example. Behaviour can therefore be understood in a non-separable household model (for example see; Palmer and Macgregor, 2009). A non-separable model has implications for the market price of what is consumed and the household internal equilibrium determines the shadow price of a product. At the case study site, however, it was not possible to estimate shadow prices for each household for each product due to resource and time limitations (see also Section 4.4.5) and therefore production and consumption decisions were assumed separable that is likely to overestimate values. These limitations are returned to in Chapter 9.

Another assumption is that low-impact forest products are sustainably harvested and high-impact forest products are not. These stylised assumptions were necessary as incomplete data existed on whether these uses are sustainable and at what level harvests can be maintained. There were also insufficient resources to estimate this at the case study site. It is acknowledged that the reduction in OCs of land as a result of low-impact forest products may, therefore, be an overestimate if they are to be restricted under the intervention. Furthermore, some harvest of biomass growth or gathering of dead biomass for fuelwood may be allowed under the intervention and would not necessarily prove unsustainable use of forest resources. As an ex-ante study of the OCs of high-impact forest conservation, however, it was also not possible to estimate the impact of restrictions on a household that were less than 100%. This is firstly as the by-laws that will generate these restrictions are yet to be negotiated and agreed by the communities with the authorities, and secondly as restrictions are difficult to relate to household harvests. Thus, the OCs of forest conservation was estimated as a total ban on harvesting of all high-impact forest products and with no restrictions on low-impact forest products. Further research into the sustainability and extractive potential of forest products is necessary.

Finally, in order to establish a per hectare value for the OCs of forest products, it is also assumed that the complete forest area in a village is utilised evenly. This assumption of area was used to calculate both the OCs of high-impact forest products, but also those of low-impact forest products per hectare. While the income per hectare of agricultural land was based on reported area of a household's cultivated land, households were unable to recall areas of forest used. It is recognised that problems of attribution of deforestation to households or individuals exist in the REDD+ literature (Börner and Wunder, 2008). This is acknowledged as a substantial assumption and explored further in Chapter 6, but was the best approach given the data available and that follow up research could improve upon.

Table 2. Assumptions made in the estimate of the opportunity costs of REDD+ via CFM at the case study site.

Assumption	Justification	Implication
Livestock income was not accounted for	Livestock are grazed both in forest and on agricultural crop residues and income is experienced over multiple years, thus there is complexity in their valuation (see Naidoo and Iwamura, 2007).	Rotational grazing restrictions required under CFM are not predicted to impact more than 10% of a household's grazing activities (Irwin, 2009).
Households are short-term, risk neutral, profit maximising agents with complete information, unlimited by capital and labour constraints	While households in rural developing economies often link production and consumption decisions this assumption was unavoidable in light of time and resource constraints.	It is possible that by using a separable model with market prices the values of OCs are overestimated in this thesis.
Foregone land uses generate the same income as a household's existing income from that land use	Information on factors which impact income, such as accessibility to markets, climate regime and soil fertility, were not available.	Land not under a specific use are likely to be more marginal, which may result in an overestimate of OCs.
The one-off benefits of deforestation and conversion costs are zero	A common assumption in OCs of forest conservation and REDD+ where these values are not known (e.g. Naidoo and Adamowicz, 2006, Grieg-Gran, 2008).	In the light of small-holder driven land conversion in the BME this assumption appears reasonable.
Newly cultivated land derives from forested land	No data are available on land conversion other than estimates of overall rates of forest loss.	More data on land dynamics in the BME are required to fully assess the impact of this assumption
Households have access to the total forest area in their village, from which they can harvest low-impact forest products	Households were unable to report the area of forest they harvested forest products from. The total forest area in the village was therefore the best assumption available.	At present it is not known if this over or underestimates the area of extraction and more research is required to understand the implication for the OC estimate.
Non-market environmental benefits and scheme benefits are assumed 0	Values are complex to calculate and methods vary in theoretical validity and acceptance, data requirements and ease of application (see OECD, 2002, Pagiola et al., 2005b).	These non-market values are likely to increase under the intervention. Thus, OCs may be an overestimate.
Transaction costs are not accounted for and assumed 0	Transaction costs (e.g. negotiation, monitoring and enforcement), particularly the time burden imposed on households is unclear.	This is likely to underestimate the costs to a household of the intervention.

4.3.2.3. *Household survey of land use income*

In order to elicit information about a household's use of the forests, agricultural production and other economic activities, semi-structured surveys were undertaken at the case study site. A household is defined here as '*the people that normally eat and sleep under the same roof*' (Rowland and Gatward, 2003). Semi-structured household surveys allow the collection of data in a formal standardised manner, but also have room for open-ended responses. The household survey was formulated according to best practice guidelines and to be as specific and simple as possible (de Vaus, 2002). It collected data on the previous year of crop production and forest product collection, with income defined to households as production both consumed at home and exchanged on markets.

Surveys were designed to be verbally administered in either Amharic or Oromifa, the two dominant dialects at the case study site. Neither postal nor telephone surveys were a viable option and self-completing questionnaires would suffer from problems of illiteracy. Households were considered as the appropriate unit for decision-making and respondents were largely household heads, defined by Adhikari *et al.* (2004) as '*the person who makes all decisions on behalf of all the family and decides livelihood activities for the welfare of family members*'. Each respondent was given an introduction to the research, a promise of confidentiality of the information gathered, and an estimation of the survey duration. Respondents were then asked if they wished to proceed.

The survey began with questions regarding attitudes to the environment and forest management. Values and beliefs were elicited in agree/disagree statements and open-ended questions explored environmental concerns as well as opinions of past, present and proposed forest management regimes. More sensitive data were gathered in part two which explored the household's forest use and agricultural production including the products, yields, any inputs (seeds, fertiliser and equipment per year) and the share of production consumed at home versus that sold in local markets. Part three of the household survey went into more detail on the proposed Bale REDD+ Project and the final section elicited household information including family size and the education level of the household head (see Appendix 1).

The quality of the survey data relies on the reliability of self-reporting by households. Reliability of data can be called into question where respondents have motives to alter their apparent resource use or if respondents are unable to accurately recall production information over a given time span (Milner-Gulland and Rowcliffe, 2007, Angelsen et al., 2011). In the first instance, respondents might be reluctant to answer accurately where it is feared that information would reach the authorities, for example, where resource use is illegal as hypothesised in Gross-Camp *et al.* (2012). Alternatively, respondents may inflate their use of resources where they perceive future benefits, for example strategic responses might be given when households are asked their willingness-to-accept restrictions (Whittington, 1998). In order to minimise the risk of false self-reports, interviews were designed for a sole respondent and those participating in the survey were given the assurance of anonymity. Respondents also had the opportunity to opt out of participation. No government staff accompanied the fieldwork team, although permissions to conduct surveys were necessarily sought from the Federal government and also the regional Ministry of Agriculture and Rural Development. At each survey location permission to conduct surveys was also requested from village leaders after an introduction to the research aims and the fieldwork team had been given. Time was invested at each survey location in earning the trust of communities; three to four weeks were spent at each location, and no payments were offered for participation in the survey.

Recall error may also erode confidence in household survey data (Angelsen et al., 2011). In order to address recall issues focus groups were conducted to ask locals the longest recall periods that were possible. Four small groups of between four and seven people were brought together to discuss whether households would be able to recall production from the last 12 months. Respondents were confident they could accurately recall this information, with some focus group participants stating they could remember as much as five years back. Confidence in recall amounts is also high because many crops only have one harvest per annum. With some forest products, such as fuelwood, enumerators were able to scale up where respondents recalled monthly or weekly yields.

4.3.2.4. *Market price valuation*

Research into household incomes and household income from forest resources has been primarily undertaken with a focus on the dependence and resilience of rural households (see reviews of Lampietti and Dixon, 1995, Godoy and Lubowski, 1992, Vedeld et al., 2004). Household incomes are commonly assessed through household surveys to which market-based valuation of household production is employed, particularly to determine the relative reliance of households on forests as a livelihood-generating resource (e.g. Dercon, 1998, Dovie et al., 2005, Shackleton and Campbell, 2001).

The costs of household labour were not subtracted from the income calculations, as is common in household income studies, (Cavendish, 2000, Fisher, 2004, Babulo et al., 2009, Yemiru et al., 2010). In 2008, focus groups also revealed that job opportunities in rural areas were extremely limited, seasonal and largely only available for males. Thus, the market wage was not a good measure of the shadow wage and resources were insufficient to estimate the shadow wage indirectly as an opportunity cost of time (Palmer and Macgregor, 2009). Thus income is defined as the return to capital and labour a household has access to.

The household income calculation includes production of agricultural goods and forest goods both for home consumption and market exchange. These methods vary in theoretical validity and acceptance, data requirements and ease of application (see OECD, 2002, Pagiola et al., 2005b). Where goods and services are marketed, they have evident values. Where goods and services are not present in markets revealed preference, using surrogate markets to infer value, and stated preference methods, using hypothetical markets, can be applied (see Arrow et al., 1993). In addition to these methods, benefit transfer can be used to determine value from related studies (Splash and Vatn, 2006).

I apply market-based valuation to establish the income that households derive from both forest and agricultural land use. Following observation of a household's products and yields, for both subsistence and sale, local market prices are applied. The cost of similar goods or next best alternatives can also be used as a proxy where there is a high degree of substitution between goods (see Section 4.4.5). It is recognised, however, that production and consumption decisions are non-separable in many rural developing country households (Sadoulet and de Janvry, 1995). Multiple market failures mean that there can be a large discrepancy between seller and buyer prices of a product; each household, therefore, will have its own shadow price for a product.

There are a number of ways to establish a households' shadow price which can be used to better value non-marketed products; i.e. those consumed at home. This includes establishing barter values for non-traded commodities that are exchanged between households; using contingent valuation to ask respondents directly for their value preferences; considering substitute good values; using local prices; and, assessing time embedded in products as well as other inputs (Angelsen et al., 2011). Ideally, establishing the time and other inputs would allow better calculation of minimum values of products. Chopra (1993) for example, valued firewood collection and other NTFP through embedded labour collection time; the opportunity cost of labour time. It is however, difficult to measure embedded time and thus shadow prices for each product. Individuals often multi-task, shadow costs vary according to the household members whose labour is used and can also vary by season (Angelsen et al., 2011).

As it was not possible to establish a shadow price for each product for each household, this study uses market price valuation. Market prices reflect decision-making reality and so are good estimates of WTP (UNEP, 1998). Adopting a utilitarian concept of value, WTP reveals the value individuals hold for market and non-market goods and the trade-offs made in the pursuit of these goods (Freeman, 2003). However, using market price valuation assumes that the market is efficient and so inclusive of input costs (Bishop, 1999). As it is noted that market imperfections are commonplace in rural developing countries, it is likely that this method causes and overestimate of value as a result of included marketing and transport costs, or where middle men are buyers and seller increasing the difference between market price and shadow price.

In attempting to minimise this overestimate, local-level market prices were used and extrapolated to 'free' products that were consumed within the home but not traded. In the BME, households sell home produce in unrestricted markets where there are no barriers to entry. On market days many buyers and sellers converge to sell identical produce brought predominantly by pack animals. These local markets are not for sale to intermediaries or middle-men, and although the use of market prices may lead to overestimation it represents the best price for products that were available (see Appendix 2).

4.3.2.5. *Scenario modelling futures*

Complete knowledge of the future is just not possible. Partial forecasting of futures, however, can be achieved through systematic historical trend analysis and extrapolation (Helmer, 1977, Wack, 1985, Bell, 1997). Regarded as a strategic tool, futures research explores a range of possible, plausible futures and, therefore, differs from research attempting to converge on a single view or answer (Gordon, 1992). Alternative futures can answer questions such as: what can or could be (possible)?; what is likely to be (probable)?; and, what ought to be (preferable)? (Börjeson et al., 2006). Thus futures research is useful for strategic decision-making under uncertain but predictable situations, where adaptation is possible through the reallocation of means and resources (Kaivo-oja et al., 2004). Futures research therefore has applications for the private sector (Huss and Honton, 1987), as well as being important for policy planning where they can be used to identify and evaluate alternative policies and provide early warning of threats and opportunities. In addition, where more desirable futures can be selected, stakeholders can act to maximise the probability of desirable futures being achieved (Gordon, 1992, Kaivo-oja et al., 2004).

Futures research encompasses a number of methods. Reviewed in Gordon (1992), the most simplistic division of futures methods is by quantitative or qualitative and normative futures (those that seem desirable), or exploratory futures (those that seem plausible) (Table 3). There is a substantial terminology in futures methods, often with overlapping terms (Marien, 2002). The UK Department of Environment, Food and Rural Affairs have a dedicated horizon scanning and futures programme for example, with its own terminology to describe futures methods and techniques. While horizon scanning is considered as a first step to understand the problem being researched, methods of establishing how the future will play out include: examining *wild-card* high impact, low probability events; *road mapping* of inhibitory and enabling processes; *wind-tunnelling* to identify how economic, political, social, environmental and technical factors would need to exist for scenarios to be plausible, and *back-casting*, which works backwards from a vision to the present (DEFRA, undated).

Table 3. An outline of futures methods
(adapted from Gordon, 1992)

	Normative	Exploratory
Quantitative	Scenarios Technology sequence analysis	Scenarios Time series Regression analysis Multiple-equation models Probabilistic models - trend impact - cross impact - interax Non-linear models
Qualitative	Scenarios Delphi In-depth interviews Expert group meetings Genius Science fiction	Scenarios Delphi In-depth interviews Expert group meetings Genius

Of futures methods, scenarios can be applied for normative and explorative, qualitative and quantitative futures analysis. Scenarios embody the central principles of futures research through creative thinking and present multiple plausible futures (Bishop et al., 2007). As in Bohensky *et al.* (2006), scenarios are defined as a set of plausible narratives depicting alternative pathways to the future. They can synthesise and communicate information, including uncertainties, to stakeholders as well as the public (Alcamo, 2001). Gordon (1992) rates scenarios as less complex than alternative quantitative models methods. Models often rely on the past as being able to predict the future but in the future relationships between variables may change. Time series analysis is more demanding numerically, necessitating the fitting of mathematical models to trend data. Scenarios are also relatively low on training and data requirements as compared to other quantitative futures methods. Scenarios are also unlike other methods to deal with decision-making under uncertainty. Unlike decision theory, for example, scenarios do not require information on the probabilities of outcomes (Polasky et al., 2011). Unlike sensitivity analysis, which focuses on marginal changes in specific biophysical or economic parameters, scenarios have the benefit of being able to change groups of parameters (White and Minang, 2011).

The internally consistent and realistic narratives describing potential future states established in quantitative scenarios can lead to more resilient conservation policies (Peterson et al., 2003). Despite this utility, scenarios have been underutilised in conservation intervention planning (Peterson et al., 2003, Bohensky et al., 2006).

The application of scenarios in environment policy is, however, growing. The IPCC produces special reports on emission scenarios, or '*projections of the future state of the society and environment based on specific assumptions about key determinants such as population, economic growth, technological change, or environmental policies*' (Nakicenovic et al., 2000). The Millennium Ecosystem Assessment built scenarios to explore user needs, supply and demand for ecosystem services and how well-being might change into the future (MA, 2005, Carpenter et al., 2006). Participatory methods were used to generate four policy relevant scenarios with ecologists, economists, and social scientists from the private sector, public sector, NGOs and indigenous groups all engaged in the process (Bohensky et al., 2006). Similarly, scenarios were used in the recent UK National Ecosystem Assessment, to explore how ecosystems and their services will change in the future and the associated impacts on human-well-being. The National Ecosystem Assessment created six scenarios of ecosystem service impacts on society, economy and human well-being up to 2060. These incorporated five indirect drivers of change; demographic, socio-political, economic, science and technological, and cultural and religions, and three dominant direct drivers of change; climate change, land-use change, and resource consumption (Haines-Young et al., 2010).

Where applied for environmental policy scenarios are more commonly applied at broad spatial scale. Osvaldo *et al.* (2000) created three scenarios of the future biodiversity of ten major biomes based on assumptions about the five main drivers of biodiversity change; land use, climate, nitrogen deposition, biotic exchange, and atmospheric CO_2. The scenarios considered no interaction, synergistic interaction and antagonistic interactions between the drivers and land-use change was projected to have the biggest impact on biodiversity distribution in 2100. However, the authors recognise that regional analysis, with tailored biological, social and economic characteristics, will improve the accessibility of the scenarios to policy-makers (Osvaldo et al., 2000).

Scenarios are being increasing used to consider carbon storage in natural ecosystems. Swetnam *et al*. (2011) was also at broad-scale, building two scenarios of carbon storage in the Eastern Arc Mountains of Tanzania. Considering change in five sectors; energy, formal economy, agriculture, forestry and population, it was estimated that in 2025 there would be a 41% loss in carbon storage under business as usual charcoal production and agricultural expansion. Translating scenarios onto land use maps, it was shown that in the optimistic scenario only 3.8% of carbon storage of might be lost. Strassburg *et al.* (2012) applied scenarios of global carbon values to explore how REDD+ might influence biodiversity conservation. They found that under all scenarios, REDD+ will help reduce biodiversity losses.

A qualitative study by Wollenberg *et al*. (2000) argued that scenarios should be utilised in bottom-up conservation planning. Applied to anticipatory learning for adaptive co-management of community forests, the study finds that scenarios may not remove uncertainties, but they can help stakeholders to prepare for them, and thus cope with them.

Studies that consider the OCs of conservation largely report OCs for a single year or assume OCs are constant over time subject only to discounting (Börner et al., 2009, Naidoo and Adamowicz, 2006, Chomitz et al., 2005). Incorporating the lack of information the on on-going drivers of change and the underspecificity uncertainty in conservation objectives, scenarios are applied in Chapter 7 to understand how OCs may change over the lifespan of a conservation intervention. Three scenarios are generated which explore how assumptions of agricultural productivity improvements, proposed commercialisation of forest products, and the sustainability of land use impact upon three OCs measures of forest conservation through CFM. The annual OCs are those experienced by a household in a given project year. The cumulative OCs are those experienced for a hectare of land taken out of production at a given project-year until the end of the project. The total OCs are the sum of the cumulative OCs, over the area of avoided deforestation, for the project lifespan.

Scenario analysis can include indirect socio-political, economic, science and technological, cultural and religious, and demographic drivers (Haines-Young et al., 2010). This study focuses on the direct economic drivers of resource consumption, with simple but credible changes in income from land uses modelled under the proposed CFM conservation intervention in the BME. Two explorative, or probable, scenarios utilised subjective judgements about the drivers of land-use change to illustrate what may happen under a CFM conservation intervention. The third scenario is normative and back-casts from a goal of zero *total* OCs of forest conservation. Scenarios are calibrated with data from peer reviewed and grey literature, research institutions, government sources and non-governmental organisations outlined in Chapter 3, as well as knowledge of the region and intervention gathered through fieldwork. The potential of REDD+ revenues from the project to overcome the OCs of forest conservation is then assessed by applying revenue estimates from Chapter 5.

4.3.3. Environmental attitudes, perceptions and intention to cooperate in CFM

The qualitative study of opinions and perceptions of conservation interventions allows unobservable values to be better understood (Kotchen and Reiling, 2000). At the case study site, open ended questions and agree/disagree statements were included in the household survey described in section 4.3.2.3. A series of open-ended questions also explored opinions of past, present and proposed forest management regimes in the survey villages. Following a description of the intervention, households were also asked if they would take part in CFM as it was proposed. Chapter 8 reports these findings to provide an understanding of local attitudes to resource management and conservation at the case-study site.

Qualitative data complements the empirical estimates of households' OCs of REDD+ via CFM as a household's decision to cooperate is based on them weighing up the costs and the benefits that they perceive they will incur (Lubell, 2002, Adams et al., 2003). The household survey also elicited a household's intention to cooperate in the proposed forest conservation intervention through a voluntary contribution. The voluntary contribution was a portion of their yearly income that they would pay into the CFM cooperative so that it could be used to better manage the forest. This voluntary contribution can be considered a quantitative indicator of intention to cooperate in the proposed CFM intervention. A higher voluntary contribution is assumed to represent greater cooperative intention where cooperation is defined here as a household entering into a scheme, abiding by the rules, and undertaking pro-conservation behaviours.

Other studies have elicited WTP in order to value environmental goods or services through a method called contingent valuation. For example, Köhlin (2001) assesses the WTP for community forest plantations in India. Urama and Hodge (2006) consider WTP for a river basin restoration scheme in Nigeria. Contingent valuation relies on the stated preferences of individuals rather than their preferences revealed through behavioural trails through the elicitation of a value for changes in the level of provision of a good or service through intended action on a hypothetical market (Mitchell and Carson, 1989). Eliciting an individual's WTP to avoid a loss or for a gain, or willingness-to-accept in lieu of a gain or to suffer a loss, contingent valuation is able to capture not only direct-use values, but also indirect, option (potential to be used either directly or indirectly in the future), and non-use values (existence, bequest and altruistic values) (Christie et al., 2008). Despite difficulties in its application in developing countries (Whittington, 1998), contingent valuation has been applied in Ethiopia (e.g. Mekonnen, 2000). In 2009, however, a pilot contingent valuation survey was conducted at the case study site and the value elicitation question was met with either exceedingly high monetary amounts or protest responses, thus contingent valuation was not feasible at the case study site.

As it was unable to observe preferences through a contingent valuation question, a households' voluntary contribution was instead elicited as a quantitative measure of behavioural intention. Such an approach was has been taken elsewhere in the literature. Howe *et al*. (2011) use a voluntary pledge to measure behavioural intention to contribute to a conservation intervention in Russia. Champ *et al.* (1997) considered voluntary contributions to road removal near the Grand Canyon in the United States. In adopting the voluntary contribution approach a number of methodological limitations are acknowledged. In particular a voluntary contribution may not be incentive compatible, free-riding on the donation of others towards a public good could lead to the reduction of donation amounts and free riding on others (Champ et al., 1997). Alternatively, the hypothetical nature of the contribution could lead to inflated donation responses for a warm glow effect (Andreoni, 1989). As a result of the limitations of the measure, the voluntary contribution is not interpreted as a welfare measure, but instead a focus is given to the determinants of households' cooperative intention.

The determinants of a household's intention to cooperate were investigated through regression analysis based on *a priori* assumptions of impact on cooperation established through literature review. The literature on common pool resource and that on common property regimes have explored cooperation through a large body of case-studies. Some have found that wealthier individuals take on more of the burden of initiating collective action (Baland and Platteau, 1999, Bardhan, 2000). In contrast, others have found non-linear wealth impacts on cooperation (Dayton-Johnson and Bardhan, 2002). Many find that the poor bear a higher share of transaction costs and receive lower benefits from access to forest products (Adhikari and Lovett, 2006, Lund and Treue, 2008, Nielsen and Treue, 2012). Appropriator's returns from the forest have been shown to provide material incentives to cooperate (Baland and Platteau, 1999, Lise, 2000, Agrawal and Chhatre, 2006). Adhikari and Di Falco (2009) consider the determinants of access to participatory processes, defined as procedural justice or involvement in decision-making rather than entry to a scheme (Skutsch, 2000, Pascual et al., 2010). Looking at the probability of membership in local forest management institutions in Nepal, Adhikari and Di Falco find that lower-caste groups have lower probability of being elected as members of the committee of user groups. Dayton-Johnson (2000) creates a model of determinants of collective action supported by evidence from Mexican cooperative irrigation systems. The paper finds that cooperation is highly dependent on the distributive rules for cost sharing and water allocation, with social heterogeneity and landholding inequality associated with lower maintenance of irrigation systems.

The literature on cooperation has focussed on the impact of heterogeneity in wealth, interest, and social diversity of resource appropriators. Naidu (2009) summarises that the impact of wealth depends on the relationship between wealth and the returns from the forest resource. Naidu also finds that moderate levels of social diversity lead to low collective management, but high social diversity can lead to high collective management. This study underlines that the impact of wealth, interest and social diversity on CFM success remain mixed (see also Poteete and Ostrom, 2004 for a review). This is complicated by studies using different measures of cooperation, undertaken at differing scales and with a variety of methods. The existing body of literature on cooperation largely considers cooperation ex-post. As Cavalcanti (2010) notes, if factors to improve cooperative self-governance are known they can be actively promoted and that this is particularly relevant where common property regimes are instigated by external actors. This is the case for REDD+ via CFM at the case-study site, hence household’s attitudes and cooperative intention are explored ex-ante.

4.4. Data collection and analysis

4.4.1. Fieldwork permissions

In order to undertake research in Ethiopia, a memorandum of understanding was signed with the Ethiopian Wildlife Conservation Authority of the Ethiopian Federal Government. Permissions were also sought from the Oromia Ministry of Agriculture and Rural Development, the Bale Mountains National Park authorities, and the woreda level Ministry of Agriculture and Rural Development offices. At each survey location permission to conduct surveys was also requested from village leaders after an introduction to the research aims and the fieldwork team had been given. The Economic and Social Research Council provided funding for a total of 61 weeks of fieldwork (inclusive of 14 weeks for difficult language training). The Frankfurt Zoological Society and BERSMP provided further financial and logistical support in-country. The British Embassy in Ethiopia also provided additional finance to undertake forest carbon stock assessment in the (proposed) Bale Mountains National Park.

4.4.2. Fieldwork teams

Primary data for forest carbon stock analysis was undertaken with a team of para-ecologists who were trained how to undertake direct tree measurements. Between December 2008 and April 2009, 49 carbon plots were undertaken. In a second fieldwork period between December 2009 and April 2010 a further 59 carbon plots were inventoried by a smaller team also trained in the same methodologies.

Two enumerators were employed to conduct the household survey on the basis of their English language skills in an attempt to limit information lost in translation.

One enumerator also had prior experience with household surveys and with undertaking research. These enumerators were local to the region, but not to the villages surveyed. Thus they had in-depth local knowledge particularly about local conditions and customs, without creating data sensitivity issues. At each survey location a local liaison officer was also employed to guide the enumerators to households. Enumerators were first trained in the objectives of the research, rationale and objectives, the application of the methodologies, how to approach respondents and the recording of responses. These enumerators were accompanied at fieldwork sites and supervised during questionnaires at intervals. Enumerators recorded responses in data books also reviewed at regular intervals.

4.4.3. Forest carbon plots

Forest carbon plot sampling was based on forest stratification by UNIQUE forestry consultants into: tropical moist degraded forest; tropical moist non-degraded forest; degraded tropical dry forest; degraded woodland; and non-degraded woodland (UNIQUE, 2008). No non-degraded tropical dry forest remains. Carbon stocks were assessed in all forest types except woodlands where allometric relationships were not available for the specific location in the BME. Furthermore, the woodlands will act as a leakage belt under the proposed REDD+ project and will therefore not generate emission reductions for sale. Forest carbon plots were dispersed across the study area, but limited to logistically accessible areas. Logistical limitations of permissions and transport prevented *a priori* calculation of the sample size required to estimate mean forest carbon stocks with a particular level of confidence. However, retrospective power analysis was undertaken to establish the maximum predictive power achieved by the primary data collection.

Drawing on forest inventory protocols (MacDicken, 1997, Pearson et al., 2005, Greenhalgh et al., 2006), tree measurements were collected from a total of 108 forest plots of 20m by 20m (Figure 5). The geo-coordinates of forest plots were identified by overlaying 1km by 1km latitude and longitude grids on maps of the selected study areas, with random number generation used to identify crosshairs representing the centre of forest plots. Plots were then located on foot with a compass and a handheld global positioning system. Within each plot, the diameter at breast height (dbh) – or 1.3 metres above the ground – of all trees was recorded with a lower limit of 5cm dbh was used to define a 'tree' and buttress roots not encountered. In addition to canopy cover, the angle of the slope of the land, altitude and aspect was also recorded. Tree measurements were noted on data sheets, later entered into Excel after which documented allometric relationships were applied to estimate forest carbon stocks.

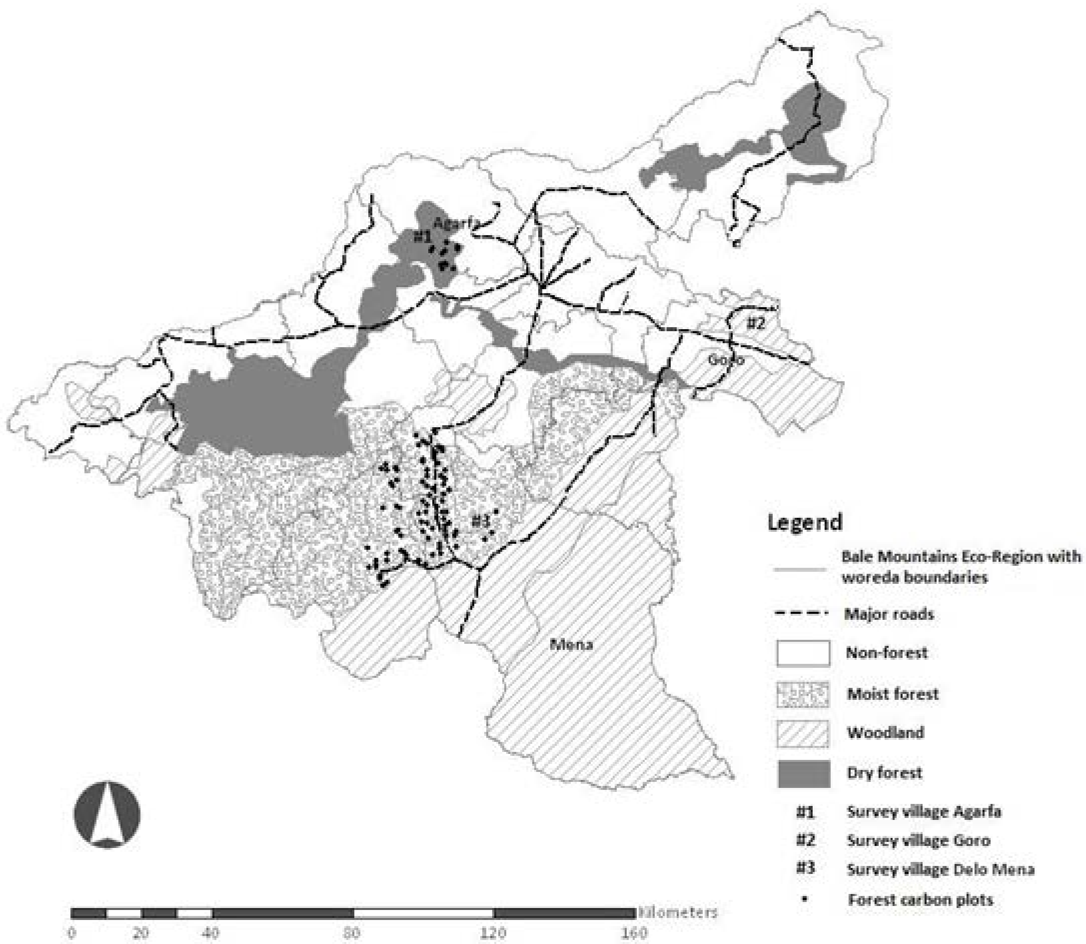

Figure 5. Data collection in the Bale Mountains Eco-Region.
Showing the case study site with *woreda*, or district boundaries and the three household survey locations, Agarfa, Goro and Delo Mena. The three major forest types and location of forest carbon plots are also shown as well as the major roads in the Bale Mountains Eco-Region (BME). *Source: author generated*

4.4.4. Household survey data

The provisional household survey design was informed by discussions with staff of two NGOs involved in the management of the BME resources; Frankfurt Zoological Society and FARM-Africa/SOS-Sahel. Fourteen pilot surveys were also conducted in Dinsho village, where the (proposed) Bale Mountains National Park headquarters are situated. Both discussions and pilot surveys enabled questions to be revised for clarity and ease of understanding, checked for political and cultural sensitivities, and tailored to the dominant activities of the region. Post pilot, the household survey was condensed substantially due to lengthy completion times could impact on results due to respondent fatigue (Angelsen et al., 2011). A by-product of the reduction of the survey is the omission of household composition, which means that the standardisation to adult equivalents is not possible, as well as more detailed information on households with respect to their distance from market and assets such as livestock holdings. Due to the sensitive nature of the question, data on total household income was also removed at the pilot stage.

Between January and April 2010, 237 household surveys were undertaken in three survey villages (see Chapter 3; Figure 5). Given the disbursed nature of households at the household survey villages, respondents were selected opportunistically from walks through town and agricultural fields. It is acknowledged that this non-probabilistic sampling method suffers from self-selection, but was an unavoidable limitation of the survey. Ideally, to reduce bias complete randomisation of households would be achieved given prior knowledge of number and identity of households in the area. This information was not available. A further limitation was that survey respondents were also all male. This was a result of cultural barriers preventing enumerators approaching females within their households. These limitations and their implications are discussed further in Chapter 9.

It had been intended that the biomass needs of households could be established through survey data. Households reported fuelwood and other products in 'donkey loads', however. Although an attempt was made to assess the weights of donkey loads at a major market place, neither sellers no buyers were willing to participate as both sale and purchase is currently illegal: dead firewood can only be collected for home consumption. Furthermore, I was also unable to find consistent or valid estimates of donkey load volumes for the region. As households were also unable to estimate the area of forest which they utilise I was also unable to estimate the area over which forest products were sourced. Even with the forester's rule of thumb of 1m3/ha/year of growth, I was therefore unable to estimate if wood extraction was sustainable. This is a limitation of the thesis that could be conducted in the future to increase the utility of this analysis, and is returned to in Chapter 9.

4.4.5. Market price survey

In the BME, households sell home produce in unrestricted markets, there are no barriers to entry, and on market days many buyers and sellers converge to sell identical produce. As noted in Section 4.3.2.4. there are limitations to the market price approach under imperfect market conditions, however, overestimation was attempted to be minimised as much as possible through surveying local-level markets as establishing shadow prices was not possible.

In order to determine market prices, twelve market price surveys for key forest and crop products were conducted during the household survey period (see Appendix 2). A limitation of the market price survey is that seasonality in prices could not be assessed; field work was restricted to dry season due to transport limitations and lack of all-weather roads. Three market surveys were conducted in major towns and three at survey villages, with two individuals gathering price data at each. Market prices were averaged out over all locations. These market prices were applied to products that households derive from the forest area and to households' crop yields. Income was converted from Ethiopian Birr (ETB) to US dollars at 1 ETB to US$ 0.0749, the average exchange rate of the first quarter of 2010, when the survey was carried out.

4.4.6. Econometric analysis

All data was first entered into Excel, cross-checked to limit input errors and cleaned. Data analysis was carried out using STATA 10 software. In Chapters 6 and 8 ordinary least squares (OLS), Logit and Tobit regression analysis were employed to explore the determinants of household income from forest and agricultural sources, as well as to understand household cooperative intention through a voluntary contribution proxy.

Based on a linear relationship between independent variables and the dependent variable Y_i, OLS regression coefficients are obtained by the minimisation of the sum of the squared error terms assuming homogeneous influence of the independent variables on the dependent variable (Verbeek, 2004; Equation 1). Coefficients are reported in model results throughout.

$$Y_i = X_i \beta + u_i \qquad \text{Eq 1.}$$

where X_i is a vector of the independent explanatory variables
and $X_i \beta = E(Y_i \mid X_i)$

OLS assumes an error term u_i with normal distribution which is unlikely in cross-sectional data. Heteroskedasticity robust standard errors were therefore estimated as the square root of White's variance estimator (Carter-Hill et al., 2007). In order to test OLS model specification a Ramsay Regression Equation Specification Error Test (RESET) was used post-estimation. This tests whether the functional form is incorrect, for example, if non-linear combinations of the estimated values explain the endogenous variable, and is designed to detect omitted variables (Carter-Hill et al., 2007).

In Chapter 6, Logit and Tobit regression models were used to analyse the determinants of household income from low-impact forest products. This was necessary as 50% of the households in the sample did not have income from these forest products, but where they did the income was a continuous random variable with positive values. Both the Logit and Tobit suppose a latent variable y_i* which remains dependent on x_i (Verbeek, 2004; Equation 2).

$$y_i^* = X_i \beta + u_i \qquad \text{Eq.2}$$

Where the observed y_i is defined by:

$$y_i = \begin{cases} y_i^* & if y_i^* > 0 \\ 0 & if y_i^* \leq 0 \end{cases}$$

The Logit model predicts the probability of the occurrence of an event and is therefore a binomial model. In Chapter 6 the Logit model predicts the presence or absence of income from low-impact forest product income. We observe $y_i = 1$ if low-impact forest product income is derived, thus if $y^*_i > 0$ and $y_i = 0$ otherwise. Therefore, the response y_i is binary and a realisation of random variable Y_i and takes the value of one and zero with probability p_i and $1-p_i$, respectively. The Tobit model is a censored version of the regression model. The Tobit model supposes a latent variable y_i* is only observed for values greater than 0 and censored otherwise. In Chapter 6 the Tobit predicts the probability of being above the censor and the determinants of low-impact forest product income if income is greater than zero. The estimation of both the Logit and Tobit model is achieved through maximum likelihood estimation. Assuming a distribution, parameter values are estimated as those that give the observed data the highest probability (Verbeek, 2004).

The explanatory variables used in regression models were selected based on their predicted impacts on the dependent variable. Thus cause and effect relationships are based on *a priori* assumptions. This is opposed to an ecological or more natural sciences reductionist approach where non-significant explanatory variables are eliminated in a step-wise approach (Armsworth et al., 2009). The cross-sectional data-set is limited given that there are three survey locations with three differing forest types. Effects due to forest characteristics and village characteristics, therefore, cannot be separated. A village dummy variable was included to encapsulate these differences to help control for unobserved but constant variation across survey locations. These village fixed effects should provide consistent estimates even in the presence of correlation between village-specific heterogeneity, which is time invariant, and the right hand side variables. A correlation matrix was assessed pre-estimation to assess the possibility of multicolinearity – where a linear relationship between explanatory variables gives an unreliable regression estimate – as the individual impact of each variable is hard to determine (Verbeek, 2004).

Chapter 5: Uncertain emission reductions from forest conservation

5.1. Introduction

5.1.1. Problem statement

Assessing the decrease in emissions from projects or policies impacting on forests still contains substantial uncertainty despite a global proliferation of REDD+ activities. This emission reductions accounting is necessary to illustrate both climate change mitigation potential of forests, as well as monitoring progress towards climate change mitigation targets through forest conservation activities. Emission reduction estimates are therefore necessary irrespective decisions to be made on the ultimate financing mechanism of REDD+ under the United Nations Convention on Climate Change (UNFCCC) (see Section 2.1 for a discussion of financing options for REDD+). Under a project-based approach to REDD+, however, emission reductions accounting is critical. These sub-nationally implemented REDD+ projects generate lessons for future REDD+ implementation, with a view to trading emission reductions in voluntary carbon markets (see Section 4.3.1 for a discussion on REDD+ and the voluntary carbon markets). Thus the revenues available to alter economic incentives for forest conservation in such REDD+ projects will be dependent on the market value of the emission reductions and the costs of getting them to market.

Emission reductions accounting requires the quantification of forest area, forest area change and forest carbon stock. Advances are being made in the technology and accessibility of remote sensing imagery for the measurement of forest area and forest area change and it is being increasingly used to infer forest biomass and so foret carbon stocks (Achard et al., 2004, Mayaux et al., 2005, DeFries et al., 2007, Ramankutty et al., 2007, Baccini et al., 2008, Goetz et al., 2009, Bucki et al., 2012).

Baker *et al.* (2010) report that remote sensing is mature enough to use in national systems of forest cover monitoring systems; although more research could still improve accuracy and detail of this imagery. Romijn *et al.* (2012), however, found capacity gaps for forest monitoring for REDD+ still existed in many forested nations and particularly in Africa and many countries lacked resources and expertise to make the most in advances in satellite imagery technology, for example.

Appropriate methods to establish the past and predicted rates of forest change in order to calculate the emission reductions resulting from an intervention continue to be developed (Angelsen, 2008, Olander et al., 2008, Bond et al., 2009, Griscom et al., 2009, Huettner et al., 2009, Estrada, 2011). For project-based REDD+, standards have emerged that set out detailed methods and procedures, including for the establishment of baselines (Estrada and Joseph, 2012). The Voluntary Carbon Standard (VCS) is the most commonly applied in voluntary carbon markets, and price premiums can be received for emission reductions registered to the VCS and other carbon standards (e.g. VCS, 2007, CCBA, 2008).

This Chapter focusses on the third aspect of emission reductions accounting; forest carbon stocks. Forest carbon stock refers to the carbon content in the dry biomass of a forest per unit area, often measured in tonnes of carbon per hectare (UNDP, 2009). High uncertainty in forest carbon stock estimates often results from a lack of data on key forest variables and parameters, resources or capacity (Brown et al., 1989, Smith and Heath, 2001, Andersson et al., 2009). Changes in the estimates of forest carbon stock in the FAO Forest Resource Assessment, a widely used database of global and national forest statistics, for example, are found to have changed due to information availability rather than stock changes (Houghton, 2005). In 2009, a technical paper of the UNFCCC considering the costs of monitoring systems for REDD+ indicated that a number of developing countries have insufficient capacity to undertake forest monitoring and mapping; inclusive of forest carbon stocks (UNFCCC, 2009). Three years later, Romijn *et al.* (2012) found similar conclusions to the UNFCCC in their assessment of the status and development of monitoring capacities for REDD+, also identifying that Africa suffers the greatest capacity gap.

As popularity in REDD+ has grown, so has literature on the estimation of forest carbon stocks and uncertainty in forest carbon stocks as they pertain to a REDD+ mechanism (Houghton and et al., 2001, Houghton, 2005, Mollicone et al., 2007, Ramankutty et al., 2007, Pelletier et al., 2010). The uncertainty of forest carbon stocks has also been demonstrated through global and regional forest carbon stock mapping efforts. Saatchi *et al.* (2011), for example, produced a global map of forest carbon stocks through satellite imagery and on-the-ground forest plots. Propagating errors through the estimation process they found uncertainty in forest carbon stocks of 38% over Latin America, sub-Saharan Africa and Southeast Asia; although the analysis was not applied at a country level. Work is ongoing to improve forest carbon stock estimates; Le Toan *et al.* (2011) outline an ongoing initiative to map global biomass, of which approximately 50% is carbon, with error not exceeding 20%.

As a result of lack of data at finer resolution at national and sub-national scales, the application of biome-averaged forest carbon stock data to estimate emission reductions has, therefore, become widespread where data on forest carbon stock is not available locally (Brown and Gaston, 1995, Gibbs et al., 2007, Djomo et al., 2010). The Intergovernmental Panel on Climate Change (IPCC) have compiled best available methods and published guidance and guidelines for countries to undertake GHG inventories and to identify the emissions and removals of GHGs from land use, land-use change and forestry activities (IPCC, 2003, IPCC, 2006). IPCC guidance is intended to promote broad engagement of countries irrespective of their data sets and capacities to manipulate this data (see Baker et al., 2010). As a result there are three Tiers of methods with increasing levels of uncertainty, with countries selecting Tiers based on data requirements and methodological complexity. While Tier 3 uses advanced estimation approaches that involve complex models and highly disaggregated data, Tier 2 employs more country-specific carbon stock information and requires activity data disaggregated to smaller scales, and Tier 1 is based on biome-averaged data for carbon stocks (Böttcher et al., 2009).

Biome-averaged data used in Tier 1 is able to capture broad ecological variables influencing forest carbon stocks, such as temperature and rainfall (Chave et al., 2004, GOFC-GOLD, 2008), but it obscures substantial local forest heterogeneity (Houghton and et al., 2001, Bradford et al., 2010). An emission reductions estimate using this simple accounting method is, therefore, likely to contain more uncertainty than applying more complex and data intense methods which statistically relate measured forest attributes to above-ground carbon stock using allometric relationships (Brown, 1997, Chave et al., 2005). Comparisons across six countries by GOFC-GOLD (2008) found that application of biome-averaged defaults overestimated forest carbon stock as much as 33% in Mexican temperate forest and underestimated by as much as 44% in African rainforest when compared to plot measurements. The uncertainty introduced by carbon accounting methods is non-trivial, but the magnitude and direction of the discrepancy so far varies from case to case.

While IPCC guidance was not designed to produce emission estimates for REDD+ projects, the UNFCCC has supported the use of guidance by countries for REDD+

(UNFCCC, 2009). A large discrepancy between the use of simple and complex forest carbon stock methods in estimating emission reductions could be the difference between making a decision to implement a REDD+ project or not. However, there is no standardised method to assess or communicate uncertainty in emission reductions accounting. Where carried out, uncertainty assessments have relied upon published information and expert judgement. Less commonly measurement data has been used and total uncertainty quantified through propagation of error and Monte Carlo simulation methods (Heath and Smith, 2000, Smith and Heath, 2001, IPCC, 2003, Peltoniemi et al., 2006, Monni et al., 2007).

The principle of conservativeness remains a dominant approach to dealing with uncertainty in emission reductions accounting (Mollicone et al., 2007, Grassi et al., 2008). The principle of conservativeness requires omitting carbon pools or taking lower bound estimates to ensure a low probability that carbon emission reductions are overestimated (GOFC-GOLD, 2008). However, conservativeness assumes zero uncertainty and decision-makers are left without an idea of the confidence interval of estimate of emission reductions (Andersson et al., 2009). Attempts are being made to communicate the uncertainties of emission reductions accounting to policy-makers and to aid decision-making (Brown, 2002, Andersson et al., 2009, Waggoner, 2009). Kerr *et al.* (2004), for example, quantitatively translate errors in estimating carbon stocks into environmental integrity of emission reductions for avoided deforestation in their assessment of potential emission reductions in Costa Rica, finding that uncertainty is impacted strongly by forest type; particularly in tropical wet forest. Pelletier *et al.* (2010) used five carbon stock estimates for Panamanian forests in land conversion and transition models, finding 144% difference in emission reductions resulted from highest to lowest. Acceptance of Tier 1 accounting, however, remains high.

Feasibility studies for project-based REDD+ will often combine uncertain forest carbon stocks with uncertain market variables. With a 20-100 year project lifespan, a feasibility assessment for a REDD+ mechanism requires assumptions and best-guesses to be made regarding voluntary carbon market price trends, implementation and transaction costs. Using a back of the envelope calculation Pelletier *et al*. (2010) take their emission reduction accounting using five forest carbon stock estimates further, by demonstrating that break even prices for emission reductions were more than twice as high with lowest global default forest carbon stocks as compared to local forest carbon stock estimates in Panama.

There are a number of REDD+ projects and activities emerging in Sub-saharan Africa (Diaz et al., 2011, Climate Funds Update, 2011, Forest Carbon Portal, 2012). The Kasigua Corridor REDD+ Project in Kenya, run by Wildlife Works, for example, has been generating emission reductions since 2005 and has been exemplary in being the first REDD+ project to deliver validated, verified and issued VCS certification emission reductions (Wildlife Works, 2012). Other countries in East Africa are following this example, but Africa suffers substantial data gaps for forest carbon stocks (Glenday, 2006, FPAN, 2010, Mustalahti et al., 2012, Romijn et al., 2012). While simple accounting methods can be, and often are, applied to calculate emission reductions potential in REDD+ feasibility studies, complex accounting methods are applied during project development and to meet carbon standards (Shoch et al., 2011). Resulting discrepancies in emission reductions between these estimates are likely to erode the credibility of a REDD+ project. It may not, therefore, be surprising that expectations of wealth transfer through REDD+ mechanisms have been high but not always forthcoming (Clements, 2010).

REDD+ may not be suitable to overcome the opportunity cost of private incentives driving deforestation in all situations. For REDD+ projects, if estimated revenues are insufficient to meet cost demands of REDD+ then other tools to fund forest conservation should be considered (Fisher et al., 2011). Conversely, climate change mitigation potential is lost where emission reductions are more substantial than a feasibility assessment would indicate. Uncertainty in emission reductions accounting must be quantified, reduced where possible, and communicated more appropriately (Waggoner, 2009, Baker et al., 2010).

5.1.2. Aims and objectives

Using a proposed REDD+ project in the Bale Mountains Eco-Region (BME) of Ethiopia, this paper quantifies the discrepancy between simple and complex forest carbon stock methods to estimate emission reductions. It then explores the potential REDD+ revenues under uncertainties in both forest carbon stock and market variables and the resultant implications for project implementation at the case study site. This paper adds to current knowledge through the collection of primary forest data and calculation of forest carbon stock in the BME. It also builds on a limited literature on the financial implications of emission reductions accounting discrepancies as well as implications on the environmental integrity of REDD+ projects.

5.2. Methods

5.2.1. Assessing carbon stocks and estimating emission reductions

In the BME of Ethiopia a REDD+ project is being developed by the Oromia Forest and Wildlife Enterprise (OFWE), with the support of the Bale Eco-Region Sustainable Management Program (BERSMP): a joint NGO program between

FARM-Africa and SOS Sahel Ethiopia (see Chapter 3 for a full project description). Ethiopia is not well known for REDD+ activities and East African forests are often characterised by miombo and acacia woodland; thus they are not as dense as rainforest of the Congo Basin or West Africa. The dry and moist, montane forests of East Africa, however, are gaining prominence for REDD+ project activities (see FPAN, 2010, Diaz et al., 2011).

The proposed REDD+ project lies in the south eastern Ethiopian Highlands in Oromia Regional State between 50˚22'-80˚08'N and 38˚41-40˚44'E. The annual temperature of the Bale zone is 17.5˚C ranging from 10˚C to 25˚C, with annual rainfall of 875mm experienced in one long season between June and October, and one short rainy season between March and May (Yimer et al., 2006). Moist tropical forest is found between 2600 masl and 1500 masl, characterised by *Hagenia abyssinica* and wild coffee (*Coffea arabica*). North of the plateau habitats comprise of dry forest, woodlands, grasslands and wetlands, largely between 2500 masl and 3500 masl. The dry forests contain high-value commercial species such as *Juniperus procera* and *Podocarpus falcatus* as well as *Prunus africanus,* a threatened species. The lower altitude land of the south east of the BME, below 1500 masl, is dominated by acacia woodland (Teshome et al., 2011, UNIQUE, 2008).

The BME has deforestation rates four times the national average at 4% losses in forest area annually (Dupuy, 2009). Ethiopia is also in the top ten countries for forest loss in tropical Africa (FPAN, 2010). The main drivers of deforestation and forest degradation in Ethiopia are small scale conversion to agriculture, large scale conversion to agriculture, and unsustainable forest management (R-PP, 2011). This pattern of exploitation is consistent over the BME, with rural communities rapidly deforesting to procure land for crops and livestock grazing and to meet livelihood needs through timber and firewood extraction (BERSMP, 2006, BMNP, 2007).

To address the decline in forest area, the Oromia Forest and Wildlife Enterprise (OFWE) are implementing CFM across all forests of the BME. Therefore, CFM is regarded here as a mechanism to implement the REDD+ project, alongside the creation of 15,000 hectares of woodlots and fuel efficient stoves to reduce household wood fuel demands (see Chapter 3 and Chapter 9 for a discussion on REDD+ via CFM). The project is in the early stages of development and secondary data in this paper is based on an early feasibility studies by forestry consultants UNIQUE (UNIQUE, 2008, UNIQUE, 2010). The estimates of required area for woodlots to meet household demands, however, are based on their expert judgement rather than through assessment of biomass needs per households.

The project area covers 923,593 hectares, of which 60% is dry and moist tropical forest, the REDD+ project aims to reduce deforestation to 1% a year by project-year 20 within this area. The decline in deforestation is predicted to be gradual as the project is implemented, with rates of deforestation slowed to 3% in years 1 to 5, 2% in years 6 to 10, and 1% in years 11 to 20. REDD+ revenue is generated from avoided deforestation only on dry and moist forest. The area of avoided deforestation amounts to 5,769 ha/yr in years 1-5, 11,537 ha/yr in years 6-10 and 17,306 ha/yr in years 11 to 20. This amounts to 259,585 ha of avoided deforestation over the project lifespan. Although emission reductions generated on woodland are not sold, they still must be generated, thus the area of avoided deforestation including dry forest, moist forest and woodland amounts to 9,236 ha/yr in years 1-5, 18,472 ha/yr in years 6-10 and 27,708 ha/yr in years 11 to 20: a total of 415,617 ha.

As reported across wider Africa, local estimates of forest carbon stocks for use in modelling emission reductions from REDD+ in Ethiopia are few, and what exists is wide-ranging (FPAN, 2010). The IPCC present an Africa specific forest carbon stock estimate of 122tC/ha in tropical moist forest and 56tC/ha in tropical dry forest, as well as estimates based on ecological zones of 85tC/ha in tropical moist forest and 61tC/ha in tropical dry forest (IPCC, 2006based on converting biomass to carbon using 0.47 carbon fraction of biomass). Gibbs *et al.* (2007) reviews forest carbon stock estimates across forest types in Africa with estimates in the range of 30 to 200tC/ha. A later study estimated forest carbon stocks in Africa between 0 and 454tC/ha, although only three countries are used to produce this estimate; Republic of Congo, Cameroon and Uganda (Baccini et al., 2008). Lewis *et al.* (2009) estimated forest carbon stocks from permanent plots across Africa with average of 202 tC/ha. Ethiopia's national average forest carbon stocks have been reported at 37tC/ha and 47tC/ha (FAO, 2000, Brown, 1997). The national forest inventory of Ethiopia, however, is criticised for conflicting data (Teketay et al., 2010) and no estimates of forest carbon stock are known by the author for the BME. The country-wide estimate, however, is predicted to underestimate forest carbon found in the BME REDD+ project area as a result to Ethiopia's wide-ranging topography.

Three forest carbon stock estimates were used to model emission reductions:

1. Ecological zone specific forest carbon stock from the IPCC Land Use, Land-Use Change and Forestry (LULUCF) Good Practice Guidance (IPCC, 2003)
2. Africa specific forest carbon stock from the IPCC Agriculture, Forestry and Other Land Use guidelines (IPCC, 2006)
3. Primary estimate of forest carbon stock reliant on field sampling of above-ground tree biomass in the BME.

The application of default data from the Intergovernmental Panel on Climate Change (IPCC) illustrates simple forest carbon stock methods, whereas primary data collection in the moist and dry tropical forest of the BME represents more complex forest carbon stock methods.

The annual emission reductions generated by the BME REDD+ project can be represented by Equation 3. Where $ER_{t,i}$ are emission reductions in tons of carbon dioxide (tCO_2) in year t, utilising forest carbon stock estimate C_i (tC/ha) where i can take the value of 1, 2 or 3, representing the three forest carbon stock estimates used to model emission reductions. D_{BAU} is the annual business-as-usual (BAU) deforestation in a without project baseline in hectares; D_{REDD} the area of deforestation (ha) during the project in year t; and *44/12* is the ratio of the molecular weight of carbon dioxide to that of carbon.

$$ER_{t,\,i} = C_i \left(D_{BAU} - D_{REDDt} \right) \frac{44}{12} \qquad \text{Eq. 3}$$

The annual area of deforestation under a BME REDD+ project baseline, D_{REDD}, is based on project goals to reduce deforestation below the annual BAU baseline in three stages. In years 1 to 5 D_{REDD} is 3% as compared to D_{BAU} of 4%, in years 6 to 10 D_{REDD} is 2%, and in years 11 to 20 D_{REDD} is 1%. The total emission reductions generated by the project, $E_{project,\,i}$ (tCO_2), can be represented by Equation 4 which sums annual emissions over the 20-year project lifespan.

$$ER_{project,\,i} = C_i \left(20\, D_{BAU} - \sum_{t=1}^{20} D_{REDDt} \right) \frac{44}{12} \qquad \text{Eq. 4}$$

5.2.1.1. Forest plots and carbon stocks

Primary data collection was focussed on the above-ground tree biomass carbon pool. Containing the greatest fraction of total living biomass in a forest, this pool is most immediately impacted by deforestation and degradation (Brown, 1997, FAO, 2003). This estimate therefore omits below-ground carbon in tree roots, soil organic carbon, and that contained in dead wood and litter. The forest was stratified using satellite imagery by UNIQUE forestry consultants into: tropical moist degraded forest; tropical moist non-degraded forest; degraded tropical dry forest; degraded woodland; and non-degraded woodland. No non-degraded tropical dry forest remains (UNIQUE, 2008). Carbon stocks were assessed in all forest types except woodlands, which will act as a leakage belt under a REDD+ project and will not generate emission reductions for sale.

Drawing on forest inventory protocols (MacDicken, 1997, Pearson et al., 2005, Greenhalgh et al., 2006), data were collected from 108 forest plots of 20m by 20m between December 2008 and April 2010 (see Figure 5, Chapter 4). Plots were dispersed across the study area, but limited to logistically accessible areas and regions for which permissions to undertake field sampling was granted by the Ministry of Agricultural and Rural Development and village elders. The geo-coordinates of forest plots were identified by overlaying 1km by 1km latitude and longitude grids on region maps, with random number generation used to identify crosshairs representing the centre of forest plots. Plots were then located on foot with a compass and a handheld global positioning system (Figure 6). Within each forest plot, the diameter at breast height (dbh) – or 1.3 metres above the ground – of all trees was recorded. Butress roots that obstructed dbh measurements were not encountered. In addition to dbh, canopy cover, the angle of the slope of the land, altitude and aspect were also measured. Tree saplings with dbh less than 5cm dbh were not measured; a lower limit of 5cm dbh was used to define a 'tree'.

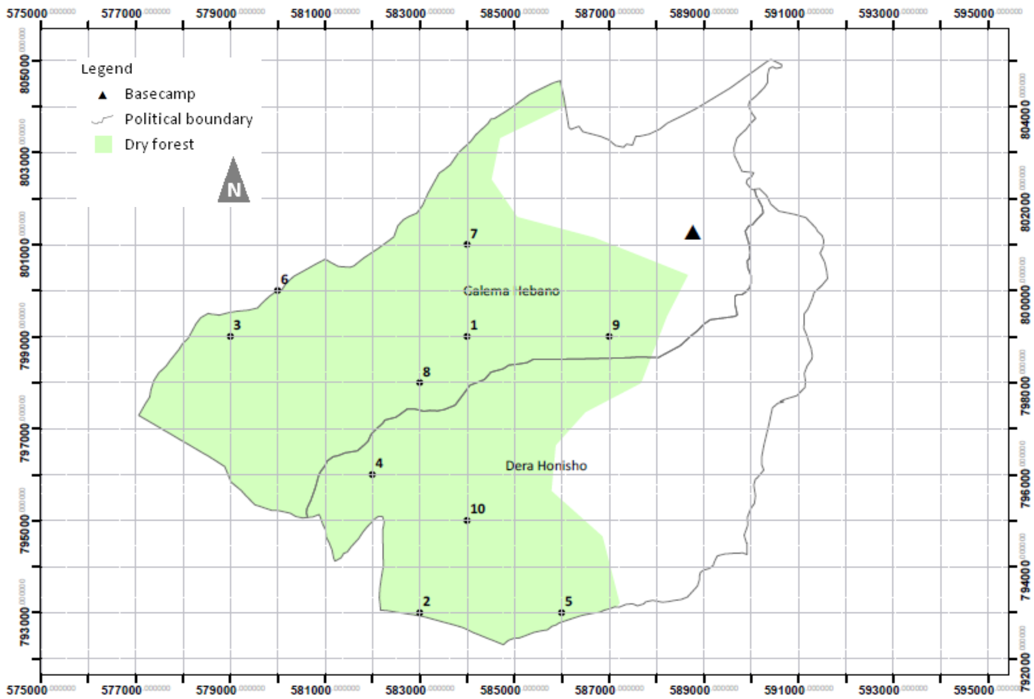

Figure 6. Example forest plot selection for degraded dry forest of Argafa.
In selecting forest plots a grid was overlaid on a forest map and crosshairs numbered. Random number generation then determined the geo-coordinates of sampling points indicated by the numbered points. The points were then located by compass and handheld global positioning system. This map illustrates the plots located in Galema Hebano and Dera Honsho, both areas within Agarfa.

5.2.1.2. Power analysis

Logistical limitations of permissions and transport prevented *a priori* calculation of the sample size required to accept the estimate of mean forest carbon stocks with a particular level of confidence. Retrospective analysis following Pearson *et al*. (2005) was instead undertaken to illustrate the minimum number of plots required for the mean forest carbon stock estimate to be within an error bound of 20% of the mean with 95% probability. The error bound is that within which the mean can be found with probability $1\text{-}\alpha$, where α is the probability of rejecting the null hypothesis when it is true (or Type I error). The error bound E of the forest carbon stock estimate can be calculated using $E{=}C\beta$ where C is the estimate of forest carbon stock and β is the precision: the half-width of the allowed error interval around the mean expressed as a percentage of the mean. The precision is also the probability of not rejecting a false null hypothesis (or a Type II error), and from this the probability of finding a difference that does exist, or the power of a statistical test, can be calculated using *1-β*.

This retrospective power analysis can, therefore, establish the maximum predictive power achieved by the primary data collection and estimation of forest carbon stocks. Given that the total project area, total size of each forest strata, the forest plot area, and standard deviation of carbon stocks for each stratum is known, for L strata, n plots are required and can be calculated in Equation 5.

$$n = \frac{\left(\sum_{j=1}^{L} N_j \times s_j\right)^2}{\frac{N_{TOT}^2 \times E^2}{t^2} + \left(\sum_{j=1}^{L} N_j \times s_j^2\right)} \qquad \text{Eq. 5}$$

where, N_j is the maximum number of sample forest plots in stratum *j*, s_j is the standard deviation for stratum *j*, N_{TOT} is the maximum number of sample plots in the project area, *E* is the allowable error or the half-width of the desired error interval (calculated by multiplying mean carbon stock by the desired precision in percentage), and *t* is the sample statistic from the t-distribution for the 95% confidence level.

Once n has been determined, the required distribution of plots across strata is shown by Equation 6. The actual number of plots sampled can then be compared to the number of plots estimated by this power analysis to achieve a forest carbon stock estimate with at least 20% precision with a 95% probability.

$$n_j = n \times \frac{Nj \times sj}{(\sum_{j=1}^{L} N_j \times s_j)} \qquad \text{Eq. 6}$$

5.2.1.3. Biomass regression equations

To determine carbon stocks, the above-ground biomass was first estimated per tree. This required the application of mixed species broadleaf regression equations, or allometric equations, for dry and moist tropical forest to dbh measurements.

Few allometric equations exist in Sub-Saharan Africa for estimating biomass in trees and woodland (Henry et al., 2011, Shackleton and Scholes, 2011). Where available, site and species specific allometric equations allow better biomass estimation as they better capture heterogeneity in forest characteristics. While 95% of the variation in the above-ground tropical forest carbon stocks of trees can be explained by dbh (Brown, 2002), studies indicate that using measurements of tree height and wood density – the dry weight per unit volume of wood – in allometric equations can improve biomass estimates (Brown et al., 1989, Chave et al., 2005, van Breugel et al., 2011, Marshall et al., 2012). Height is often difficult to measure accurately in tropical forest, however, and studies of tropical forests often omit this variable. Henry *et al.* (2011) reviewed 850 allometric equations for Sub-Saharan Africa, finding only 15% of African allometric equations use height. The review also indicated that allometric equations do exist in Ethiopia, but many were for single species forests rather than mixed forests, and equations were of varying quality.

Pan-tropical allometric equations were applied to estimate biomass (Brown, 1997; Table 4). Destructive tree sampling to generate site-specifc allometric equations was not feasible at the case study site. Height measurements were also impractical with time restraints, and wood density estimates did not exist for the area. These allometric equations are not applicable to woodland and forest carbon stocks in woodland are therefore not estimated. Woodland allometry is under development, however, and Shirima *et al.* (2011) estimated mean above ground carbon storage in Tanzania's miombo woodlands to be between 13-30tC/ha.

Trees with diameters that exceeded the upper limit of the range used to create the regression equations were restricted to 148cm dbh as Chave *et al.* (2005) found that tree allometry is conserved across sites in different continents and so regression models should be applicable in all forests, within their range of validity as determined by the maximum and minimum tree dbh used to generate the equation. Of 2698 measured living trees, 12 exceeded the limit of 148 cm dbh, implying the resultant carbon stock estimate may be an underestimate of forest carbon stock. Given that large diameter trees account for a large proportion of above-ground biomass (Brown, 2002), it is acknowledged that this is a further source of uncertainty that could be substantial.

Table 4. Biomass regression equations applied to direct tree measurements in order to establish the above-ground tree biomass in forest plots.
These equations are valid when applied to trees within the range of diameter at breast height (dbh) of trees used to generate the equations (sourced from Brown, 1997).

Climatic Zone	Equation	Range in dbh (cm)
Moist Forest	$Y = \exp(-2.134+2.530\ln(dbh))$	5-148
Dry Forest	$Y = \exp(-1.996+2.32\ln(dbh))$	5-40

Tree biomass was converted to carbon using a carbon fraction of 0.47 (IPCC, 2006). Forest carbon stock per hectare was established by adjusting plot areas for their average slope angle using *cos (slope)*. This slope correction is necessary as forest area is estimated without taking topography into account. This correction improves the likelihood that each quadrat contained the same total area as seen on a two-dimensional satellite image.

The weighted mean of forest carbon stocks in non-degraded moist, degraded moist and degraded dry forest were calculated. Forest plots were randomly resampled with replacement 1000 times to obtain an empirical bootstrap distribution for forest carbon stock (Efron, 1979, Guan, 2003). Bootstrapped confidence intervals were established using the percentile method; where the 2.5 and 97.5 percentiles constitute the limits of the 95% confidence interval.

While natural variation in the forest ecosystem will always result in some uncertainty, in the final estimate of forest carbon stock, uncertainty arises from sampling error, measurement error, and that inherent in underlying equations and assumptions. Table 5 identifies these sources of uncertainty and the methods applied in this study to reduce uncertainty. This study focussed on sampling error as errors in measurement. The application of the allometric equation, and the ratio of biomass to carbon has been addressed elsewhere in the literature (Clark and Clark, 2000, Keller et al., 2001, Ketterings et al., 2001, Chave et al., 2004).

5.2.1.4. Estimating emission reductions

Estimated forest carbon stocks were utilised to estimate emission reductions, evaluated by the difference between a BAU deforestation baseline and an avoided deforestation REDD+ project baseline. The establishment of a BAU baseline relies on forest area and area change data as well as predictions of future drivers of deforestation. As noted in Baker *et al.* (2010), the IPCC guidance was not developed with REDD+ in mind and therefore does not deal with estimating a baseline (nor issues of leakage, additionality, and permanence). Methods to establish this deforestation baseline are controversial and have been discussed in length elsewhere in the literature (see Olander et al., 2008, Huettner et al., 2009).

Table 5. Inputs and sources of uncertainty in estimates of forest carbon stock as well as methods applied to reduce these uncertainties as far as possible.

Input	Source of uncertainty	Method to reduce uncertainty
Selection of forest plots	Sampling error	Forest plot geo-coordinates were selected using random number generation, but within logistical constraints, and good practice for sampling design and forestry inventory was followed (MacDicken, 1997, Pearson et al., 2005, Greenhalgh et al., 2006, Grassi et al., 2008).
Measurement of dbh	Measurement error	Training and education in measurement of dbh was conducted to reduce measurement error. It was ensured that trees were not measured twice or dead trees counted as living. Measurement uncertainty on a single tree of diameter 10cm or greater has been estimated at 16%, but found to average out at forest stand level (Chave et al., 2004) and so it is not addressed in this study.
Application of allometric equation	Estimation error: allometric equations originating from Asian and Latin American data	Allometric uncertainty is not addressed here. Although acknowledged as a potential source of error, pan-tropical equations are based on a large number of trees that span a range of dbh. As the destructive sampling of a sufficient number of trees to create an area-specific allometric regression equation was not possible, their application is appropriate. The dbh was, however, restricted to values used to create the regression equations. Error due to the application of the allometric equation is estimated at 10-20% and can be amplified where large trees are numerous (Clark and Clark, 2000, Keller et al., 2001, Ketterings et al., 2001, Chave et al., 2004).
Application of ratio of biomass to carbon	Estimation error: the carbon content of biomass components and tree species differ	The error of the carbon fraction is not addressed here. The IPCC (2006) present a default value of 0.47 for tropical and sub-tropical forest, but within an interval estimate of 0.44-0.49. This is an improvement on 0.5 suggested by Westlake (1966), but suggests relative error of 5%.

The BAU deforestation baseline for the BME was generated from GIS imagery analysed by the BERSMP and assumes a linear deforestation rate of 4% in all forest types. The uncertainty of this rate of loss is dependent on the resolution of GIS imagery and method of image analysis, as well as assumptions regarding the rate and location of changes in forest area in the future. Research has shown that forest area data based on satellite imagery has accuracies between 80% to more than 99% (Achard et al., 2001, DeFries et al., 2007, Grassi et al., 2008, Gonzalez et al., 2010). In the (proposed) Bale Mountains National Park the accuracy of estimation of forest area follows these findings, ranging between 81% and 97% (Teshome et al., 2011).

Deforestation resulting during implementation of the BME REDD+ project is based on stated project goals to reduce deforestation. These predictions of are subjective rather than based on past experience of intervention. Until the project in the BME progresses, estimating how much deforestation can be reduced as a result of REDD+ policies, actions and measures will continue be uncertain and models of emission reductions will need to be revised regularly as new information is acquired.

It is acknowledged that this carbon accounting exercise is a static representation of the forest ecosystem in the BME. The methodology contains an inherent assumption of a steady state in mature forest. This assumption is still under debate and there is no easy way to assess if this is the case (Phillips et al., 1998, Houghton, 2005, Bonan, 2008, Grote et al., 2011). Anthropogenic impacts on carbon stocks of forests are also ongoing and non-linear, which makes it difficult to differentiate between inter-annual variability in the forest ecosystem and indirect feedbacks from direct human activities (UNDP, 2009).

In the case of avoided deforestation, it is common to assume that all carbon in biomass would be emitted to the atmosphere at the time of forest loss. It is possible that this may overestimate emission reductions where harvested wood products (HWPs) are manufactured (Lim et al., 1999, Karjalainen et al., 1999). In the BME, HWPs are used in construction of housing and farm implements, however, as no consensus on accounting for HWPs exists (Winjum et al., 1998) the assumption that all carbon is emitted during deforestation is necessary.

This study does not add to the debate on the definition of forest which varies by country. A lower limit of 5cm dbh was used to define a 'tree' and 'forest' was determined by the authors, and so no assumptions on canopy cover were made. The implications forest definition on the BAU deforestation baseline, and on emission reductions generated through forestry carbon activities and distribution of REDD+ funds between countries is addressed elsewhere (Neef et al., 2006, Zomer et al., 2008, Meridian Institute, 2009). The limitations of this study further highlight the research needs within forest carbon stock methods and for even more complex emission reductions accounting with advanced estimation approaches that involve complex models and highly disaggregated data on key forest carbon stocks through time (IPCC, 2006, Böttcher et al., 2009).

5.2.2. Estimating revenues and REDD+ rent

Taking the best- and worst-case emission reductions estimates, the potential revenues of the BME REDD+ project were calculated. The discounted REDD+ revenue can be expressed by Equation 7 where: π_i is the profit in 2010 US$ over the 20-year lifespan of the REDD+ project utilising forest carbon stock estimates denoted by subscript i; $E_{t,i}$ are the emission reductions generated by the project in year t (tCO_2); B is the buffer of emission reductions expressed as a proportion; p is the price per ton of CO_2 in US$; r is the registry cost per ton of CO_2 in US$; A are the annual operating cost of the project in US$; δ is the discount rate; and, K is the upfront costs (US$) of project establishment experienced in project year 1.

$$\Pi_i = \left(\sum_{t=1}^{20} \frac{\left(E_{t,\,i}\left(1-B\right)\left(p-r\right)\right)-A}{\left(1+\delta\right)^{t-1}} \right) - K \qquad \text{Eq. 7}$$

Not all emission reductions generated by the project activities can be sold. Forestry carbon projects must account for the risk that emission reductions will not persist over time. Termed 'permanence', it is possible that forest carbon stocks could deteriorate or be depleted over time due to natural disturbances such as fire, pests and disease, or anthropogenic disturbances such as political instability leading to land-use change (see Sedjo and Marland, 2003). No assumptions are made regarding the liability for non-permanence, however see Chapter 9 where this is discussed in the context of the BME REDD+ project. In addition, leakage might relocate emissions outside of the woodlands and REDD+ project area. To deal with non-delivery risks of permanence and leakage, a non-tradable buffer, or reserve, of emission reductions is commonly set-aside as insurance (Peskett and Harkin, 2007).

In this study, 40% of emission reductions were set-aside for non-permanence, and a further 25% of emission reductions were set-aside in case of leakage. These buffers are at the higher ranges for project activities, and were chosen to reflect imminent infrastructure development, a history of forest fire, and potential land tenure disputes and political instability in the BME (UNIQUE, 2010).

Remaining emission reductions were valued at predicted over-the-counter (OTC) voluntary carbon market prices. Although social costing of carbon would value emission reductions more highly at US\$23/tCO_2e (Tol, 2008), the voluntary carbon market is currently the only trading platform from which value can be realised from avoided deforestation. The OTC voluntary carbon market is motivated by corporate social responsibility and individuals wishing to contribute to a 'solution' for climate change. In 2009, the price of emission reductions on the OTC ranged from US$0.30 to US$111/tCO_2e, with an average price of US$6.50/t$CO_2$e. In the following two years, average OTC prices have remained around US$6/t$CO_2$e, although price ranges have remained wide. Volume weighted prices for REDD+ emission reductions were US$2.9/t$CO_2$e in 2009, varying from approximately US$1 to US$13/tCO_2e, and lower than the previous three years. However, REDD+ emission reduction prices rebounded to US$5/tCO2e in 2010 and US$12 in 2011, but with large range in prices for emission reductions. For emission reductions from Africa, average 2009 prices are lower than those in 2006 and 2007, but since 2009 prices have remained around US$8/t$CO_2$e (Hamilton et al., 2007, Hamilton et al., 2008, Hamilton et al., 2009, Hamilton et al., 2010, Peters-Stanley et al., 2011, Peters-Stanley and Hamilton, 2012; see Table 6).

With no clear trend in the value of emission reductions from REDD+ or African projects, there is uncertainty in the price that can be expected for emission reductions from the BME REDD+ project. Early interest indicates that the first tranche of emission reductions could sell for US$3/t$CO_2$e (UNIQUE, 2010). When the BME REDD+ project is certified to Voluntary Carbon Standard (VCS) and Climate, Community and Biodiversity Alliance (CCBA) standards, as is planned, emissions reductions might receive a price premium. In light of this, and current OTC market prices for emission reductions, potential revenues were predicted using both US$3 and US$6/tCO_2e.

The costs of generation and sale of emission reductions are subtracted from expected revenues to give the REDD+ revenues of the BME REDD+ project. The costs of listing the BME REDD+ emission reductions in a public register, which increases transparency of the voluntary carbon market, were estimated at $0.10/t$CO_2$e. One-off costs of US$3,225,000 for REDD+ project establishment, estimated by carbon consultancy UNIQUE (UNIQUE, 2010), were assumed to be experienced in year 1. Ninety-eight percent of this cost is for establishment of 15,000 ha of woodlots to meet fuelwood demand in the BME, the remainder was for the establishment of CFM across the forest area; project design documentation development; and, validation to VCS and CCBA standards. Annual monitoring, verification, and operational costs of CFM, similarly estimated by consultants, of US$650,000 over the 14 woredas were also subtracted from sales revenues (UNIQUE, 2010). These costs estimates reflect other literature showing that REDD+ project implementation costs can be substantial (Cacho et al., 2005, Antinori and Sathaye, 2007, Nepstad et al., 2007, WCS, 2009).

Table 6. Over the counter carbon price trends and markets.
The Ecosystem Marketplace and Bloomberg New Energy Finance annually reports weighted averages of voluntary carbon market prices. Over-the-counter (OTC) market prices are presented with sample sizes and range where available (n). OTC prices are those of all locations and project types, REDD+ prices are for all locations, Africa prices are for all project types within the continent. Prices illustrate that the value of emission reductions is uncertain and without clear trends (US$/t$CO_2$e) *Source: Hamilton et al., 2007, 2008, 2009, 2010; Peters-Stanley et al 2011; Peters-Stanley and Hamilton, 2012.*

	Carbon Price US$/t$CO_2$e		
Year	**OTC**	**REDD+**	**Africa**
2006	4.1 (0.45-45) -	14 (10-18) -	18 (6-19) n=4
2007	6.1 (1.8-300) n=155	4.8 (2-30) n=11	13.7 (10-34) n=9
2008	7.34 (1.2-46.9) n=137	6.3 (5-28) n=10	5.1 (5-30) n=12
2009	6.5 (0.03-111) n=410	2.9 (1-13) n=10	8 (-) n=26
2010	6 (-) n=461	5 (1-25) -	9.1 (-) -
2011	6.2 (-) n=1798	12 (-) -	8 (-) n=12

The REDD+ revenue over the 20-year project lifespan is then calculated in 2010 US$ by applying a discount rate. The implications of discounting in the forestry sector have been reviewed by Hepburn and Koundouri (2007). They provide a rationale for time-declining discount rates in long-term forestry projects to both increase intertemporal efficiency and intergenerational equity. However, they also conclude that a constant discount rate will generally be appropriate for short-term projects of 22 years or less. As the choice of constant discount rate does, however, remain influential on the net present value, both 5 and 10% discount rates are modelled in this study following Greig-Gran (2006) of the Stern Review (Stern, 2007).

It can be seen that the REDD+ revenue is an outcome that relies on uncertain inputs in addition to the forest carbon stock estimated in section 6.2.2. Table 7 summarises these uncertainties and presents the method by which these uncertainties are addressed in this paper. Total uncertainty is communicated using an interval estimate of the possible values of REDD+ revenue that a REDD+ project in the BME could generate.

Table 7. Inputs and sources of uncertainty in profit assessment and methods by which uncertainty is dealt with in this study.

Input	Source of Uncertainty	Method to deal with uncertainty	Values used
Project risk	Uncertain impact and success of project	A non-tradable buffer of emission reductions is set aside to deal with leakage (Sohngen and Brown, 2004) and non-permanence (Sedjo and Marland, 2003) following requirements of the VCS (2007). Over time, it is possible that a portion of the buffer emissions could be sold.	Under high project risks faced in the BME, 25% of emission reductions are set aside for leakage and 40% for permanence non-delivery risk. Following the principle of conservativeness, it is assumed that none of the buffer is sold.
Carbon Price	Subjective judgement, Variability	With uncertainty in future of forestry emission reductions in carbon markets, OTC voluntary carbon market prices are predicted from best-guess under current market circumstances.	To illustrate the sensitivity to market price for emission reductions two carbon prices are modelled: US$3/tCO2e and US$6/tCO2e
Costs of Implementation	Subjective Judgement, Variability	The implementation and transaction costs of REDD+ are often high and underappreciated (Grieg-Gran, 2006, Nepstad et al., 2007, Boucher, 2008, Antinori and Sathaye, 2007, Böttcher et al., 2009). Cost estimates therefore rely on expert judgement of the implementing agencies in the BME.	Costs included in this analysis are: Registry costs of US$0.1/tCO2e; One off costs of US$11,475,000 to establish CFM; and annual costs of US$650,000, as predicted by (UNIQUE, 2010).
Discount rate	Subjective judgement, Variability	The choice of discount rate for environmental cost-benefit analysis and forestry is addressed in detail elsewhere (Weitzman, 1998, Pearce et al., 2003, Groom et al., 2005, Hepburn and Koundouri, 2007).	The sensitivity to variable discount rate is shown by modelling discount rates of both 5 and 10% following Greig-Gran (2006) in the Stern Review (Stern, 2007).

5.3. Results

5.3.1. Carbon stock and emission reductions potential

5.3.1.1. Forest carbon stock

The dbh of 2698 trees were measured, with average of 35 trees per plot, with higher tree occurrence in moist non degraded forest (59). Both moist degraded forest (25) and dry forest plots had fewer trees (20) as expected. Four plots in moist forest had no trees present when the geo-coordinates were reached, reflecting either delay between imagery and survey, or misclassification of forest glades.

Applying allometric equations to primary field data indicated the highest carbon stocks of 289tC/ha ± 108 (expressed as the 95% confidence interval of the mean) are found in moist non-degraded forest, followed by moist degraded forest at 199tC/ha ± 54 and dry degraded forest at 132tC/ha ± 73 (Figure 7). The confidence interval of the mean of the forest carbon stock estimates is large, particularly for dry degraded forest, due to the small sample size (n=18) and large variation between plots. Despite this, non-parametric comparison of carbon stock between forest types shows a significant difference between forest types at the 5% level (Kruskall-Wallis, K=6.942, df=2, p=0.0311*).

High variation in above-ground forest carbon stocks has been observed elsewhere (Henry et al., 2011). This can be due to differences in temperature, precipitation and soil fertility as well as disturbance such as; selective wood harvest, ground fires, shifting cultivation, browsing and grazing (Houghton, 2005). Signs of human disturbance were observed in a number of plots ranging from pathways and evidence of grazing. Estimated canopy cover of the plots was 50% in dry forest, rising to 58% in moist forest. Some plots in moist forest contained very high carbon stocks as a result of the presence of high dbh trees.

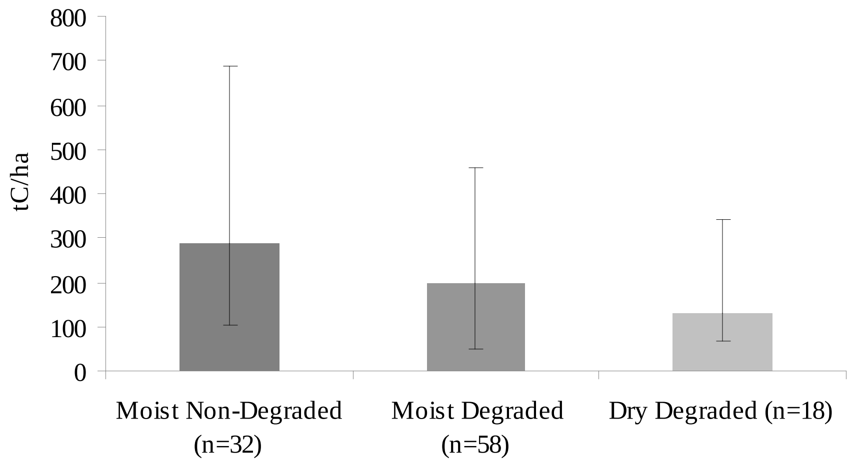

Figure 7. Average forest carbon stocks by forest type.
Shown with 95% bootstrapped confidence intervals the highest forest carbon stocks are found in moist non-degraded forest, follwed by moist degraded and dry degraded forest in the Bale Mountains Eco-Region.

The area weighted mean forest carbon stock across the forests of the BME is 195tC/ha ± 81. Although forest carbon stock distribution is found to be non-normal for all forest types (Shapiro-Wilks for moist non-degraded forest n=32, W=0.77, p<0.000; moist degraded forest n=58, W=0.76, p<0.000; dry forest n=18, W=0.68, p<0.000), a more robust bootstrapped distribution that resampled with replacement 1000 times, gave a very similar result to the normal approximation (Table 8).

The above-ground weighted mean forest carbon stock estimate from primary data is consistent with global forest ranges of 20 to 400 tC/ha reported by Hairiah *et al.*, (2001), but substantially higher than published Ethiopia-wide data of 37tC/ha and 47tC/ha (FAO, 2000, Brown, 1997). The all forest weighted mean is comparable to Africa wide estimates (Gibbs et al., 2007, Baccini et al., 2008, Lewis, 2009; see 5.2.1.) as well as forest carbon stock studies in the region. Glenday (2006) found forest carbon stocks of 330tC/ha in tropical moist forest in Kenya; although her estimates include below-ground carbon stocks. Munishi *et al.* (2010) reported tropical moist forest in the Eastern Arc Mountains to be in the range of 252 and 581 tC/ha. Although, Marshall *et al.* (2012) more recently estimated forest carbon stocks at 174.6tC/ha also in the Eastern Arc mountains of Tanzania.

Table 8. Bale Eco-Region forest carbon stock by forest type.
Comparing forest carbon stock mean and confidence intervals (tC/ha) between normal approximation and resampling with bootstrapped percentile confidence intervals, minimum and maximum carbon stock estimates are presented.

Forest Type	**Mean and 95% Confidence Intervals**				**Min**	**Max**
	Normal approximation	**Bootstrap (1000 reps)**				
		Mean	*Upper CI*	*Lower CI*		
Moist Non-Degraded (n=32)	289 ± 108	289	187	400	0	1439
Moist Degraded (n=58)	199 ± 54	199	148	258	0	1024
Dry Degraded (n=18)	132 ± 73	132	66	208	25	569
All forest (weighted mean)	**195 ± 81**	**195**	**120**	**278**	**0**	**1439**

Comparing primary data forest carbon stock estimates to biome-averaged data from the IPCC LULUCF-GPG, in both moist and dry forest the default figures and the lower bound of the primary data confidence interval coincide. Simple defaults would however, underestimate the moist forest carbon stock of the BME by between 47% and 63% and dry forest carbon stock by an average of 56% (Table 9). This largely corresponds with GOFC-GOLD findings of 44% underestimate in forest carbon stocks in African rainforest moving from Tier 1 to Tier 3 methods (GOFC GOLD 2008).

Table 9. Comparison of primary data and documented biome-averaged forest carbon stocks. Comparison of primary data with default above-ground carbon stock estimates, by forest type, show the discrepancy between simple and complex forest carbon stock accounting (tC/ha). Default data sourced from IPCC, 2006; note that the Ecological Zone data has only a point estimate.

	Primary Data	Ecological Zone Specific		Africa Specific	
Forest Classification	**tC/ha**	**tC/ha**	**As a % of primary data**	**tC/ha**	**As a % of primary data**
Tropical moist	231 (179-283)	85 (-)	- 63%	122 (75-202)	- 47%
Tropical dry	132 (58-206)	61 (-)	-54%	56 (56-61)	- 58%

5.3.1.2. Power analysis

The 95% bootstrapped confidence intervals indicate relative uncertainty of 37% in moist non-degraded forest, 28% in moist degraded forest and 53% in dry forest or 39% over all forest types. Comparing this error to Africa specific default data provided by the IPCC (2006), the interval estimate gives relative uncertainty of 46% in moist forest to 4% in dry forest (adjusting to assume symmetrical confidence intervals). Retrospective power analysis, using Equations 5 and 6, was used to determine the minimum number of forest plots required in the BME to say with 95% certainty that the results have precision of 20%: the allowable error interval is within 20% of the mean. Table 10 gives the value of variables used in the calculations for the BME as outlined in Pearson et al. (2005).

The required number of plots using the standard deviation of primary data was calculated at 108. While the total number of plots concurs with the total plot numbers required to achieve 80% power, the actual number of plots completed exceeded that required in dry forest and were less than that required in moist degraded forest (Table 11). This implies that dry forest has precision level between 20% and 30%, while moist degraded forest has precision between 10% and 15%. The estimate for moist non-degraded forest can be assumed with maximum precision of at least 20%. This concurs with the actual findings of precision estimated through calculating the standard error as a percentage of the mean; 19% in moist non-degraded forest, 14% in moist degraded, 28% in dry degraded forest. The sampling error of the primary field data is, therefore, much higher than Chave *et al*. (2004) who reports sampling error of 10% of the mean but is within the bounds of sampling errors expected for ecological studies of relatively small sample size.

Table 10. Variables used to calculate the number of plots required for statistical rigour in the Bale Mountains Eco-Region

<table>
<tr><th>Abbreviation</th><th>Description</th><th>Value</th></tr>
<tr><td>N</td><td>maximum number of sample plots in the project area</td><td>1442</td></tr>
<tr><td rowspan="3">N_i</td><td rowspan="3">maximum number of sample forest plots in stratum i</td><td>215, moist degraded</td></tr>
<tr><td>621, moist non-degraded</td></tr>
<tr><td>606, dry forest</td></tr>
<tr><td rowspan="3">s_i</td><td rowspan="3">standard deviation for stratum i</td><td>211, moist degraded</td></tr>
<tr><td>312, moist non-degraded</td></tr>
<tr><td>159, dry forest</td></tr>
<tr><td>E</td><td>allowable error or the desired half-width of the error interval, as calculated by multiplying the non-weighted mean carbon stock by the desired precision of 20%</td><td>21.4</td></tr>
<tr><td>t</td><td>sample statistic from the t-distribution for the 95% confidence level</td><td>1.96</td></tr>
<tr><td>L</td><td>Number of forest strata</td><td>3</td></tr>
</table>

As can be seen in Figure 8, to increase the precision of the forest carbon stock estimate to 10% would require data from three times as many, or 347, forest plots. It can also be seen that 108 plots is also past the point where the curve begins to level out, and there are diminishing gains to precision as sample size increases.

Table 11. Ex-post assessment of the number of forest plots required and those completed.
Logistical limitations meant that retrospective sample size analysis was conducted and although total number of plots concur with required plot numbers at 20% precision with 95% confidence, plot number exceeded that required in dry forest and were less than that required in moist degraded forest.

Forest Strata	Area (ha)	Carbon Stock (tC/ha)	Standard Deviation (s)	Plots Required (n)	Plots Completed
All	576,856	214	242	108	108
Moist Degraded	86,101	289	211	15	32
Moist non-degraded	248,350	199	312	62	58
Dry	242,405	132	159	31	18

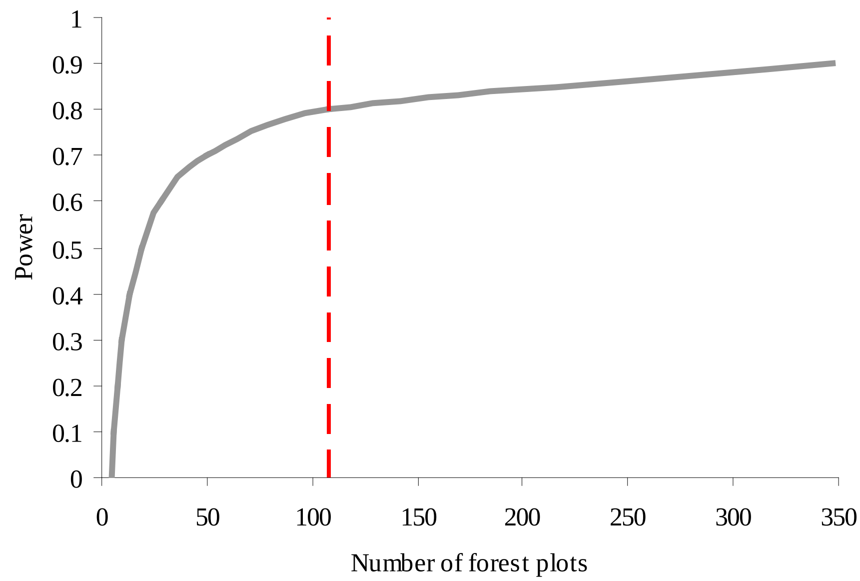

Figure 8. Power curve showing the total number of forest plots required to accept the outcome with particular level of confidence.
The figure demonstrates that 108 forest plots will achieve power of 80%, or 20% precision, and that increasing this precision to 10% would require 347 forest plots to be surveyed.

5.3.1.3. Emission reduction estimation

Estimating the cumulative emission reductions generated over a 20-year project lifespan, primary data give results more than twice as high as those generated using IPCC ecological zone default data; 180,272 ktCO_2 compared to between 71,305 and 89,723 ktCO_2 using ecological zone and Africa specific data, respectively (Table 12). These estimates support existing findings that local estimates give higher emission reduction estimates (Grassi et al., 2008, Pelletier et al., 2010, Preece et al., 2012).

Table 12. Comparison of annual and cumulative emission reduction estimates illustrating the discrepancy between simple and complex forest carbon stock accounting.
Annual emission reductions (tCO_2e) figures illustrate the increasing protection of forest and reduction of the deforestation rate.

Emission Reductions (tCO_2e)		Primary Data	Ecological Zone Specific	Africa Specific
Annual emission reductions	Years 1-5	4,006,040	1,584,661	1,993,849
	Years 6-10	8,012,080	3,169,103	3,987,698
	Years 11-20	12,018,121	4,753,654	5,981,547
Cumulative Emission Reductions		**180,271,808**	**71,304,816**	**89,723,208**

5.3.2. Revenues and profit

The difference between REDD+ profits estimated using primary data and IPCC Ecological Zone default data, the lower of the two default estimates, is substantial. Primary data suggest that after costs, a REDD+ project in the BME could bring in an estimated US$48 million as compared to US$9 million using default data with a conservative market price of US$3/t$CO_2$e and a 10% discount rate (Table 13). It should be noted that reported returns are pre-tax and no assumptions have been made about the sharing of revenues between the various forest stakeholders, as these details are yet to be decided by the REDD+ project developers.

Calculating REDD+ revenue per hectare, positive net revenues are generated under both simple and complex forest carbon stock methods. Shared over all conserved ha of forest, primary data estimates generate between US$115 and US$445/ha depending on the area under consideration, the carbon price and discount rate chosen. Default, secondary data returns are less substantial over the same area at between US$21 to US$152/ha (Table 13). In fact, the cumulative REDD+ revenues show that using secondary data with US$3/t$CO_2$e and a 10% discount rate, the project does not break even until year 6 (Figure 9).

Table 13. Net present value of profits under different forest carbon stock methods.
Calculated by subtracting the costs of REDD+ project implementation from revenues generated through sale of emission reductions. Two prices are modelled, US$3 and US$6, and two discount rates (5 and 10%) are presented and net profits given in 2010 US$.

		Primary Data				**Ecological Zone IPCC default**			
Carbon Price		*US$6*		*US$3*		*US$6*		*US$3*	
Discount Rate		*5%*	*10%*	*5%*	*10%*	*5%*	*10%*	*5%*	*10%*
Profit US$ (000)		184,978	113,607	82,671	47,591	63,359	35,129	22,893	9,017
REDD+ revenue (US$/ha)	conserved moist and dry forest	713	438	318	183	244	135	88	35
	all conserved forest	445	273	199	115	152	85	55	21
	all forest	200	123	90	52	69	38	25	10

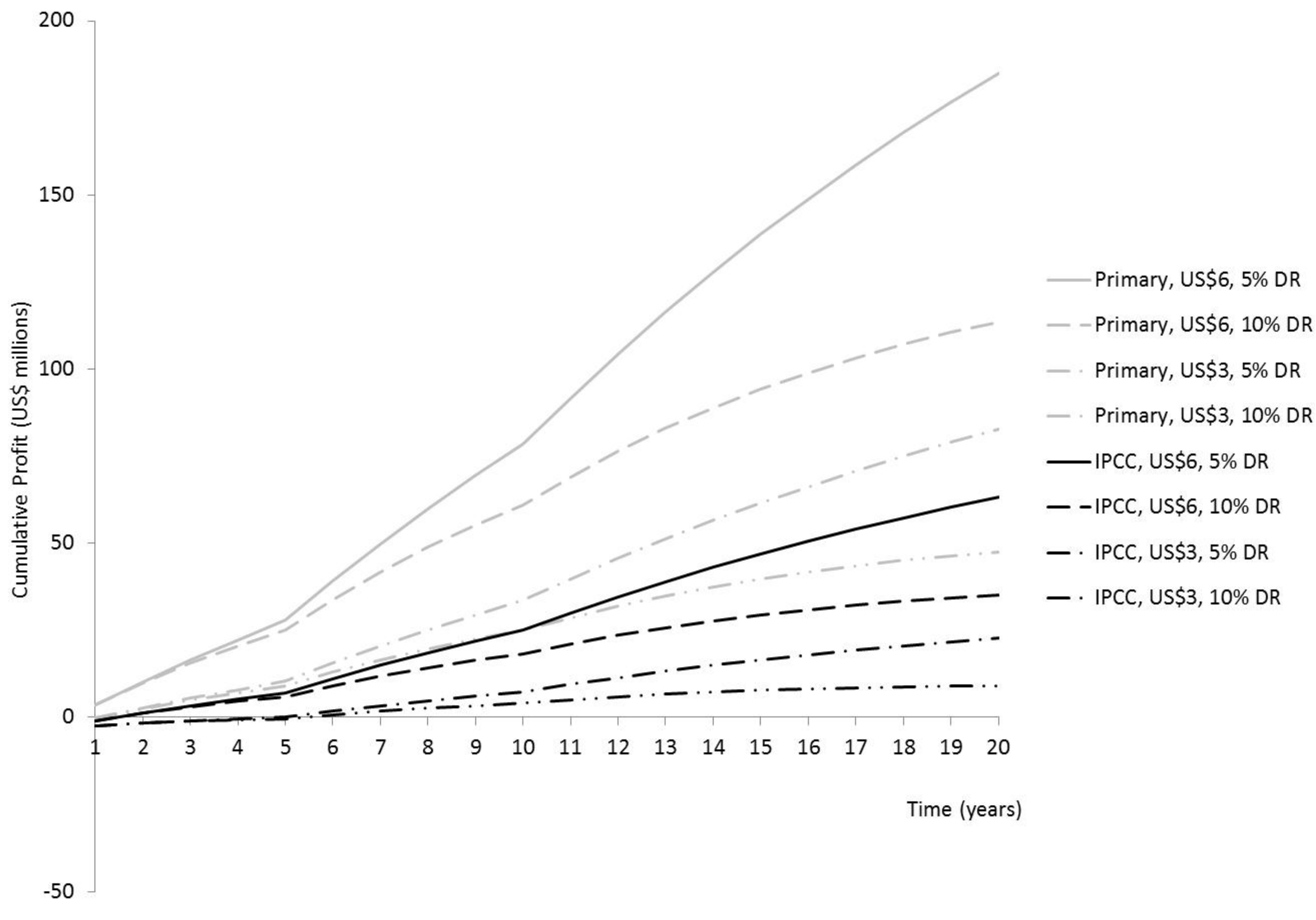

Figure 9. Estimated projected cumulative profits over the Bale Mountains Eco-Region REDD+ project lifespan showing primary and secondary IPCC data under variable carbon price and discount rates (DR).

5.4. Discussion

Comparing primary data from the BME to default forest carbon stock given by the IPCC, however, reveals a large discrepancy between Tier 1 and an estimation of forest carbon stock through tree measurements. Primary data estimated area-weighted average forest carbon stock of 195tC/ha ± 81, ranging from 298tC/ha in non-degraded tropical moist forest to 132tC/ha in tropical dry degraded forest. Secondary data, therefore, underestimated carbon density by as much as 63% in combined moist forest and 58% in dry forest. This scale of discrepancy is higher than the 44% for African rainforest reported in GOFC-GOLD (2008) but, in the same direction as four of the five comparisons that the GOFC-GOLD project made in tropical rainforest. These results suggest that diversity of forests is not sufficiently captured by the twenty ecological zones and four climate domains encompassed by the IPCC data (IPCC, 2006). Thus Tier 1 does not meet the call of the IPCC for accurate emission reductions accounting that is neither an under nor an overestimates, with uncertainties reduced where possible (Nakicenovic et al., 2000). The application of biome-averages appear to underestimate forest carbon stock at the project site.

Findings also confirm high uncertainty surrounding the use of mean estimates. The 95% confidence intervals for primary forest carbon stock estimates are, on average, 39% of the forest strata mean. The large uncertainty results in the overlap of the lower confidence interval bounds of primary data with upper bounds of the secondary data interval. The total uncertainty of forest carbon density estimates is likely to have been even higher if measurement and estimation errors were included this study. While increasing sample size can also improve the precision of forest carbon stock estimates in this case, it would take three times as many forest carbon plots to achieve precision of 10% rather than 20%. Given the substantial time and resource requirements of field data collection, the costs of increasing the statistical power of forest carbon stock estimates may, therefore, be greater than the benefits given diminishing returns to sampling effort.

The discrepancy between accounting methods leads to more than a two-fold difference in potential emission reductions from a REDD+ project in the BME. At a voluntary carbon market price of US$3 and discount rate of 10%, this difference in emission reductions is worth close to US$39 million over the project lifespan, even after project costs and project delivery risks have been accounted for. Despite the fact that emissions accounting using simple biome-averaged data can be undertaken immediately for low or no cost, which has made them an attractive option in developing countries, there are clear financial incentives for investing finance and time, and building the capacity to gather primary data. However, with costs of reducing uncertainty rising as methods become more data-intensive, trade-offs may emerge. Tools such as sensitivity analysis could be employed to identify components with the most impact on total uncertainty which can then be prioritised (Elston, 1992). With the popularity of REDD+ partially dependent on the transfer of finance from developed to developing countries, more complex accounting can also ensure that rewards for reducing deforestation and degradation are of appropriate scale.

Despite differences between accounting methods and subsequent revenues, models predicted net positive profits at the end of the 20-year BME REDD+ project lifespan. Best case returns were US$445/ha while worst case returns were US$21 per hectare of avoided deforestation in dry forest, moist forest and woodland. Even though the financial calculation does not include tax that might be taken by federal and regional government or payments to forest stakeholders which have yet to be negotiated, the returns to investment in the BME REDD+ project are positive. Given that finance for forest conservation in the BME is low and currently largely donor funded, even small net positive revenue from this forest management strategy may be a sufficient argument for implementing REDD+. This is further supported by the fact that the implementation of REDD+ through CFM also intends to shift forest resource use onto a sustainable path from the current unsustainable one. Decisions on whether to implement REDD+ may therefore not rely on completely of cost-benefit feasibility analyses. This decoupling of REDD+ policy decisions and cost-benefit analysis is evidenced by many cases where the costs of REDD+ project and policy development are being absorbed by intermediaries or met through donor finance.

While this paper considers project-based REDD+ financed through the VCM, whatever a future REDD+ mechanisms looks like, there is a need to understand carbon stocks better. Verchot *et al.* (2012) report slow progress to generate new data for GHG inventories from forests and the capacity of countries to implement higher tier inventories. Expecting all countries to be able to undertake higher Tier accounting in the near term might be infeasible, but ensuring that countries are making efforts to do so will speedy up the process. This study, therefore, makes a case for earmarking a portion of international finance flowing to prepare countries for a REDD+ mechanism for reducing uncertainty and improving national forest inventories through long-term institutional backing and resources. This can come from dedicated REDD+ initiatives such as UN-REDD and the World Bank's Forest Carbon Partnership Facility, but also through a number of other climate finance initiatives supporting REDD+ (Climate Funds Update, 2011).

This Chapter emphasises the uncertainty in emission reductions accounting for REDD+ projects. It is not intended to offer authoritative results on the carbon stocks of the Bale Mountains; further study could improve forest carbon stock estimates by increasing sample sizes and through the testing or development of allometric equations. With large discrepancy and no standardisation of methods to estimate forest carbon stocks, expected emissions reductions may not actually be realised due to the choice of method of accounting rather than changes in actual forest carbon stock. If this occurs then the environmental integrity, and so credibility, of a REDD+ mechanism will be called into question. There is, therefore, a need to improve the quantification of uncertainty, reduction of uncertainty where possible and better communication so that uncertainty forms part of policy decisions. Reducing the sectors reliance on the conservativeness principle to deal with uncertainty in emission reductions accounting will aid a more appropriate handling of uncertainty. While the conservativeness principle will remain important to ensure emission reductions are not overestimated, it should not preclude the quantification and communication of forest carbon stock uncertainties.

Dealing with decision-making under uncertainty is not novel in climate change policy (see Webster et al., 2002). Under UNFCCC negotiations, countries are encouraged but not obliged to include uncertainty estimates in their national communications to the UNFCCC (UNFCCC, 2002). While highly uncertain accounting might be acceptable for national communications, it is insufficient for a performance-based incentive mechanism like REDD+. Although additional costs will be incurred to reduce uncertainty, and trade-offs between factors in the accounting process may be introduced, the financial incentives for improved emission reductions accounting are clear.

Chapter 6: Household heterogeneity in forest income and the opportunity cost of forest conservation

6.1. Introduction

6.1.1. Problem statement

There are growing calls for community forest management (CFM) to be used a policy mechanism to deliver REDD+ (Klooster and Masera, 2000, Murdiyarso and Skutsch, 2006, Agrawal and Angelsen, 2009, Hayes and Persha, 2010). CFM establishes a common property regime where members of a well-defined group of forest users establish collective regulations for resource use, membership, monitoring, and sanctioning procedures (Arnold, 2001, Baland and Platteau, 2003, Agrawal and Angelsen, 2009). It has been shown that CFM can lead to reductions in GHG emissions where forest use becomes more sustainable (Chhatre and Agrawal, 2009, Skutsch and Ba, 2010). Thought to inherently address the livelihood needs of communities, REDD+ via CFM could reduce the risks and associated costs of dealing with the displacement of emissions outside of the project area, termed leakage (Smith and Scherr, 2003). Research is also emerging that indicates that the costs of MRV and community enforcement of forest conservation regulations can also be lower under CFM than the equivalent labour and administration requirements provided by professionals and central forest departments (Somanathan et al., 2009, Palmer Fry, 2011).

REDD+, however, emerged as a PES scheme whereby the environmental service of carbon dioxide emission reductions are sold, through a voluntary transaction, and payment is conditional upon the provision of that service (Wunder, 2005). This view of REDD+ as a PES scheme predominantly discusses a mechanism with direct, output based payments that are strongly additional and conditional on continued service provision (see Santilli et al., 2005, Parker et al., 2008, Bond et al., 2009). Such theory of PES works best where there are well-defined buyers and providers with clear property rights; thus property rights are a foremost issue in PES (Wunder, 2007, Engel and Palmer, 2008, Clements et al., 2010). With voluntary enrolment into a local-level PES scheme, theory predicts that households will not engage with a REDD+ scheme with voluntary participation at the household level if their costs of participation are not met (Wünscher et al., 2008). Although it is also known that decisions to participate in PES will also be driven by non-use values and individuals preferences for altruism, reciprocity, and conformity with the wider community (Velez et al., 2009; see also Chapter 8). It has been proposed that an estimate of the private opportunity costs (OCs) of forest conservation could be used to anchor the level of payment needed to achieve the desired level of forest conservation for REDD+ (Pirard, 2008, Wertz-Kanounnikoff, 2008, Pagiola and Bosquet, 2009, White and Minang, 2011).

Estimates of the OCs of forest conservation are predominantly based on agricultural productivity as the main driver of deforestation. These are estimated as forgone revenues per hectare, thus the OCs of land (e.g. Chomitz et al., 2005, Naidoo and Adamowicz, 2006, Börner et al., 2009). The OCs of land resulting from other drivers of deforestation such as; logging, cattle ranching or small-scale staple crop and fuelwood collection, are less frequently assessed. Fisher *et al.* (2011) is rare in considering the OCs of charcoal production on land as well as of agriculture; finding the OCs of agriculture to exceed those of charcoal per hectare of forest in the districts across Tanzania. Karky and Skutsch (2010), in considering the abatement costs of REDD+ via CFM in Nepal, note that households may incur OCs of agricultural land, but will also experience the impacts of restrictions on their use of fuelwood, fodder, timber and NTFP. Although they do not estimate per hectare values, they suggest that the OCs of agriculture may not be the appropriate measure to gauge incentives for REDD+. It is also clear that forgone agricultural benefits may be somewhat offset by the benefits that standing forest can continue to provide through NTFPs, but also a host of other use and non-use values (see also Chapter 8).

The PES literature has afforded less attention to community-level PES than PES negotiated with private individuals (Muradian et al., 2010). Community-level PES schemes provide payment at the community level, such as in Mexico where the national Programme of Payments for Biodiversity, Carbon and Agroforestry Services distributes funds to communities from the government (Corbera and Brown, 2008). Others provide in-kind payments including, clinics, schools, public transport and infrastructure (Sommerville et al., 2009). This contrasts with PES schemes that typically operate on a per hectare payment where individual land rights can be established (Engel et al., 2008).

Community level payments are often made when the attribution of costs and benefits to one household or individual is complex. In undertaking REDD+ via CFM this will be the case. Multiple members of the CFM group will overlap in their use of the forest, all members of the CFM group will have legitimate forest use rights and it is not clear who would have deforested under the status quo, either for agriculture or via other unsustainable practices. The resulting difficulty in establishing costs and benefits on any hectare, and to a particular household, complicates the establishment of a payment level. This is particularly true given that the literature on household income from forest resources shows that reliance on forests and households returns from agriculture are highly heterogeneous even within a small geographical area (Godoy and Lubowski, 1992, Byron and Arnold, 1999, Cavendish, 2000, Coomes et al., 2004, Dovie et al., 2005).

It is worth noting that many PES schemes implemented in developing countries find environmental service provision hard to attribute to individuals. As a result, payments are commonly uniform and input-based with indirect and in-kind incentives; but this is especially so where PES operates at a community level (Sommerville et al., 2009, Southgate et al., 2009, Skutsch et al., 2011).

Studies that consider the OCs of forest conservation have often deliberately excluded forests that are *de facto* common pool resources. Some note that use and deforestation on such common pool resources are often forbidden by statutory law (e.g. Grieg-Gran, 2006, Börner and Wunder, 2008). It is, therefore, suggested that the OCs of land may be an inappropriate measure for assessment of the feasibility of REDD+ policy as either illegal behaviours would be rewarded or emission reductions may not be additional (Börner and Wunder, 2008). It is for this reason that Wünscher *et al.* (2008), in their calculation of OCs of forest conservation, assume natural forest produces no commercial income. The costs incurred by the government of improving laws and law enforcement have in some cases been considered more relevant than the OCs of land for forest users in planning interventions where forest use is illegal (Börner and Wunder, 2008, Busch et al., 2009, Gregersen et al., 2010).

Under CFM, however, communities are recognised as forest management agents and legally become environmental service providers. A common property regime is established, often on previously *de facto* common pool resources. Despite the complexity, the quantitative assessment of the OCs of forest conservation under a CFM regime can, therefore, still be used to provide information on the private economic incentives that need to be overcome to generate the desired level of conservation, and so to help assess payment levels and implementation design. Appreciating the heterogeneity in OCs could, at the least, inform obligations to ensure that REDD+ projects 'do no harm' to forest communities (e.g. Griffiths, 2007, CCBA, 2008, Griffiths, 2009).

In considering the OCs of forest conservation for REDD+ on community forest lands in the Brazilian Amazon, Nepstad *et al.* (2007) propose the creation of a

Public Forest Stewardship Fund for avoided deforestation on 'social' forest reserves comprising 26% of the area's forests, including indigenous lands, extractive reserves, and sustainable development reserves. From this fund, it is proposed that direct uniform payments should be made to households of these communities anchored to half a minimum salary at US$1200 per year. A uniform payment for REDD+ via CFM that is not linked to the OCs of forest conservation for stakeholders, however, is more of a 'PES-like' scheme, in which the voluntary or conditional nature of the payment is relaxed (Landell-Mills and Porras, 2002, Wunder, 2008). Uniform payments in PES, however, are not able to account for heterogeneity in the OCs of forests conservation. The conditionality of the payment on service provision is reduced and the additionality of the emission reductions can also be called into question if payments are in excess of forgone benefits. Alternatively, payments may be insufficient to fully overcome the OCs of those most involved in damaging activities. With the potential to make households worse off if this is the case, it may threaten the longevity, and thus permanence, of REDD+ via CFM.

In Ethiopia a project to generate REDD+ through CFM is being undertaken in the Bale Mountains Eco-Region (BME; see Chapter 3 for a full description of the Bale REDD+ Project). A number of national REDD+ strategies submitted to multilateral initiatives financing REDD+ activities, such as the World Bank's Forest Carbon Partnership Facility (FCPF), are also pursuing REDD+ via CFM (see FCPF, 2011). If CFM continues to be a favoured approach for implementing REDD+, there is a need for more empirical research into the impacts of REDD+ via CFM on households' OCs and into the mechanisms by which PES-type interventions can operate in a common property regime. Empirical, ex-ante assessment of the OCs of forest conservation at the case study site can give a greater understanding of the incentives that drive deforestation. These assessments can also inform the Bale REDD+ Project design. If the magnitude and heterogeneity of the OCs of REDD+ via CFM are not sufficiently addressed in payments to communities, the longevity of the intervention, the permanence of emission reductions and thus climate change mitigation benefits are called into question.

6.1.2. Aims and objectives

This chapter uses primary cross-sectional data gathered from household surveys in southwest Ethiopia to estimate household forest income and heterogeneity in forest use. Household income from agriculture is also established. Multivariate regression is used to investigate the determinants of household income from forest use and agriculture. The OCs of forest conservation under the proposed Bale REDD+ Project are then estimated as the forgone income from a hectare of agricultural revenues and the forgone revenues from forest products at the case study site. As an ex-ante study, these OC estimates are discussed with a view to understanding how the design of a proposed Bale REDD+ Project, achieved via CFM, might capture income heterogeneity in payment incentives. The study adds to the limited literature on the OCs of forest conservation, particularly the OCs of REDD+ via CFM. I also contribute to the limited literature on community-level PES, with wider implications for REDD+ via CFM in tropical forests.

6.2. Methods

6.2.1. Household income from forests and agriculture in the Bale Mountains

Through household surveys information about a household's use of the forest, agricultural production and other economic activities were elicited. From these household surveys the income from the previous years' crop production and forest products was estimated using market price valuation. Total household income was not estimated as income from other sources such as livestock and trade were not valued (see Chapter 4). Income is defined here as the return to capital and labour that a household has access too. It includes both products consumed at home as well as those sold on markets. To reported products and yields, local-market prices were applied (see Appendix 2).

It was not possible to establish the shadow price for each household, such as through embedded time or barter values (see Section 4.5.5). Being able to attribute a shadow price to each household for each product would provide a more accurate estimate of income. The estimated income may, therefore, be an overestimate. However, market prices were applied from local markets which had no entry restrictions or middle men, in an attempt to minimise overestimation. In establishing OCs the main input costs were subtracted. The agricultural income established was net of inputs including fertiliser, seed and equipment, which were estimated as costs by the survey respondents, however, the costs of household labour were not subtracted. The main input for forest products was labour and was not subtracted. As noted in Section 4.3, the market wage was not a good measure of the shadow wage and resources were insufficient to estimate the shadow wage indirectly.

Forest income was separated into two types of forest products based on broad assumptions about their impact on the Bale REDD+ Project. Timber and fuelwood are considered high-impact forest products as they involve the removal of large amounts of biomass from the forest. Bamboo, coffee, climber and honey are considered low-impact forest products, as they perceived to remove lower amounts of biomass when harvested. These broad categorisations are in line with the Bale REDD+ Project intentions to impose forest product use restrictions on high-impact forest product extraction and none on low-impact forest product extraction. It is acknowledged, however, that further research would need to be conducted on their wider impact on biomass, such as of cultivation and harvest method, and thus on emission reductions. For example, high-impact forest products such as timber, can still be sustainable if harvest is less than growth, and the production of forest coffee may lead to losses in biomass if canopy cover is altered. Schmitt *et al.* (2009), for example, consider wild forest coffee management in the southwest of Ethiopia. They find that natural forest yields of coffee are lower than in semi-managed systems where canopy cover is and undergrowth vegetation is removed.

6.2.2. Econometric analysis

An understanding of what drives household incomes allows a more detailed exploration of how households will be impacted by the Bale REDD+ Project intervention. Household's income from agricultural production, and from forests, disaggregated into low-impact and high-impact forest income, is regressed against predicted determinants. This enables the OC estimates to be placed in the context of the proposed Bale REDD+ Project intervention.

An ordinary least squares (OLS) regression model is used to investigate the determinants of a household's income from high-impact forest products and agricultural production. Continuous income variables were log transformed to allow for a non-linear relationship between the dependent and independent variables. The log transformation also normalises the residuals and reduces potential outliers. A Logit and a Tobit regression were used to determine household income from low-impact forest products. These model specifications were necessary as 50% of the households in the sample did not have income from biomass conserving forest products, neither for home consumption nor sale, but where they did the income was a continuous random variable with positive values. The Logit model predicts the presence or absence of income from low-impact forest products with a binary, yes or no, response. Tobit, on the other hand, predicts the probability of being above zero and the determinants if the income is greater than zero (see Chapter 4 for a description of the regression models).

The explanatory variables used in the regression models were selected based on their predicted impacts on agricultural or forest production. Observable socio-economic factors expected to influence a household's forest and crop incomes are hypothesised to be; household size (HH-size), education of the household head (education), age as equated by the number of years the household head has lived in the village (village_years), and the number of livelihood sources that respondents reported to derive income from (livelihood_sources). The existence of alternative sources of income is predicted to reduce income from any single source (Godoy et al., 1997, Angelsen and Kaimowitz, 1999). However, it is not assumed that other income sources have equivalent impacts on income; some might generate proportionally higher revenues.

Village dummy variables were included to control for unobserved but constant variation across survey locations (Agarfa and Goro). These variables and their justifications are discussed in Table 15. A correlation matrix showing the degree of dependence posed by a linear relationship was established for the explanatory variables; no strong colinearity between any of the independent regressors was observed (Table 14).

Table 14. Correlation matrix of independent variables.
Correlation coefficients describing the degree of relationship between the variables used to predict household (HH) income from low-impact forest products (lifp) and high-impact forest products (hifp).

	hh_size	**education**	**village_years**	**livelihood_sources**
hh_size	1			
education	-0.14	1		
village_years	0.12	-0.35	1	
livelihood_sources	-0.02	0.07	0.00	1

Table 15. Explanatory variables for household income.

Description, hypothesised impact and justification for variables used to explore household (HH) income from forest products. + denotes a positive impact, - a negative impact, 0 a neutral impact and n/a a determinant not included in the regression analysis. High-impact forest products (hifp) include timber and fuelwood, low-impact forest products (lifp) include bamboo, climber, coffee, and honey.

Explanatory variable	Description	Hyp impact			Justification
		lifp	hifp	crop	
HH_size	*The total number of people in the household*	+	+	+	Larger households have a larger labour force and labour is a dominant input for forest product harvesting (Davies and Richards, 1999). Crop production has also been shown to increase with labour and HH food requirements (Godoy et al., 1997).
education	*The number of years of education of the HH head ranging from 0 to 13 years*	+	0	+	Education is expected to increase pro-conservation behaviours as well as improving knowledge and skills to extract forest products more sustainably and cultivate land more intensively (Godoy and Contreras, 2001, Adhikari et al., 2004).
village_years	*The number of years the HH head has lived in the village*	+	-	+	Experience through age and through knowledge gained during length of residence is expected to increase pro-conservation behaviour, thus lifp, as well as crop value through better land practices. In contrast youth is likely to represent physical strength more appropriate for hifp (Mamo et al., 2007).
livelihood_sources	*The number of sources of income the HH has including; agriculture, forest products, livestock, trade, remittance, and paid labour*	-	-	-	The existence of alternative sources of income is predicted to reduce income from any single source (Godoy et al., 1997, Angelsen and Kaimowitz, 1999).
agarfa	*Location dummy variable; 1= Afarfa, 0= not Agarfa*	Dummy variables for location were included in the model to control for village and forest type fixed effects.			
goro	*Location dummy variable; 1= Goro, 0= not Goro*				

6.2.3. The opportunity costs of forest conservation

6.2.3.1. The opportunity costs of forest products

Under the proposed Bale REDD+ Project, households will move from the open access status-quo to the proposed CFM, common property regime. A cooperative of households will be required to negotiate a Forest Management Agreement with the forest government agency that outlines resource use and deforestation restrictions as well as household responsibilities (see Chapter 3 for a more detailed discussion on the Bale REDD+ Project design and case study site).

Under Forest Management Agreement, to prevent the loss of forest cover, households will be limited in their collection of timber and fuelwood; collectively termed high-impact forest products. Households will be allowed to continue to gather forest coffee, honey, bamboo and climbers; collectively termed as low-impact forest products. The complete by-laws of forest use had not yet been negotiated with the forest government agency and the community at the time of the survey. These would dictate the restrictions on the households for high-impact forest products. It was necessary to assume, therefore, that all high-impact forest products were restricted. Predominantly as by-laws have yet to be agreed, and secondly as restrictions are difficult to relate to household harvests. For example, if fuelwood collection is restricted to deadwood for three days a week, it is not clear what proportion reduction this would have on a household.

The OCs of forest conservation were estimated as a total ban on harvesting of all high-impact forest products and with no restrictions on low-impact forest products. As households need to access fuelwood and alternative energy sources are few, the Bale REDD+ Project implementers are distributing more fuel-efficient stoves, mechanisms to generate biomass briquettes and are establishing woodlots (BERSMP, 2006). These measures will reduce or substitute household demand and thus reduce the OCs of these high-impact forest products. However, it may be some years until woodlots are mature enough to substitute the needs of households completely, thus the OCs of high-impact forest products may only be experienced by households at the outset of the intervention.

Household's reported the forest products and quantities that were collected, but were understandably unable to estimate the area of land from which they harvested these products. It is therefore difficult to establish the area of land and associated income from forest products for any given hectare of forest, as has been used in other studies of the OCs of forest conservation (Börner et al., 2009). To establish the OCs of forest products on a per hectare basis, average forest income of the survey households was first aggregated to all households to estimate the total village's income from forest products, which is then divided by the total area of forest available to the village population. This assumes that the whole of the forest area of the village is available for exploitation; that the forest income is representative of the village as a whole; and, the forest is freely accessible to all villagers. Although some dense areas of forest may be underused, the majority of forest observed in the BME is exploited. The number of households in the survey locations and the area of forest used in the estimates of OCs of forest products per hectare were sourced from local Agricultural and Rural Development Offices at the relevant villages (see Table 1, Chapter 3).

A second option to establish area of forest use, if sufficient data had been available, would be to use estimates of the biomass in a hectare of forest and then value this biomass as the diversity of products from that hectare. Fisher *et al.* (2010), for example, established the OCs of charcoal production by using statistical relationships between yield of wood available for charcoal, kiln efficiencies for turning biomass into charcoal and profit data from charcoal supply chain analyses. To have estimated the value of biomass in a hectare of forest at the case study site, however, it would be necessary to know what proportion of the biomass can be used for each purpose, so for example the above-ground tree biomass established in Chapter 5 would be a starting point of the biomass per hectare if all of it was to be used for firewood. Secondly, it would be necessary to establish the biomass content of a donkey load of fuelwood, so as to establish a market price. But this was not possible at the case study site for reasons noted in Chapter 4. Although it is reported by the Bale REDD+ Project implementers (see Chapter 3) that on average, $6m^3$/household/year is required for fuelwood consumption in the BME, it is also observed that forest use is not sustainable. Therefore, without an estimate of the biomass content of a donkey load of fuelwood, it was not possible to estimate the OCs of high-impact forest products in this way.

6.2.3.2. *The opportunity costs of land for agriculture*

Under the Forest Management Agreement, households will be unable to expand their cultivated land. Agricultural production is a rival and excludable use of land in direct opposition to REDD+ via CFM. Households that would have expanded in the status-quo will have to forego this income. No established land markets exist in Ethiopia, which under perfect market conditions could serve as a proxy for OCs, therefore, in order to establish the OCs of agricultural production per hectare, the household incomes that were established were divided by the area of cultivated land from which households reported yields (see Chapter 4, Section 4.3.2.3).

The OCs of agriculture may be reduced by the presence of low-impact forest product income on the conserved forest. Low-impact forest product income (including extraction of bamboo, climber, coffee and honey) is not in opposition to the project and households can continue to derive this income source under the Bale REDD+ Project intervention. Thus net OCs of a hectare of land can be established from the difference between agricultural and low-impact forest product revenues per hectare which may better represent the trade-offs that household may make on a hectare of land. Two agricultural OC measures were generated for each household; the OCs of agricultural land (US$/ha) and the *net* OCs of land offsetting forgone agriculture with low-impact forest production (US$/ha).

6.3. Results

6.3.1. Descriptive statistics

Of 237 household surveys, two households were dropped due to apparent misreporting of yields. Descriptive statistics for the remaining surveys support the assumption that the sample population is representative of the wider BME as they are largely consistent with other household surveys of Ethiopia (Mekonnen, 2000, Mamo et al., 2007, Babulo et al., 2009, Yemiru et al., 2010, Tesfaye et al., 2011). The average number of people in survey households is 6.5 where other Ethiopian studies find household size between 5.35 and 8.3. The average years of education of the household heads in the survey population was 4.27. The number of years the household head has lived in the village can be roughly equated to the age of the household head. An average of 42 years aligns with existing studies in Ethiopia of 35 to 50. Average land holdings of 2.2 are comparable with means reported in other household surveys from Ethiopia of between 1 and 2.1 hectares. The average number of livelihood sources was three; agriculture, forest and livestock incomes, with only a few households engaging in trade or waged labour as also found in other household income studies within Ethiopia (Table 16) (Mekonnen, 2000, Mamo et al., 2007, Babulo et al., 2009, Yemiru et al., 2010, Tesfaye et al., 2011).

Table 16. Mean household characteristics of survey respondents.
Socio-economic characteristics of the household (HH) survey population with mean, standard deviation in brackets and range.

	Description	Agarfa	Goro	Delo Mena	Total
n	Number of surveys conducted	87	50	98	235
HH head	Percentage of survey respondent who were the HH head	94%	100%	100%	98%
HH size	Number of people living within the HH	6.2 (2.27) 2-14	6.8 (2.23) 2-11	6.6 (2.49) 2-15	6.5 (2.36) 2-15
HH head Education	Years of education received by the HH head	5.22 (3.17) 0-13	3.04 (2.10) 0-9	4.05 (2.70) 0-10	4.27 (2.70) 0-13
Years HH in village	Years the HH head has lived at the survey location	44 (13) 19-83	44 (14) 22-100	39 (13) 20-74	42 (13) 19-100
Polygamous	Percentage of respondents where the male of the HH has more than one wife	16%	18%	37%	26%
Land holding	The hectares of land a HH cultivates for crop production	2.28 (1.47) 0-10	2.02 (0.65) 1-3	2.13 (1.58) 0.3-9	2.16 (1.39) 0-10
Livelihood sources	The number of income sources reported by the HH	3.1 (0.42) 1-4	3.2 (0.40) 3-4	3.1 (0.33) 3-4	3.1 (0.42) 1-4

6.3.2. Household forest income

All survey households derived income from forest products. Six major forest products were collected by households, four of which were collected by more than 30% of households: fuelwood (99.6%), timber (54%), coffee (41%), and honey (30%). Bamboo and climbers were collected less, at 7% and 1% respectively. Fuelwood is the dominant forest product with only a single household not collecting it.

The harvest of other forest products differs by location. Forest coffee is only present in Delo Mena where the moist forest type is suitable for coffee to grow. Forest honey is also most common in the moist forests of Delo Mena. Bamboo, in contrast, is only collected in Agarfa where dry forest dominates. The dry forest also contains more, and more accessible, commercially valuable species including *Podocarpus* and *Hygenia absynicca* and the proportion of households collecting timber in Agarfa exceeds that of other villages (Figure 10).

The differences between villages are also represented in the amount of forest products sold by households. Over all survey areas, an average of 40% of a household's forest product value was sold in markets. Some households sold all forest product value while others none. Two households sold all forest products in the market place, while 55 households sold no forest products on the market. The amount sold varied by forest product. Products most likely to be sold were forest honey (62%), bamboo (63%), and coffee (92%). Bamboo is only found in Agarfa, while coffee is only found in Delo Mena. Honey was consistently sold at high percentage, between 61% and 84% in survey locations. Interestingly, while 36% and 24% of fuelwood was sold on markets in Agarfa and Irba, respectively, less than 1% of fuelwood was sold in Delo Mena. The remaining major forest products were predominantly for home consumption (Figure 11).

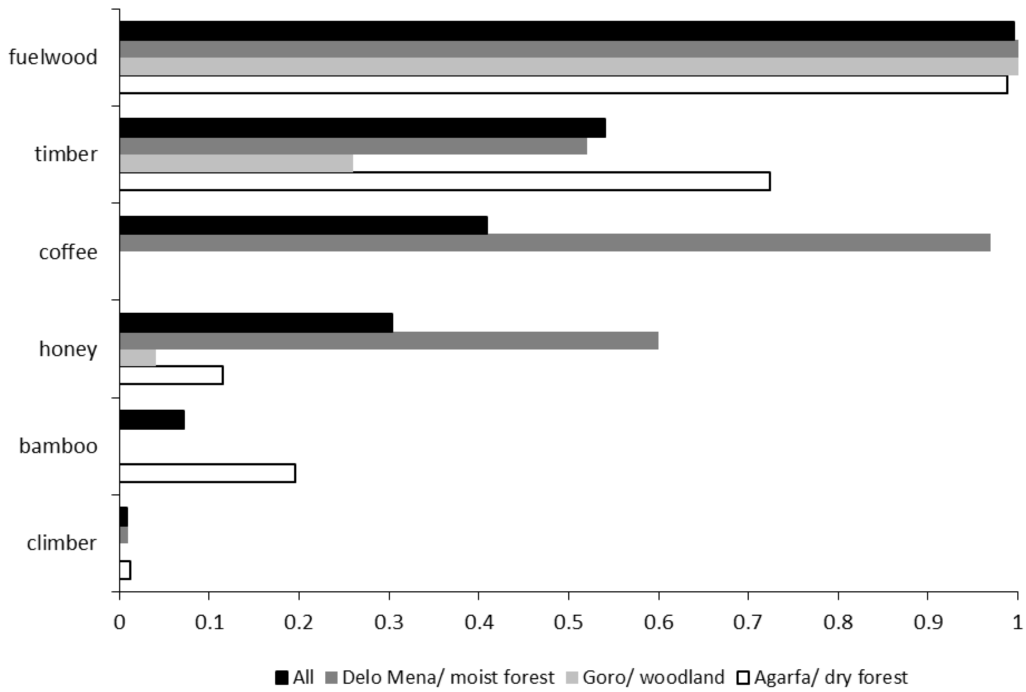

Figure 10. Forest product collection by survey location.
The proportion of households collecting major forest products by survey location with forest type.

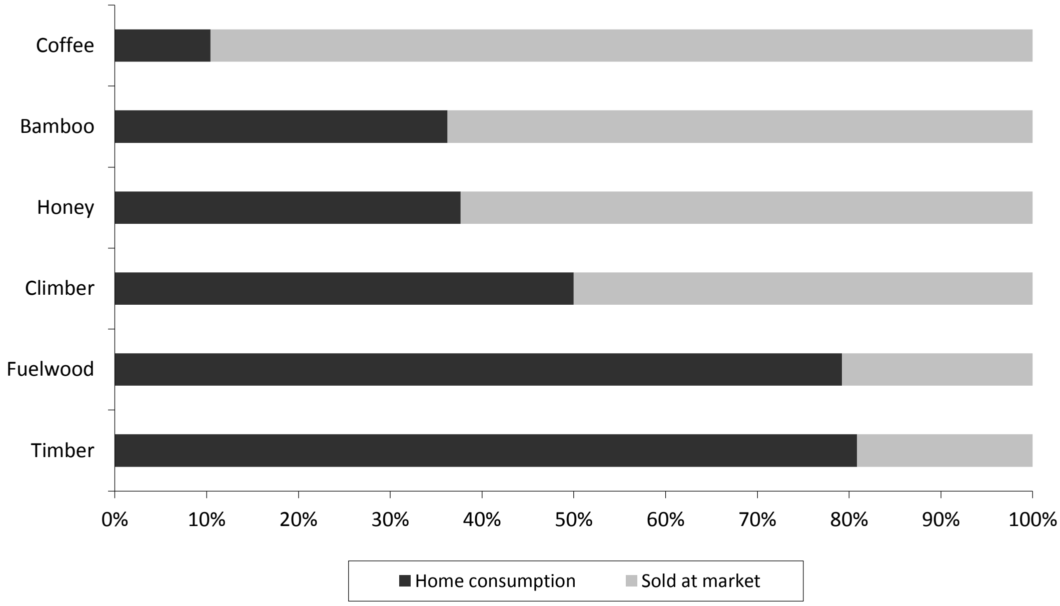

Figure 11. Forest products sold on markets.
The average percentage of a household's forest products collected for home consumption versus that sold on markets, across all survey locations.

The average income from forest products collected by households was US$1,344 in the survey year, ranging between US$51 and US$12,969 (Figure 12). Significant differences exist between survey locations in the value of total forest production, forest production for home consumption, and forest production for sale on markets (Kruskall-Wallis tests: total forest production, K=81.189, df=2, p=0.0001***; home consumption, K=5.514, df=2, p=0.0635*; forest production for sale on markets, K=94.969, df=2, p=0.0001***). Woodland households in Goro derive the lowest average forest value at US$444 and moist forest households in Delo Mena the highest at US$1,978; more than four times greater. The average household forest income in the BME is higher than the mean forest income of US$678 per household found in a meta-study of 54 cases over 17 countries (Vedeld et al., 2004), but comparisons are complicated by a host of context-specific differences in the forest resource base.

Dividing forest uses into categories of low-impact (honey, coffee, climber and bamboo) and high-impact (fuelwood and timber), households derive more income from low-impact forest products overall at US$791 ± 167 as opposed to

US$553 ± 94 from high-impact forest products. There are statistically significant differences in both forest income types across survey locations (Kruskall-Wallis tests: low-impact products, K=144.620, df=2, p=0.001***; high-impact forest products, K=50.846, df=2, p=0.0001***). This difference is driven largely by the substantial income from forest coffee in Delo Mena. In Agarfa and Goro, the income from high-impact forest products to the average household is much greater than low-impact forest products as a result of timber and fuelwood, respectively (Figure 13).

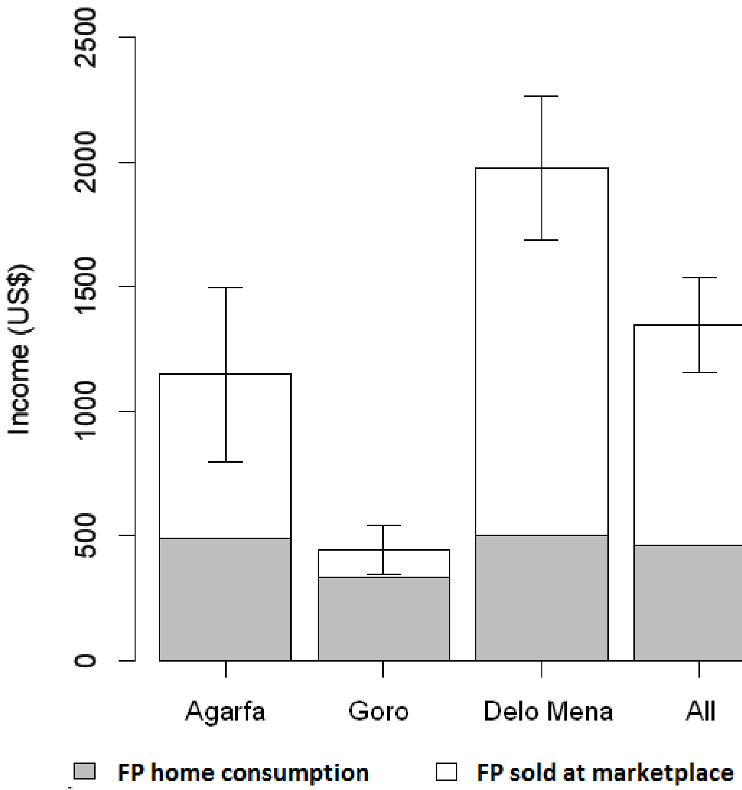

Figure 12. Mean household forest income.
Mean household income of total forest products, home consumption and that sold on markets (US$) established through market price valuation reported by survey location with total forest income 95% confidence intervals.

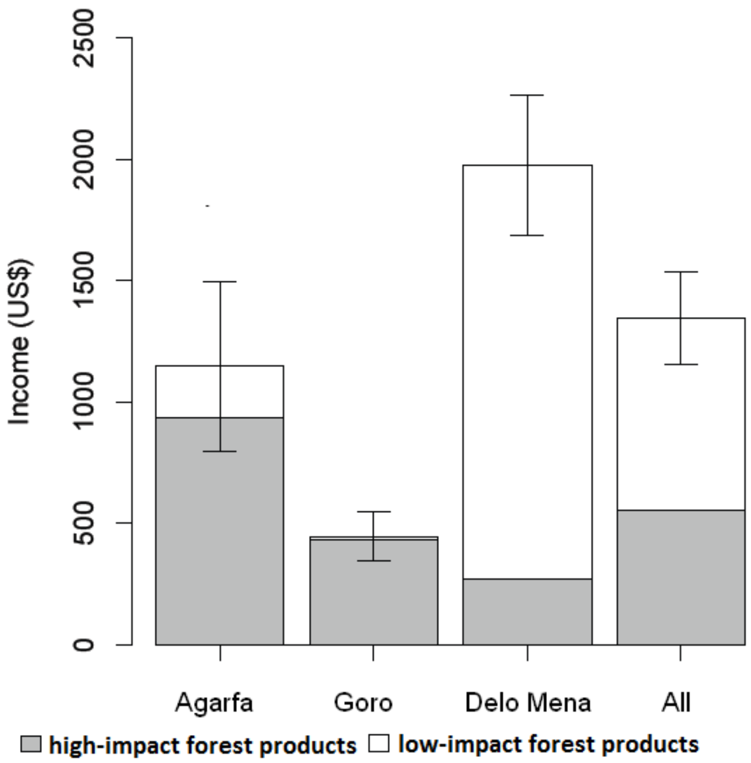

Figure 13. Mean household forest income from low-impact and high-impact forest products. Mean household income of low-impact forest products, bamboo, climber, coffee and honey (FP)(US$), and high-impact FP, timber and fuelwood (US$), established through market price valuation reported by survey location with total forest income 95% confidence intervals and range.

6.3.3. Household agricultural income

The average land holding across survey locations was found to be 2.16 ha, with a range from 0 to 10 ha. Only three of 235 surveyed households reported no gross income from agricultural land during the 12 month recall period. Of these, two did not have land holdings while the third experienced total crop failure due to drought, although others reported no such drought. Twenty crop types were identified. The top five most commonly cultivated were cereal crops: Maize, Teff, Wheat, Barley, and Sorghum.

The average household crop income was US$907 in the year of the survey ranging between US$ -157 and US$5,355 per household (Table 17). Two households had negative incomes due to higher inputs from fertilizer, seed and equipment in that year than the market valuation of their yields. Unlike income to forest use, no significant differences in agriculture were found between locations (Kruskall-Wallis tests: crop total US$/household, K=2.110, df=2, p=0.348). Over all crop types, the average household sold 20% of gross crop value with no significant differences found in the amount kept for home consumption, but significant differences found at the 1% level for the proportion sold. Goro sold 38% of agricultural yields, whereas Agarfa and Delo Mena sold less at 22% and 6% respectively (Kruskall-Wallis tests: crop home consumption US$/household, K=739, df=2, p=0.6912; crop sale US$/household, K=26.700, df=2, p=0.0001***; Figure 14).

Table 17. Mean household income from forest products and agriculture.
Mean household income (US$) for low-impact forest products (including bamboo, climber, coffee and honey) and high-impact forest products (fuelwood and timber) and agriculture, established through market price valuation reported by survey location with 95% confidence intervals and range.

Location	Income description (US$/household)		
	High-impact forest products	Low-impact forest products	Crop production
Agarfa (n=87)	937 ± 222 (102 to 5,123)	210 ± 189 (0 to 7,755)	914 ± 160 (-154 to 3,490)
Goro (n=50)	435 ± 95 (51 to 1,746)	10 ± 18 (0 to 459)	1,088 ± 266 (174 to 5,355)
Delo Mena (n=98)	272 ± 36 (88 to 985)	1,705 ±276 (0 to 7,126)	808 ± 125 (54 to 3,726)
All (n=235)	553 ± 94 (51 to 5,213)	791 ± 167 (0 to 7,755)	907 ± 98 (-154 to 5,355)

Babulo et al.'s (2009) findings for the marketed and non-marketed value of household crop production in Tigray, Northern Ethiopia, are substantially lower than these findings, with annual household income from crops ETB 414 (or approximately US$ 50). Mamo *et al.* (2006) instead found mean agricultural income of households of approximately US$675 in the Dendi district of Ethiopia. As with household forest incomes, comparisons are complicated by the context-specific differences in ecological and market variables. Mamo *et al.* also demonstrates the significant ranges in household income; with standard deviation of household incomes from agriculture ranging at just over US$ 600.

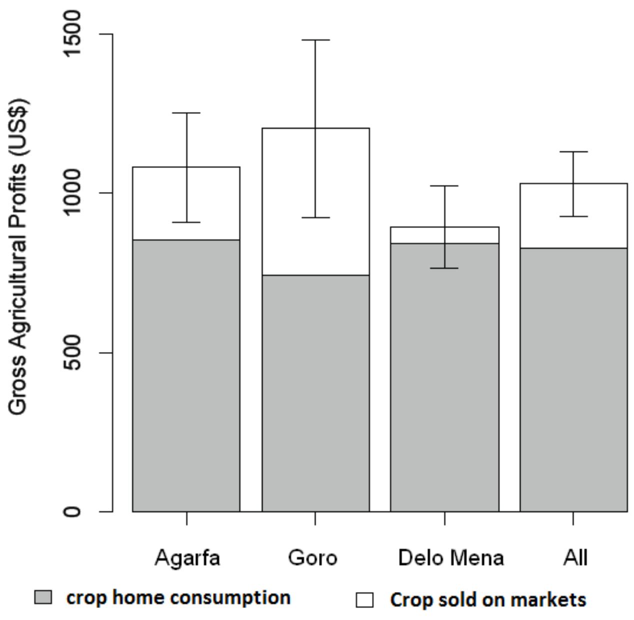

Figure 14. Proportion of gross agricultural income for sale and home consumption.
Mean household gross agricultural income for home consumption and sale (US$) established through market price valuation reported by survey location with total gross agricultural income 95% confidence intervals.

6.3.4. Relative reliance on agriculture and forest income

Overall, 40% of households had total forest incomes that exceeded income from agricultural land. This differed greatly by location, with 60% of households of the moist coffee growing forest of Delo Mena deriving more income from forest than agriculture, as compared with Agarfa (50%) and Goro (26%). Considering only forest products and dividing them into low-impact and high-impact, 57% of households overall had greater income from high-impact than low-impact forest products. Again, there is a split by location; Agarfa (94%) and Goro (100%) are both substantially higher than the moist forest of Delo Mena where only 3% of households have high-impact forest product income greater than that of low-impact forest products (Figure 15).

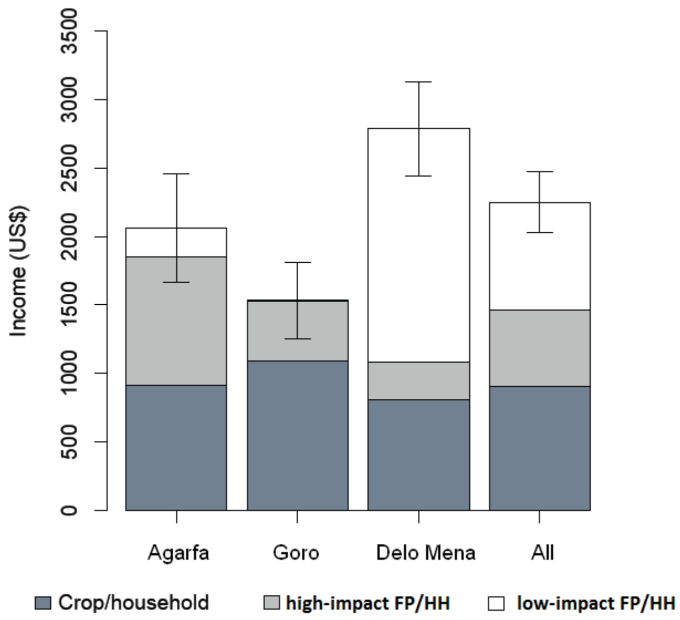

Figure 15. Proportion of household income from forest and agriculture.
Mean household agricultural income and forest income (US$) established through market price valuation reported by survey location with total forest product and crop income 95% confidence intervals.

6.3.5. Econometric analysis of household income from land uses

As in the descriptive results, the strong effect of location on forest incomes is also clear in the regression results. As noted in Chapter 4, this cross-sectional data-set is limited given that there are three survey locations with three differing forest types. Effects due to forest characteristics and village characteristics, therefore, cannot be separated. The inclusion of a village dummy variable however, encapsulates differences to help control for unobserved but constant variation across survey locations. By controlling for the location differences with dummy variables, further socio-economic factors driving income from forests and agriculture were investigated. The majority of findings correspond with hypotheses of forest and crop income reliance (see Table 15; Table 18).

Table 18. Determinants of household income per household.

OLS, Logit and Tobit regression results for predictors of household income from high-impact forest products (hifp), low-impact forest products (lifp) and agricultural production (crop) reporting coefficient, robust standard errors in parentheses and significance where; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Determinants	Description	Logit dependent variable	Tobit dependent variable	OLS dependent variable	
		lifp_yn	**lifp**	**log_hifp**	**log_crop**
HH_size	*The total number of people in the household*	0.0479 (0.0947)	60.89 (50.82)	0.0433** (0.0209)	0.105*** (0.0244)
education	*The number of years of education of the HH head ranging from 0 to 13 years*	0.135 (0.0898)	132.81*** (45.24)	0.0148 (0.0210)	0.0690*** (0.0215)
village_years	*The number of years the HH head has lived in the village*	-0.00579 (0.0194)	-1.755 (7.906)	-0.00922** (0.00458)	-0.000256 (0.00396)
livelihood_sources	*The number of sources of income the HH has including; agriculture, forest products, livestock, trade, remittance, and paid labour*	-0.514 (0.416)	-424.4** (208.7)	0.103 (0.126)	0.0766 (0.130)
agarfa	*Location dummy variable; 1= Afarfa, 0= not Agarfa*	-5.297*** (0.728)	-2937.8*** (324.6)	0.999*** (0.122)	-0.00340 (0.136)
goro	*Location dummy variable; 1= Goro, 0= not Goro*	-7.00809*** (1.0457)	-4189.9*** (652.3)	0.401*** (0.126)	0.286** (0.126)
constant		4.932** (1.784)	2142.6*** (779.8)	5.122*** (0.471)	5.251*** (0.503)
N *Pseudo R^2/ R^2*		235 0.607	235 0.0774	235 0.256	231 0.133

The R^2 value for the OLS regression for log transformed high-impact forest product income indicates that the models explain 31% of the variation. For log transformed crop income, the OLS model explains only 13% of the variation. These R^2 indicate that there are other factors not observed here which impact on income, in particular for agricultural income. A Ramsey RESET test was used to assess the functional form of the OLS models to detect omitted variables, suggesting that the models were not miss-specified (Ramsey RESET; log_hifp $F(3, 224)=0.64$, $p=0.5922$); and log_crop $F(3, 221)=1.74$, $p=0.1606$).

As in the OLS regressions, there are likely other factors that influence on income that are not included in the Logit and Tobit model. There is not an equivalent test to the Ramsey RESET for the miss-specification of the Logit model and Tobit models. A Lagrange Multiplier test, however, was used to determine whether the Logit specification was affected by omitted variable bias. Results for the Logit model showed that household size was an important control variable to include in the model, significant at the 10% level. Other controls of education, village years and livelihood sources, are not significant and could have been omitted them from the Logit specification. The Lagrange Multiplier test indicated that in the Tobit model other controls were not significant and could have been omitted from the Tobit specification. For consistency with the OLS regressions, however, even these controls which are not significant are reported in the Logit and Tobit regressions (Table 19). The most important thing to stress is that the results from the estimated coefficients for location are significant and robust across the models.

Table 19. Lagrange Multiplier test for miss-specification of the Logit and Tobit model. Significance is noted as; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Term	LM score	degrees of freedom	p value
Logit lifp_yn			
HH_size	0.02	1	0.9025
education	2.87	1	0.0901*
village_years	1.55	1	0.2130
livelihood_sources	0.77	1	0.3807
simultaneous test	8.02	5	0.1549
Tobit lifp			
hh_size	1.98	1	0.1598
years_in_village	0.01	1	0.9042
simultaneous test	3.78	3	0.2857

Household size

For both high-impact forest products and crop income, household size is a significant determinant of income. This result follows that found in the literature of increasing household size indicating increased labour availability for forest product harvesting (Davies and Richards, 1999). The coefficient can be interpreted as a one person increase in household size leading to almost a 4% increase in income from high-impact forest products. In larger households more crops may also be grown to feed more household members and labour is commonly found as a limiting factor for crop production in other household income studies (Godoy et al., 1997). A one person increase in household size leads to an 11% increase in crop income. No impact of household size was found for the presence or absence of low-impact forest product income indicating perhaps this income is not labour constrained.

Household head education and years in village

Although the average education of the household head does not exceed a primary education, education is a significant determinant of income from crops. The coefficient can be interpreted as one year of additional schooling leading to a 7% increase in crop income. This suggests that education can improve household income from a parcel of land, perhaps through knowledge of the application of fertiliser, improved seeds and farming techniques. Similar results on the impact of education have been found in other studies (Godoy and Contreras, 2001). Education was not found to impact on high-impact forest product income, or low-impact forest product income in the Logit model. However, the Tobit model shows that higher education significantly and positively determines income from low-impact forest products.

The number of years the household head has been in the village can be considered a proxy for the age of the household head. Older households derive less income from high-impact forest products, with no effects of age found on other income sources. The coefficient can be interpreted as a year less within the village increasing high-impact forest product income by 1%. This could be due to the physical intensity of labour required for high-impact forest product income sources, as Mamo *et al.* (2007) also found in Ethiopia; timber harvest requires physical strength. From household interviews and discussions at the case-study site, it was also clear that younger households resort to fuelwood and timber extraction to derive income through lack of alternatives; thus increasing their high-impact forest product income.

Alternative livelihoods and income from other sources

The number of livelihood sources a household derives income from was not found to determine income from high-impact forest products or agriculture. This could suggest that subsistence levels of high-impact forest products of fuelwood and timber as well as a certain level of crop production is necessary for household survival and, therefore, they are non-substitutable livelihood sources. While the Logit model did not reveal impact of other livelihood sources on low-impact forest product income, the Tobit model shows that if income is received from this source, the presence of other livelihood sources including trade, waged labour, remittance and livestock, reduced the income from low-impact forest products.

6.3.6. The OCs of forest conservation

There are four estimates of the OCs of forest conservation that can be provided for the case-study site. First, is the restriction on high-impact forest products (timber and fuelwood) necessitated in the project design, and not offset by the benefits received through low-impact forest products allowed by the Bale REDD+ Project design (bamboo, coffee, honey and climber). Second, is the restriction on high-impact forest products offset by ability to harvest low-impact forest products. Third, is the OCs of restriction on expansion of agriculture which is not offset by the benefits received through low-impact forest products. Finally, fourth is the OCs of agriculture, offset by low-impact forest products (Table 20).

Table 20. OCs of forest conservation per hectare assuming conversion due to high-impact forest product harvest and due to agricultural conversion, with and without low-impact forest product harvest, by location.
Average opportunity costs (OCs) per hectare (US$/ha) established through market price valuation, reported by survey location with 95% confidence intervals and range where appropriate.

	Opportunity Cost (US$/ha) of land			
	Foregone high-impact forest products only	**Forgone high impact forest products offset by low-impact forest products**	**Forgone agriculture conversion only**	**Forgone agricultural conversion offset by low-impact forest products**
Agarfa (n=80)	47	40	402 ± 71	401
Goro (n=50)	19	19	495 ± 112	495
Delo Mena (n=98)	30	-157	384 ± 60	197
All (n=228)	**28**	**-12**	**415 ± 44**	**334**

6.3.6.1. The OCs of forest products

The estimated OCs of high-impact forest products on a hectare of conserved forest was US$ 28/ha over all survey sites in the survey year. As expected by the analysis of the determinants of forest income, location played a strong role; Agarfa and Delo Mena experience the highest OCs of high-impact forest products at US$ 47 /ha and US$ 30 /ha, respectively, and with the woodlands of Goro returning US$ 19 /ha.

Assuming low-impact forest products can offset the OCs of high-impact forest products, net OCs of high-impact forest products were US$ -12 /ha across the survey site. This figure, however, obscures the fact that it is only in Delo Mena’s moist forest that negative net OCs of high-impact forest products at US$ -157 /ha are found. In both Agarfa (US$ 40 /ha) and Delo Mena (US$ 19 /ha), the net OCs of high-impact forest products are still positive (see Table 20). Thus, while on average it may appear that forest conservation appears economically viable where low-impact forest product needs can be met with the woodlots, biomass briquettes and more fuel-efficient stoves, this result is in fact location specific.

Furthermore, while 100% of households received income from high-impact forest products, only 50% of households derived income from low-impact forest products. Over all survey locations, 73% of households had below-average income from high-impact forest products and a full 93% of households in the moist forests of Delo Mena have income lower than the BME average for high-impact forest products (Figure 16). Therefore it cannot be assumed that all households will be able to capture low-impact forest product benefits.

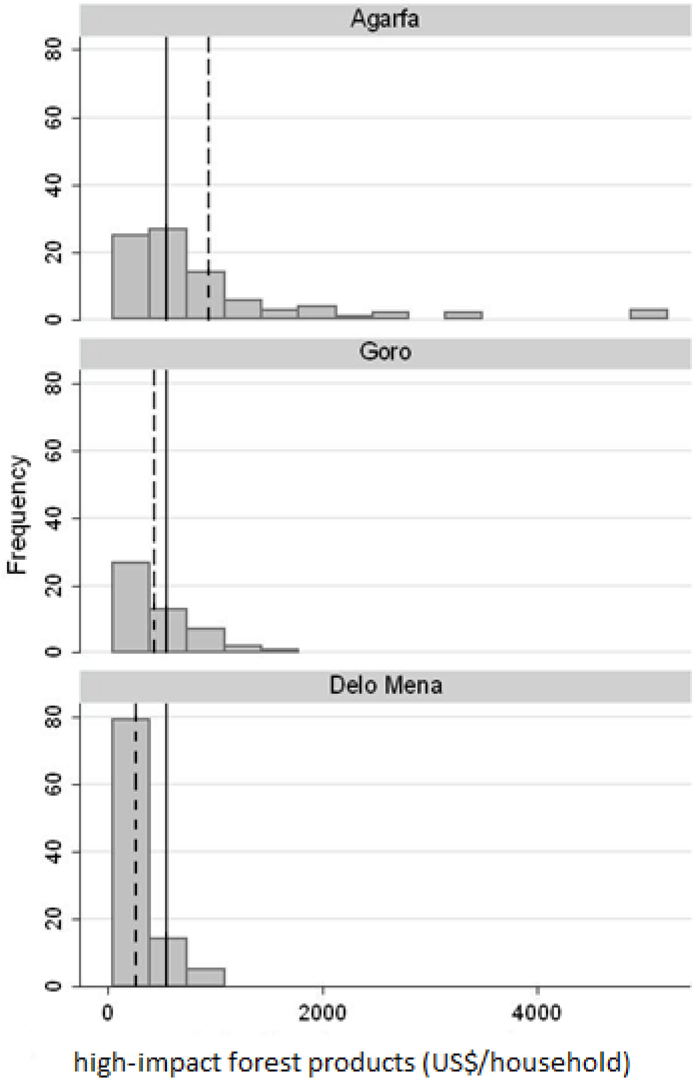

Figure 16. The distribution of household income from low-impact forest production by survey location.
Estimates of the household income from high-impact forest product harvesting (fuelwood and timber), in a single year as established using market price valuation and household survey (US$). The solid line represents the mean household income from high-impact forest products across all survey locations. The dotted line represents the survey location mean of high-impact forest product income.

6.3.6.2. *The OCs of agriculture*

A household's foregone income from a hectare of forest land conserved was first calculated as the yield per hectare of cultivated land. Three households with no land or no income from their land due to crop failure were dropped. Four additional households were dropped as agricultural income was generated from shared land holdings and thus per hectare production could not be established. Of the remaining 228 household surveys, the mean OCs of foregone agriculture were estimated at US$417 ± 43 /ha with no significant differences between survey locations (Table 20; Kruskall-Wallis tests: crop total US$/ha, K=2.791, df=2, p=0.2477).

Where low-impact forest products can offset these OCs of agricultural land, the net OCs of forgone agricultural production is estimated at US$ 375 /ha. As would be expected given the heterogeneity in income from these products, there is substantial variation in net OCs of agriculture across survey sites. A hectare of forest in Delo Mena has estimated net OCs of agriculture of US$ 197 /ha, whereas both Agarfa and Goro are higher at US$ 395 and US$ 495 respectively (Table 20).

Substantial variation in households' OCs of agriculture were found; ranging from US$ -77 to US$ +1,785. Subtracting the fixed OCs of low-impact forest products appropriate for each survey site, 29 households experienced net OCs of agriculture that were negative; again implying that forest conservation is economically attractive. However, all but one of these households was located in the moist forest of Delo Mena where coffee is a substantial income source. The remaining household was found in Agarfa, with negative OCs of agriculture resulting from the households' expenditure on crop production exceeding the market value of the yield in the survey year.

Looking at the distribution of the OCs of agriculture, 68% of households have crop income per hectare below the mean over all survey locations. Similarly, 67% of households had net OCs of agriculture below the mean over all survey locations (note that this is using a fixed OC of low-impact forest products for each survey location not for each household). By survey location, only 18% of households in the moist forests of Delo Mena have net OCs of agriculture greater than the BME mean as opposed to 40% and 48% in Agarfa and Goro respectively (Figure 17).

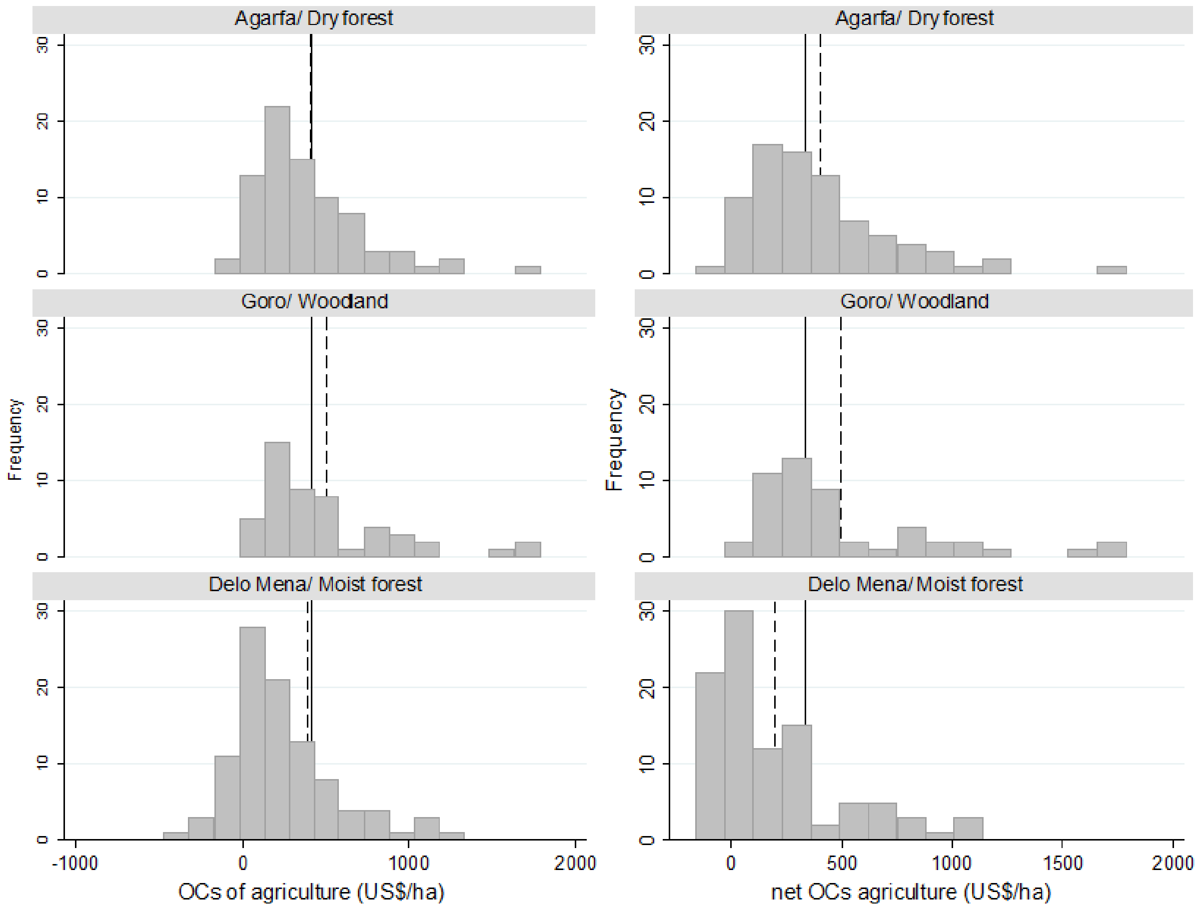

Figure 17. The distribution of household opportunity costs of agricultural production by survey location.

Estimates of the opportunity costs (OCs) of foregone agriculture (US$/ha) and foregone agriculture net of low-impact forest product income (US$/ha), in a single year as established using market price valuation. The solid line represents the mean OC across all survey locations. The dotted line represents the survey location mean OC within the survey location.

6.4. Discussion

As REDD+ has advanced on the climate change policy agenda, there have been growing calls to 'do no harm' to communities where REDD+ is implemented (CCBA, 2008). This could be understood as overcoming the local private OCs of forest conservation. Studies of the OCs of REDD+ have so far largely focused on the OCs of agricultural land. The household surveys confirmed that agriculture was a major livelihood source at the case-study site, with only two households not cultivating land in the survey year. The income that the average household derived from agriculture was US$907 ± 98. Under the proposed Bale REDD+ Project intervention, some household's will forego future expansion of their agricultural land. The average OCs of agricultural land are US$ 415/ha, but it is possible that these OCs can be reduced by income from low-impact forest products on the hectare of forest conserved. In locations where low-impact forest products can be harvested the net OCs become US$334/ha.

When considering the OCs of forest conservation, however, other drivers of forest decline should be considered, including use for energy needs (Karky and Skutsch, 2010, Fisher et al., 2011). Household surveys in the BME show that 100% of households derived income from the forest in the survey year and evidence shows this use is not sustainable and thus leading to deforestation (BERSMP, 2006). Forest use at the case study site also appears to be business-as-usual and not only a safety net in times of shocks or as a seasonal or cash flow gap filler, as hypothesised across Eastern and Southern Africa by Arnold and Townson (1998). High-impact forest products of fuelwood and timber appear products required for subsistence use with the majority of these products consumed within the home, in contrast to low-impact forest products - forest coffee, forest honey, and bamboo – which are largely exchanged on markets.

The total income from forest products for an average household, with 6.5 members, was US$1,344. The stylised analysis of the proposed Bale REDD+ Project, via CFM, further divides forest products into low-impact and high-impact categories, which broadly translate into those allowed and those not allowed at the case study site. This reveals substantial heterogeneity in forest income between villages, however, probably driven by the location specific availability of forest products. In Agarfa and Goro, high-impact forest products of timber and fuelwood contributed more to household forest incomes than low-impact forest products of bamboo, climber, coffee and honey, whereas in

Delo Mena the converse was true. This finding is largely driven by the possibility of coffee cultivation. Coffee comprises 60% of the foreign currency earnings for Ethiopia and 10% of GDP, with production predominantly based on smallholders (Teketay et al., 2010). The coffee ceremony is also part of Ethiopia's culture, thus coffee is consumed at home as well as sold. Of the household survey sites, forest coffee can only be grown in the moist forest of Delo Mena and this is reflected in forest incomes on average four times higher than in the woodlands of Goro and just less than twice as much as that in the dry forest of Agarfa. Forest honey is also dominantly available in the moist forest. Bamboo, in contrast, is only collected in Agarfa where more commercial timber species also grow; reflected in higher income from timber than in both other survey locations.

The average OCs of high-impact forest products were estimated at US$28 /ha at the case study site. Although much lower than agricultural OCs, the value does not reflect that these products are necessary for livelihoods. Households are reliant on the forest for energy needs - 99% of households gather fuelwood from the forest – and in light of rapid population growth at 2.6%, it is clear that forest conservation efforts in the BME will need to address the energy needs of households if deforestation is to be reduced and not relocated. The Bale REDD+ Project implementers do have plans for woodlot establishment, promotion of fuel efficient stoves and biomass briquettes, all of which either divert pressure or reduce biomass needs for energy. However, these will take time to implement and the OCs of being unable to collect high-impact forest products, although perhaps temporary, will be experienced by households as a result of the intervention.

It is also found that younger household heads derive more income from high-impact forest products. This suggests that this socio-demographic group may incur higher OCs of high-impact forest products than others under the proposed Bale REDD+ Project intervention. Further anecdotal evidence suggests that younger households also have lower access to cropland. Supplementary mechanisms may be required to be implemented to ensure that welfare losses are not experienced by younger household heads as a result of the intervention. This might include the preferential access for the young to woodlots, or access to livelihood diversification schemes and training. Where REDD+ is implemented through CFM, the by-laws can also accommodate socio-demographic differences; younger households might be offered greater rights to the sustainable harvest of high-impact products than other members of the group, for example. Such differential treatment due to circumstance has been seen under traditional forest management systems in the BME in the past; community members have prepared and erected bee-hives, for example, for those that have experienced sickness or death in the family (Wakijira et al., *in press*).

The OC estimates imply that households could derive more income from low-impact forest use compared with the harvest of that hectare for high-impact forest products, providing fuelwood demand can be met elsewhere. As noted above, however, this 'average' household obscures the location-specific differences in forest coffee availability. This suggests that village level differentiation in the amount paid could greatly improve the efficiency of any payments. These findings reinforce other studies in the PES literature that have found that location specific payment differentiation is more efficient than broad scale fixed payments (e.g. Wünscher et al., 2008). The existence of negative OCs might also allow for redistribution of REDD+ revenues to help overcome strongly positive OCs in other areas of the BME, or help to fund the supplementary mechanisms noted above for particular socio-demographic groups, such as the young.

At locations where low-impact forest products do generate income these activities can be encouraged under the Bale REDD+ Project intervention that gives the local communities more secure use rights. The Bale REDD+ Project implementers might achieve this through outreach and educational programmes, as education was found to increase the likelihood of household income from low-impact forest product income in the BME. Such an impact of education was also found by Adhikari *et al.* (2004). These results suggest that knowledge increases the ability of a household to extract forest products more sustainably and/or encourages pro-conservation behaviours. Such learning may also occur through institutions established through the CFM groups as evidenced by a long history of traditional forest management in the region in the 19th Century (see Chapter 3).

While according to these results, households with availability to forest coffee should have economic incentives to conserve the forest, it is also necessary to reflect on the present barriers to realising such behaviour; the lack of use rights that diminish incentive to protect valuable resources. While the additionality of emission reductions generated by these households might be challenged on financial grounds, there are clear policy barriers to generating emission reductions that could justify their additionality. This poses another interesting question for payments, as households with negative net OCs of biomass reducing forest products may not require a payment to economically incentivise forest conservation if these barriers can be removed and fuelwood and timber demands met elsewhere. This could facilitate the creation of a fund from the REDD+ revenues that will support activities across the BME, or could be used to cover running costs of the Bale REDD+ Project.

The estimate of the OCs of forest conservation relies on a number of assumptions that further research could relax. The study stylises land uses and makes assumptions concerning the restrictions that households will incur under the proposed intervention in the BME. More detailed stratification of land uses into, for example, types of agriculture and local soil conditions could also improve the OCs estimate. It is also recognised that the estimate of the OCs of land could be an overestimate as local level-market prices are applied to products consumed at home as well as those exchanged on the market place. This was necessary as I was unable to establish a household's shadow price for each product. Furthermore, no inputs were costed for forest products as they are assumed to be predominantly labour and not equipment. This is, therefore, an upper bound estimate of the OCs of forest conservation. As implementation progresses and the by-laws governing forest use are developed, it will be possible to undertake a more detailed analysis of the income that will be foregone by a household; for example, it is possible that some high-impact activities will be allowed under the intervention, but within sustainable limits such that the annual biomass increment is not exceeded by biomass removal.

In the BME, no models of land use change currently exist that can predict whether a parcel of land will be deforested for agricultural production or as a result of high-impact product harvesting such as fuelwood or timber. The development of such models would aid attribution and establish whether the true OCs of land are those of agriculture, high-impact forest products, or a combination of both (e.g. Angelsen, 1999). Such information would be useful for buyers of carbon emission reductions in establishing abatement costs, but also in establishing the true total OCs of land that must be overcome.

Another major assumption is that low-impact forest products of bamboo, climber, coffee and honey are here assumed compatible with the Bale REDD+ Project and so with forest conservation. No studies at the case study site assess the degrading impact of coffee on the forest, but it is observed that thinning of the forest canopy can occur during cultivation of forest coffee (Schmitt et al., 2009). There may also be constraints on households expanding low-impact forest product activities such as labour, upfront investments or the excludability of products which may mean that income from this source may not be feasible outside of the current production boundary. Most certainly there will be biophysical limits too. Future research into the sustainability as well as the potential for households to expand their low-impact forest product activities is necessary. This can also be something addressed over time as the by-laws of the CFM groups in the BME are flexible and will be revised as monitoring of forest extent and quality occurs.

While it can be seen that this ex-ante exploration of the OCs of forest conservation at the case study site can be used to guide design of any payments for the Bale REDD+ Project, it is acknowledged that this does not provide an estimate of the OCs that will be incurred by any one household in the BME under the Bale REDD+ Project. Instead, the OCs established here are those for any given hectare of land. This is because it is not possible to know which household would have been the beneficiaries of the next hectare of agricultural land or of the high-impact forest products harvested in the counterfactual to the Bale REDD+ Project. Such problems of attribution of deforestation to households or individuals is a recognised problem in the REDD+ literature (Börner and Wunder, 2008). It would be economically unfeasible for all households to be paid the maximum OCs of forest conservation. Even if it were economically feasible, this would impact on the effectiveness and additionality of emission reductions; there would be payments made without emissions reductions generated which a buyer would be unlikely to accept. For the same reason, the conditionality of the payment would therefore be eroded and efficiency would be reduced such that REDD+ may no longer be a cost-effective climate change mitigation measure.

One option that could be explored in REDD+ via CFM is whether the local level institutions that are established will be able to attribute the costs and benefits of forest conservation appropriately to households within their group. Local forest users should have comparative advantage over government agents for monitoring, particularly where group size is small; CBO size in the BME may well be limited to 30 households with defined forest areas (Meinzen-Dick et al., 2002). At CBO level, self-monitoring and social and cultural incentives and sanctions may also improve forest management success (Ostrom, 2000). Implementer oversight will be necessary, however, to avoid possible social risks. This might include; the capture of benefits by elites, loss of access to land, lack of voice, exacerbation of existing income and political power disparities and inequitable benefit sharing (Landell-Mills and Porras, 2002, Smith and Scherr, 2003, Griffiths, 2009, Skutsch and McCall, 2010). While CFM is proposed to deliver more equitable benefit sharing (Agrawal and Angelsen, 2009), experience holds mixed evidence on whether this is the case (Agrawal, 2001, Campbell et al., 2001, Dayton-Johnson and Bardhan, 2002).

In light of the assumptions made, it is acknowledged that this is a first estimate of OCs of REDD+ via CFM. But, with these limitations in mind, it was explored how these OC estimates can feed into design of a local-level PES scheme such as REDD+ via CFM. While previous suggestions for the implementation of REDD+ via CFM incentives have tended towards more input based, indirect or uniform payments, REDD+ via CFM could still operate as a local-level PES mechanism with improved attribution of OCs achieved through mechanisms that build on the institutions of the CFM groups. They may well be designed by the forest community based organisations themselves. Location-specific payments could be made to the CBO groups who are then responsible for establishing who bore the OCs of forest conservation and therefore differentiating payment levels between members. In this way, REDD+ via CFM could still be efficiently implemented with the conditionality and the efficiency that PES was initially proposed to deliver (Simpson and Sedjo, 1996, Ferraro and Simpson, 2002, Ferraro and Kiss, 2002).

Chapter 7: Scenarios of household opportunity costs of forest conservation over time

7.1. Introduction

7.1.1. Problem statement

Conservation interventions have historically been driven by the estimated benefits to preserving species, species populations and ecosystems. Biological hotspots, for example, are chosen for their concentrations of endemic species and severity of habitat loss (see Myers et al., 2000). Protected areas are commonly based on their representation of biodiversity and ability to ensure its persistence (Margules and Pressey, 2000). In contrast, the use of economic costs in conservation planning is still nascent (Babcock et al., 1997, Moore et al., 2004, Brooks et al., 2006, Wilson et al., 2006). This is in spite of an increasing body of evidence that shows how incorporating cost information in conservation planning leads to more effective interventions (Polasky et al., 2001, Polasky et al., 2005, Naidoo et al., 2006, Naidoo and Iwamura, 2007, Carwardine et al., 2008).

Cost information can allow the targeting of conservation interventions (Babcock et al., 1997, Adams et al., 2010). This can achieve greater biodiversity representation under fixed or limited conservation budgets (Ando et al., 1998). Even where simple correlative relationships between costs and biological variables do not exist, the incorporation of costs emphasises the trade-offs inherent in conservation planning (Balmford et al., 2000, Williams et al., 2003). The private opportunity costs (OCs) of forest conservation, in particular, provide information on both the drivers of resource use as well as the incentives that must be overcome by conservation interventions (Polasky et al., 2005). The foregone benefits of an alternative investment, activity or use of a resource, private OCs are limited to those people directly affected by foregone benefits (Pirard, 2008). Information on the private OCs of land set aside for forest conservation (referred to here as the OCs of forest conservation) aids intervention design where drivers are understood and incentives for pro-conservation activities are created. This is more likely to garner the support of local communities; necessary for intervention success (Brandon and Wells, 1992). Quantitatively assessing OCs to use in intervention design can bring greater acceptance, longevity and impact of interventions (Chomitz et al., 2005, Adams et al., 2010).

Conservation interventions largely result from concerns about the ecological and social sustainability of a resource system now and into the future. If private costs of conservation cannot be overcome through time, support for the intervention and success in meeting sustainability objectives may decline. The future OCs of land for forest conservation will be impacted by changing income from direct human activities such as agricultural and forest product extraction, and affected by infrastructure development. OCs will also be impacted by the underlying drivers of deforestation, including; demographic, economic, technological, policy and institutional, and cultural causes (Geist and Lambin, 2002).

Of the few studies that consider the OCs of land for conservation, most report OCs for a single year or assume OCs are constant over time subject only to discounting (Chomitz et al., 2005, Naidoo and Adamowicz, 2006, Börner et al., 2009). An exception is Ferraro (2002) who explores ex-ante the OCs over time imposed by the establishment of a national park in Madagascar. Without establishment of the park, the flow of benefits was first predicted to increase as locals extracted resources. As resources became degraded, however, the benefits would then decline. If the national park was established, the benefits of exploitation were assumed zero and, in the zone surrounding the national park, benefit flows would decrease more rapidly by virtue of a more limited area of access. Over a 60 year time horizon, the average present value of costs per household ranged from US$353 to US$1316. This demonstrates that where resource use is currently unsustainable and complete exhaustion of the resource base is a possibility, assumptions of constant OCs are unlikely to hold (Pearce and Markandya, 1987).

Understanding and altering economic incentives while meeting rural livelihood demands have increasingly become part of conservation interventions (Arnold, 2001). Forest conservation interventions have attempted to internalise positive environmental externalities that, through market and policy failures, have undervalued or excluded forest products and services from the income received by stakeholders (Richards, 1999). Altering economic incentives has included attempts to commercialise and increase the prices of forest products, to increase the economic value of standing forest, to diversify livelihoods to reduce pressure on forest resource systems, and to increase incomes (Brandon and Wells, 1992). Through a PES approach, stakeholders are provided with economic incentives that make conservation economically viable (Engel et al., 2008, Pagiola and Platais, 2007).

Of the market mechanisms that could be used in conservation, carbon trading is thought to have the greatest potential to capture positive externalities to the degree required to make forest conservation economically viable (Richards, 1999). This is particularly the case as carbon is often found to be the largest of the non-marketed environmental service values of forests (Pearce, 1997), which include other direct, indirect and non-use values (see Pearce and Warford, 1993). Finance through a REDD+ mechanism therefore has the potential to bring greater and more sustainable finance streams to conserve the environmental services of forest than currently exist (Landell-Mills and Porras, 2002, Pagiola et al., 2005a). The success of a REDD+ forest conservation intervention, however, will rely on REDD+ revenues being able to overcome the OCs of forest conservation over time.

The OCs of forest conservation over time will depend on the targets set by the Bale REDD+ Project implementers as well as their success in meeting these objectives. There are, however, a lack of explicit goals and quantitative operational targets in conservation (Margules and Pressey, 2000). According to Regan *et al.'s* (2002) classification of uncertainty in conservation biology, this can be considered a form of linguistic uncertainty, as opposed to more commonly researched epistemic uncertainties in determinate facts. This linguistic uncertainty arises from the underspecificity, or generality, of statements - such as conservation objectives - and/or from insufficient consideration of project goals on the part of implementers. Both increase the difficulty of understanding the dynamic nature of OCs. It is therefore hard to predict how the drivers of land-use change will evolve over time. More epistemic uncertainty is introduced where, lacking information, best-guesses and subjective judgement are used to select parameters in OCs modelling (Regan et al., 2002).

Scenarios can be used to model the dynamic nature of OCs that incorporates the lack of information on on-going drivers of change and underspecificity uncertainty in conservation objectives. Scenario modelling creates a set of plausible narratives depicting alternative pathways to the future (Bohensky et al., 2006). Stimulating thinking and allowing for the evaluation of future eventualities by describing potential future states, scenario modelling is useful to synthesise and communicate information to stakeholders and the public (Alcamo, 2001). Scenarios have recently been applied in broad-scale analyses such as the UK National Ecosystem Assessment, a nationwide exploration of how ecosystems and their services will change in the future and the associated impacts on human well-being (Haines-Young et al., 2010). Swetnam *et al.(2011)*

used participatory scenario building to consider carbon storage in the Eastern Arc Mountains of Tanzania in 2025 under an optimistic and pessimistic scenario. Presenting the findings on spatial land use maps visualised potential changes for decision makers, illustrating a 41% loss in carbon storage under business as usual compared to only a 3.8% loss under a more sustainable scenario considering change in energy, formal economy, agriculture, forestry and population.

The application of scenarios has been more limited in bottom-up conservation planning. Wollenberg *et al*. (2000), an exception, discuss how scenarios can encourage learning and adaptive co-management of community forests in Indonesia and Madagascar. Their study was not quantitative in nature, but the authors find that scenarios can help stakeholders to anticipate and adapt to large-scale forces of land-use change. If private costs of conservation cannot be overcome through time, support for the intervention and success in meeting sustainability objectives may decline. Scenario modelling of OCs can lead to more resilient conservation policies where they help stakeholders to cope with the dynamic nature of OCs and linguistic uncertainty (Peterson et al., 2003).

7.1.2. Aims and objectives

In this Chapter I use scenarios to explore potential changes in the OCs of forest conservation over time in the BME, Ethiopia. Scenarios take into account the uncertainty introduced by underspecificity of conservation objectives and paucity of data on how agricultural and forest productivity is changing. Three scenarios are generated that explore how assumptions of improvements in agricultural productivity, the proposed commercialisation of forest products, and the sustainability of land use impact upon three OC measures; the annual OCs, cumulative OCs and the total OCs of land under REDD+ via CFM in the Bale REDD+ Project intervention, over a 20-year time horizon. The potential of

REDD+ revenues from the project to overcome the OCs of forest conservation is then assessed. This study adds to a limited literature on the OCs of conservation and the application of scenarios for conservation planning.

7.2. Methods

7.2.1. The opportunity costs of forest conservation over time

The OCs of forest conservation for the proposed REDD+ via CFM intervention at the case study site were estimated in Chapter 6. Under the proposed Bale REDD+ Project intervention, communities sign a forest management agreement that prohibits high-impact forest product harvest (comprising fuelwood and timber) and prohibits the expansion of agricultural land. Households will still be able to harvest low-impact forest products (bamboo, climbers, coffee and honey). The Bale REDD+ Project implementers plan to meet household fuelwood and timber demands through woodlot establishment, fuel efficient stove distribution and biomass briquette manufacture. It is unclear, however, when fuelwood and biomass needs will be sufficiently met by these actions. This Chapter, therefore, considers the OCs of forest conservation to be forgone agricultural production, net of low-impact forest products that could be harvested on the conserved forest area.

The mean household net OCs of agricultural production in the BME were found to be US$334/ha in the survey year, with significant differences by forest type (see Chapter 6). Scaling up across the three forest types gave area weighted OCs of US$358ha. Using the weighted mean OC per hectare of forest conservation over the BME, the *annual* OCs (US$/ha) were calculated as the difference between the predicted crop (*Crop*) income and low-impact forest product (*Forest*) income, subject to a discount rate δ at a given project year t where goes from 0 to 19 (Equation 8). The *cumulative* OCs (US$/ha) for a hectare of land taken out of production in project year t are calculated as the sum of the annual

OCs for that hectare from year t to the end of the 20-year project (Equation 9). The cumulative OCs are, therefore, the discounted agricultural returns to the hectare of land over the 20-year period. The *total* OCs (US$) are those incurred over the total area conserved, estimated using the cumulative OCs per hectare and the area of avoided deforestation area A under the proposed intervention (Equation 10).

$$OC_{\text{annual, t}} = \frac{Crop_t - Forest_t}{(1+\delta)^t} \qquad \text{Eq. 8}$$

$$OC_{\text{cumulative, t}} = \sum_{t=0}^{19} \left(\frac{Crop_t - Forest_t}{(1+\delta)^t} \right) \qquad \text{Eq. 9}$$

$$OC_{\text{total}} = \sum_{t=0}^{19} A_t \bullet \left(\sum_{t=0}^{19} \left(\frac{Crop_t - Forest_t}{(1+\delta)^t} \right) \right) \qquad \text{Eq. 10}$$

All OCs are expressed in present values so that they are representative of the cost of taking land out of production now. Individuals have preferences for current over future consumption and this trade-off between different points in time can be characterised using a discount rate. As a result of the sensitivity of cost estimates to the discount rate applied and issues of intergenerational equity, the existence of a 'correct' discount rate is debated in environmental valuation (Weitzman, 1998, Pearce et al., 2003, Groom et al., 2005). In calculating the OCs of land under forests for the Stern Review (2007), Grieg-Grann (2006) applied discount rates of 5-10%. Naidoo and Adamowicz (2006) found that discount rates of 15-25% best represented observed data in calculating the OCs of land uses in Paraguay. Holden *et al.* (1998) found individuals had discount rates as high as 53% in Ethiopia and Yesuf and Bluffstone (2008), a decade later, found similarly high annual discount rates with a median of 43%. In establishing OCs in this study, a discount rate of 10% is applied to optimistic scenario A and Zero-OC scenario C, whereas a higher discount rate of 20% is applied in Pessimistic scenario B. In all cases, non-declining discount rates are used in the light of the short lifespan of the project (Hepburn and Koundouri, 2007).

The details of the assumptions made in order to estimate the OCs of forest conservation at the case study site are addressed in-depth in Chapter 4. In order to model these OCs over time, it was also assumed that all deforestation resulted from agricultural production. This assumption was necessary as no data on the conversion of land to agriculture versus that lost to high-impact forest product harvesting was available. It is also assumed that deforested area would generate the same income as existing cultivated land for a given household and once it is converted, that land will remain under agricultural production. It is acknowledged that not all land will be suitable for agriculture and land that is not currently under production is more likely to be more marginal. Similarly, it is assumed that all forest land will be suitable for low-impact forest product harvest to the level that households derive at present. The OCs may therefore be an overestimate. However, these scenarios represent a best first approximation until further information becomes available.

The land area for which OCs are incurred was based on stated project goals for deforestation to be avoided. This equates to a reduction in the existing rate of 4% deforestation annually to 3% in years 1-5, to 2% in years 6-10 and to 1% deforestation in years 11 to 20. While it is recognised that the deforestation counterfactual may increase over time in the absence of the intervention, the forest lost through conversion from forest to agriculture in the absence of the project was assumed to remain constant at 4%. This is a commonly made assumption in light of the complications of establishing deforestation baselines (see Parker et al., 2008, Böttcher et al., 2009).

The impacts of a growing rural population on the OCs of forest conservation are not considered in these scenarios. The population of Ethiopia is growing at 2.6% annually and is approaching 74 million according to the latest census (CSA, 2008). In Oromia, where the BME is located, this rate of growth is higher at 2.9%; joint third of 11 regions in the country (CSA, 2008). The effect that population growth will have on the demand for agricultural land is assumed to be met by the deforestation that occurs even under the CFM intervention; deforestation is not assumed to be completely halted. Insufficient data are available to include the potential effect of the intervention on land-market dynamics, for example on commodity prices due to changing availability of land area (see Armsworth et al., 2006, Busch et al., 2009). The price elasticity of demand is, therefore, assumed to be perfectly inelastic or zero; there are negligible price feedback mechanisms from forest policy changes.

7.2.2. Scenario calibration

Scenario analysis can include indirect socio-political, economic, science and technological, cultural and religious, and demographic drivers (Haines-Young et al., 2010). This study focuses on the direct economic drivers of resource consumption, with simple but credible changes in income from land uses modelled under the proposed Bale REDD+ Project intervention. Three scenarios were generated. Two were explorative, or probable scenarios utilising subjective judgements about the drivers of land-use change to illustrate what may happen under the Bale REDD+ Project intervention. The third scenario is normative and back-casts from a goal of zero *total* OCs of forest conservation. In all three scenarios productivity is considered for only two land uses in the BME; agriculture and low-impact forest products. All scenarios take into account the possibility of productivity decline due to degradation on these land uses as well as future plans for productivity, in order to analyse the net change over time:

1. '*Optimistic*' scenario A presents a storyline which assumes that existing production can continue into the future and is ecologically sustainable as well as sustaining income and material flows (Arnold, 2001). Scenario A is also optimistic that increases in productivity can be achieved through agricultural intensification and low-impact forest product market development and there is, therefore, growth in income and material flows.
2. '*Pessimistic'* scenario B presents assumes that resource use fails to become sustainable and continues in accordance with past trends, with efforts at agricultural intensification and forest product market development unsuccessful.
3. '*Zero OC'* scenario C seeks to achieve an economically viable forest situation. It assumes that while agricultural practices might be unsustainable and intensification of agriculture beyond the control and remit of the CFM intervention, forest management does become sustainable under CFM and efforts to add value to forest products, through price increases, are successful. This success is to such a degree that the total OCs are zero; i.e. the increasing income from forests per unit area is sufficient to completely overcome the income of agricultural production on the same land area over the 20-year period. Thus taking land out of agricultural production is an economically viable option.

Following Alcamo's (2001) criteria, the main elements of the scenarios are given in Table 21; these are the major driving forces, a description of step-wise changes, and a storyline. The base year is set at 2010, the year of the survey, and the time horizon is 20 years. Scenarios are calibrated with data from peer reviewed and grey literature, research institutions, government sources and NGOs.

Table 21. Conservation scenario storylines.
The major driving forces of land-use change, their direction and a description is provided for three plausible scenarios that might occur under implementation of the proposed participatory forest management (↑ indicates increasing, ↓ indicates decreasing, and → indicates no change).

	Scenario A: *Optimistic*		**Scenario B:** *Pessimistic*		**Scenario C:** *Zero OC*	
Driving force	***Δ***	***Storyline***	***Δ***	***Storyline***	***Δ***	***Storyline***
Agricultural productivity	→	No land degradation occurs over the 20-year period.	↓	Land degradation causes 2.2% decline of agricultural productivity per year.	↓	Land degradation causes 2.2% decline of agricultural productivity per year.
	↑	Agricultural intensification is achieved at a rate of 2.1% yield increase per year due to intervention and countrywide policy.	→	Despite intensification efforts, productivity is not increased due to the intervention or countrywide policy.	→	Despite intensification efforts, productivity is not increased due to the intervention or countrywide policy.
Forest productivity	→	No forest degradation occurs over the 20-year period.	↓	Forest degradation leads to productivity declines in low-impact forest products at a rate equivalent to area lost; 3% in years 1-5, 2% in years 6 to 10 and 1% in years 11-20.	→	No forest degradation occurs over the 20-year period.
	↑	Forest income increases due to the development of low-impact forest products at 5% per year.	→	Despite efforts, low-impact forest product development has no impact on household income.	↑?	Forest income increase s through low-impact forest product development so as to add sufficient income such that the total OCs of forest conservation are zero.
Discount Rate	-	A fixed 'low' discount rate of 10% is applied following Busch *et al.* (2009) and Grieg-Grann (2006,2008).	-	A fixed 'high' discount rate of 20% is applied following Naidoo and Adamowicz (2006).	-	A fixed 'low' discount rate of 10% is applied following Busch *et al.* (2009) and Grieg-Grann (2006,2008).

7.2.2.1. *Agricultural productivity*

Deforestation has impacts on watersheds, affecting the quantity, quality and regularity of the flow of water. While the relationship between forest cover and watershed forest cover is uncertain (see Calder et al., 2004, Bruijnzeel, 2004), it is commonly accepted that soil erosion will increase as a result of forest cover loss. The impacts of deforestation on watershed and soil quality manifest themselves as changes in agricultural productivity. An estimate of soil erosion-induced productivity decline in Ethiopia established by Shiferaw and Holden (2000) of 2.2% a year was used in scenarios B and C.

Agricultural intensification is not out of reach in Ethiopia. Byerlee *et al.* (2007) note that success will require progress in strengthening smallholder access to inputs, technology, information as well as incentivising their use and adoption. Diao and Pratt (2007) generated economy-wide simulations for Ethiopia based on agricultural and non-agricultural growth trends for 1995 to 2002. They suggest investments could raise staple yields by 3.4% annually, of which 1.3% would result from the expansion of crop area and 2.1% from growth in yields. Scenario A follows this suggestion, using a 2.1% productivity increase to simulate improvements in market infrastructure across the BME.

Scenarios B and C, however, take a more pessimistic view. This is in line with the historical performance of agricultural intensification efforts in Ethiopia where, despite ongoing public investment and the availability of technology for agricultural intensification, higher or more sustainable cereal yields, reduced food aid dependency, improved food security or lower prices for staples has not been delivered (Byerlee et al., 2007, Spielman et al., 2010). While grain production grew by 74% between 1989/90 and 2003/4, cultivated area increased by 51% (Gebreselassie, 2006). Productivity gains can largely be attributed to the expansion of agriculture rather than successful agricultural intensification (Diao and Pratt, 2007, Byerlee et al., 2007).

7.2.2.2. *Forest productivity*

While in scenarios A and C it is assumed that forest use is sustainable as a result of CFM implementation, scenario B challenges this assumption. No data are available on the impact of forest degradation on household incomes from low-impact forest products in the region and the best guess of lost profit as forests decline is equivalent to the area lost under project goals; 3% in years 1-5, 2% in years 6 to 10 and 1% in years 11-20 (UNIQUE, 2008). This is likely to be a conservative estimate; biomass in moist degraded forest was found to be 31% less than that in non-degraded moist forest which suggests that the availability of forest products is more substantially reduced in degraded forests than assumed in this simulation (Chapter 5).

Scenario B is similarly pessimistic about the development of markets for forest products. In contrast, scenarios A and C allow for value to be added to forest products. Evaluation of the indigenous plant material in the BME has identified that organisational improvement and expansion of activities could result in value addition at the local-level for wild coffee, bee products, and other indigenous plant species (Wren, 2007). As such, a number of activities have been undertaken to promote commercialisation of forest products including Business Management Training for local actors and the construction of a Collection and Processing Learning Centre with a bamboo workshop, honey processing and packaging unit and retail shop. BERSMP have been actively seeking business development partners and a *Bale Wild* brand has been established.

The change in profitability of forest products as a result of CFM activities in the BME is, however, difficult to predict. There are few empirical or quantitative case studies of the outcomes of NTFP development on household incomes (Sheil and Wunder, 2002, Gram, 2001, Neumann and Hirsch, 2000). The level of benefits and sustainability of NTFP use are also site and species specific (Wollenberg and Ingles, 1998). A similar CFM approach was followed in another project in the south-west of Ethiopia. This Non-Timber Forest Product Research and Development Project was successful in raising the price of a kilogram of honey from 3-7 Ethiopian Birr (ETB) to 16-18 ETB, a 340% increase, through organic certification (BERSMP, 2008). The Bale Forest Enterprise has agreed to pay 25% above the local market price for quality coffee beans in the BME and specialised coffee buyers have expressed interest in paying up to US$10 per kg of forest coffee; considerably more than the US$2.20 per kg in local markets (BERSMP, 2009).

Despite the uncertainty introduced by the qualitative project goals, which are simply to add value to forest products in the BME, scenario A represents a best-guess at productivity increases. The forest products included in the OCs calculation are bamboo, climber, coffee and honey. As coffee and honey are not ubiquitous across the forests of the BME and it is not clear that value added will be obtained by all households, a conservative average estimate of a 5% price increase is applied in scenario A across these forest products. Scenario C instead assesses the level of value addition required to be added to forest products in order to result in zero total OCs of forest conservation. Value addition raises the price for products rather than yields and therefore would not increase pressure on the forest resource base. Both scenarios assess net productivity gains, thus these are gains after any additional costs of harvesting, processing, trading, transporting and marketing products have been deducted.

7.2.3. Income from REDD+

Realising the value of previously non-marketed environmental services from the forest can go some way to overcoming the OCs of forest conservation. Carbon is considered the largest of the non-market values (Pearce, 1997) and is included in this Chapter while other non-market values are not. These include environmental values such as watershed protection, biodiversity protection and landscape beauty (Pearce and Warford, 1993, Davies and Richards, 1999). The omission of these values are common in studies of the OCs of conservation (e.g. Naidoo and Adamowicz, 2006, Grieg-Gran, 2008), although it is acknowledged that inclusion of other non-market values is likely to decrease the net OCs of forest conservation (Nepstad et al., 2007, Bellassen and Gitz, 2008).

While CFM and REDD+ can both be undertaken as separate policy interventions, in the BME these are therefore considered together: the Bale REDD+ Project undertakes REDD+ via CFM. Thus emission reductions do not have to be additional to that achieved through CFM, but rather are those generated by CFM (see also Chapter 3 for a full description of the Bale REDD+ Project). Covering a total of 923,593 ha, the REDD+ project area consists of the dry and moist tropical forest as well as the southern woodlands. Documentation for the proposed Bale REDD+ Project states that emission reductions will be generated only from change in the tropical dry (Agarfa) and moist forest (Delo Mena) regions. Changes in the area of woodland (Goro) will be set-aside to account for emissions that may be relocated rather than reduced; termed 'leakage' (see Sohngen and Brown, 2004). Therefore, only emission reductions in the 576,856 ha of dry and moist forest are assumed to generate carbon revenues. In Chapter 5, it was determined that carbon stocks in these dry and moist forests of the BME had an area-weighted average of 195tC/ha. The proposed Bale REDD+ Project in the BME was found to be able to generate 180,271,808 tonnes of CO_2 over the 20-year project period. In assessing whether carbon revenues from REDD+ are able to overcome the OCs of forest conservation incurred by forest users, carbon income is calculated per hectare of deforestation but shared over the total are of avoided deforestation. The OCs of forest conservation will still need to be overcome on the woodland even though it will not generate saleable emission reductions (see Table 22).

Table 22. Deforestation rate and area of avoided deforestation according to documented conservation project goals.

	Deforestation rate	Area of forest generating emission reductions (ha/yr)	
		dry and moist forest	dry forest, moist forest and woodland
Years 1-5	3%	5,769	9,236
Years 6-10	3%	11,537	18,472
Years 11-20	1%	17,306	27,708

The finance available to compensate OCs will depend on the price of a tonne of emission reductions and the costs of getting the emission reductions to market. A financial analysis in Chapter 5 valued emission reductions on the voluntary carbon market at between US$3 and US$6/tCO_2. The lower bound represented the interest shown by early buyers, the upper bound more optimistic about achieving a price premium. This premium could be achieved once the project is certified to voluntary standards that require third-party verification of project methods and due diligence for carbon as well as environmental and social project goals (e.g. VCS, 2007, CCBA, 2008). The voluntary carbon market price for REDD+ is taken here at US$3/t$CO_2$ and US$6/t$CO_2$e.

Given no clear price trend for emission reductions on the voluntary market for REDD+, for emission reductions from Africa or overall, these carbon prices are assumed to be fixed over the project period (Hamilton et al., 2007, Hamilton et al., 2008, Hamilton et al., 2009, Hamilton et al., 2010, Peters-Stanley et al., 2011). The income from emission reductions was established by subtracting upfront and annual project costs from REDD+ revenue. These annual costs included monitoring, verification, registration, brokerage of emission reductions, and

CFM operating costs. Possible leakage and non-permanence risks were accounted for through a 65% buffer of emission reductions pre-sale (UNIQUE, 2008). Thus far, however, the incidence of these costs of the Bale REDD+ Project is not clear. BERSMP has, for example, so far absorbed upfront project development costs. Furthermore, with national REDD+ plans underway, risks of leakage and non-permanence could be addressed at a national level. Withholding a risk buffer of emission reductions may not therefore be necessary (R-PP, 2011).

REDD+ profit is calculated net of the costs of implementation that are not likely to be overcome through national mechanisms or via NGOs involved in the project. These costs are outlined in Chapter 5 and include, here, project implementation costs of CFM operation, brokerage, and monitoring, reporting and verification of emission reductions. The resultant REDD+ revenue per hectare, is compared with the cumulative OCs of a hectare of land. It is assumed that carbon revenues will be received in the year in which the forest is conserved, thus annually over the 20-year period.

7.3. Results

7.3.1. Estimates of the opportunity cost of forest conservation over time

In Optimistic scenario A, the income from both agricultural land and forest area increases without land degradation. With incomes to cropland already greatly exceeding income from low-impact forest products per hectare, the growth in agricultural productivity offsets the added value to forest products, so that the present value of annual OCs in scenario A remain positive at US$74/ha at the end of the project period. Under this storyline, there remains an economic incentive to convert forest to agricultural land throughout the lifespan of the project.

In Pessimistic scenario B and Zero-OC scenario C, crop income declines due to assumptions of unsustainable farming practices lowering yields combined with failure to increase agricultural productivity. Under scenario B, the income from forest products also declines as a result of overharvesting and the inability to add value to forest products. Although the annual OCs in scenario B remain positive over the 20-year lifespan, they decline to US$7/ha in project year 19 suggesting that incentives to deforest, although, positive, would be weaker than under Scenario A.

Unlike scenario B, scenario C sees forest productivity per hectare rising over lifespan of the intervention. The annual OC becomes negative at project year 13. In this year, forest income is projected to be greater than agricultural income per hectare due to the continued degradation of agricultural land reducing yields, no agricultural intensification and successful development of markets for forest products. At project year 19 the income from a hectare of forest is US$99 more than the income for the equivalent area of agricultural land (Figure 18).

Looking at the cumulative OCs and total OCs of forest conservation in the three scenarios clearly illustrates how small changes in the storyline substantially change the outcome. The cumulative OCs per hectare, for a forest area taken out of production in the first year of the intervention, are US$3,658/ha in scenario A and US$1,889 in scenario B illustrating the large differences in the incentives faced when agriculture is foregone for a 20-year period. This is not just due to the difference in discount rate that is present between scenario A and B. Comparing scenarios A and C with the same discount rate, the cumulative OCs of land taken out of agricultural production in year one is the same. However, in scenario C forest profits increase, despite agricultural land degradation, to such an extent that cumulative OCs become negative at project year 8. At this point, the discounted income from that hectare of land for the remainder of the project period is such that forest conservation is the economically rational choice.

The overall discrepancy in the scenarios can also be seen in the total OCs. Summing the cumulative OCs over the area of forest conserved, the total OCs in the probable scenarios A and B are substantial. The total OCs exceed US$414 million in scenario A which is almost three times the total OCs in scenario B of U$115 million. In normative scenario C, increases in the price of forest products by 12.6% annually were able to generate zero total OCs over the entirety of the forest area conserved. In this economically viable forest scenario, the sum of the positive total OCs incurred over the full Bale REDD+ Project intervention lifespan, amounting to US$55,746,098, are offset completely by the sum of the negative total OCs (Table 23). Thus in scenario C the storyline as it stands is sufficient to overcome the OCs of forest conservation without REDD+ project development. However, it is clear that while forest conservation might become economically rational, this result relies on a 20-year time horizon being considered by local forest stakeholders.

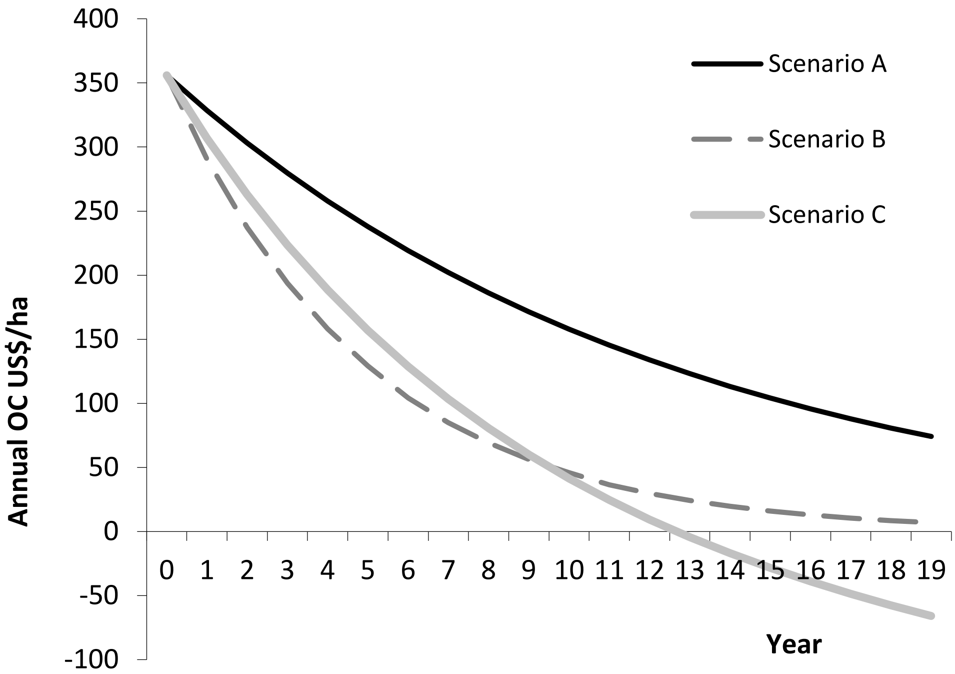

Figure 18. Annual opportunity costs over time under three future scenarios.
The annual opportunity costs (OCs)(US$/ha) are shown over the lifetime of the Bale REDD+ Project intervention for three future scenarios. Scenarios represent differing assumptions of sustainability of the resource base, efforts to intensify agricultural production and adding value to forest products.

Table 23. Annual, cumulative and total opportunity costs of forest conservation under three scenarios.
Differing assumptions of sustainability and of the productivity gains from agricultural and forest land-uses translate to large differences in opportunity cost (OC) estimates of a the Bale REDD+ Project intervention over a 20-year time horizon. All OCs estimates are reported as present values (scenarios A and C with discount rate of 10%, and scenario B with discount rate of 20%). Annual OCs (US$/ha) refer to the difference between the discounted incomes from the alternative land uses, per hectare, in a given project year. Cumulative OCs (US$/ha) refer to the discounted sum of the difference in the income of the two land uses for a 20-year period. The total OCs (US$) refers to the cumulative OCs incurred over the total area of avoided deforestation.

Project year	Forest saved (ha)	*'Pessimistic'* Scenario A			*'Optimistic'* Scenario B			*'Zero OC'* Scenario C		
		Annual OC (US$/ha)	Cumulative OC (US$/ha)	Total OC (US$)	Annual OC (US$/ha)	Cumulative OC (US$/ha)	Total OC (US$)	Annual OC (US$/ha)	Cumulative OC (US$/ha)	Total OC (US$)
0	9,236	356	3658	33,783,665	356	1889	17,446,610	356	1681	15,527,830
1	9,236	329	3302	64,279,339	291	1533	31,605,229	307	1325	27,767,668
2	9,236	303	2973	91,740,690	237	1242	43,079,703	263	1018	37,173,508
3	9,236	280	2670	116,402,374	194	1005	52,363,013	224	755	44,151,080
4	9,236	258	2391	138,481,431	158	811	59,857,630	188	532	49,063,257
5	9,236	238	2133	158,178,588	129	653	65,892,119	157	343	52,234,857
6	18,472	219	1895	193,180,340	104	524	75,577,305	129	186	55,678,970
7	18,472	202	1676	224,132,782	85	420	83,340,188	103	58	56,746,098
8	18,472	186	1474	251,353,251	69	335	89,537,053	80	-46	55,905,131
9	18,472	171	1287	275,135,089	56	266	94,458,150	60	-126	53,577,763
10	18,472	158	1116	295,749,419	46	210	98,339,934	41	-186	50,143,835
11	27,708	145	958	322,295,481	36	164	102,892,588	25	-227	43,847,226
12	27,708	134	813	344,813,375	30	128	106,436,014	9	-252	36,869,730
13	27,708	123	679	363,623,926	24	98	109,159,324	-4	-261	29,632,530
14	27,708	113	556	379,023,484	20	74	111,216,221	-17	-257	22,518,191
15	27,708	104	443	391,285,748	16	55	112,731,630	-28	-240	15,875,130
16	27,708	96	338	400,663,453	13	39	113,807,074	-39	-211	10,021,601
17	27,708	88	243	407,389,939	10	26	114,525,062	-49	-172	5,249,236
18	27,708	81	155	411,680,597	9	15	114,952,641	-58	-124	1,826,198
19	27,708	74	74	**413,734,213**	7	7	**115,144,294**	-66	-66	**0**

7.3.2. Carbon profit as a forest product

REDD+ project income per hectare is compared with the cumulative OCs of forest conservation modelled under the three scenarios. Applying the lower bound market price of emission reductions of US$3/t$CO_2$e, net of the costs of REDD+ implementation, REDD+ revenues added between US$1,192/ha to US$39/ha to forest income depending on the year that the forest was conserved. In contrast, applying a market price of US$6/t$CO_2$e that could be realised if third-party project standards are met, REDD+ revenues added between US$2,499/ha to US$80/ha (Table 24).

In scenario A, at conservative carbon prices, REDD+ revenues were insufficient to overcome the cumulative OCs of forest conservation until the final three years of the project, with the discrepancy ranging from US$2,466 in year one to US$69/ha in year 17 of the project. In scenario B, REDD+ revenues are initially unable to overcome the cumulative OCs, but at project year 11 the difference is only US$2/ha and by year 19 REDD+ revenues are able to overcome the cumulative OCs by US$32/ha. Scenario C shows the same pattern as scenario B but greatly advanced. REDD+ revenues are able to overcome the cumulative OCs at project year 3 and at the end of the 20-year period exceed cumulative OCs by US$269 (Figure 19).

Even with a more optimistic carbon market price of US$6/t$CO_2$e, in scenario A REDD+ revenues were insufficient to overcome the cumulative OCs of forest conservation until project year 13, with a discrepancy of US$1,159/ha in the first year and US$2/ha at year 12. In contrast, at this higher carbon price, REDD+ revenues are sufficient at all time periods to overcome the OCs of forest conservation in scenario B (Figure 19). The revenues in excess of the OCs of forest conservation amount to US$40,485,602 in scenario A, and US$79,276,050 in scenario B.

Table 24. REDD+ revenue for a hectare of avoided deforestation (US$/ha).
The annual income per hectare over the lifetime of the Bale REDD+ Project intervention from carbon revenue established through two voluntary carbon market prices, minus costs, over the total area of avoided deforestation applying a 10% and 20% discount rate (DR).

Project year	Emission reduction price US$3/t$CO_2$e		Emission reduction price US$6/t$CO_2$e	
	10% DR	20% DR	10% DR	20% DR
0	1,192	1,192	2,499	2,499
1	1,084	994	2,271	2,082
2	985	828	2,065	1,735
3	896	690	1,877	1,446
4	814	575	1,707	1,205
5	762	493	1,573	1,018
6	693	411	1,430	849
7	630	343	1,300	707
8	573	285	1,182	589
9	521	238	1,075	491
10	478	200	981	411
11	434	167	892	343
12	395	139	811	285
13	359	116	737	238
14	326	97	670	198
15	297	80	609	165
16	270	67	554	138
17	245	56	504	115
18	223	47	458	96
19	203	39	416	80

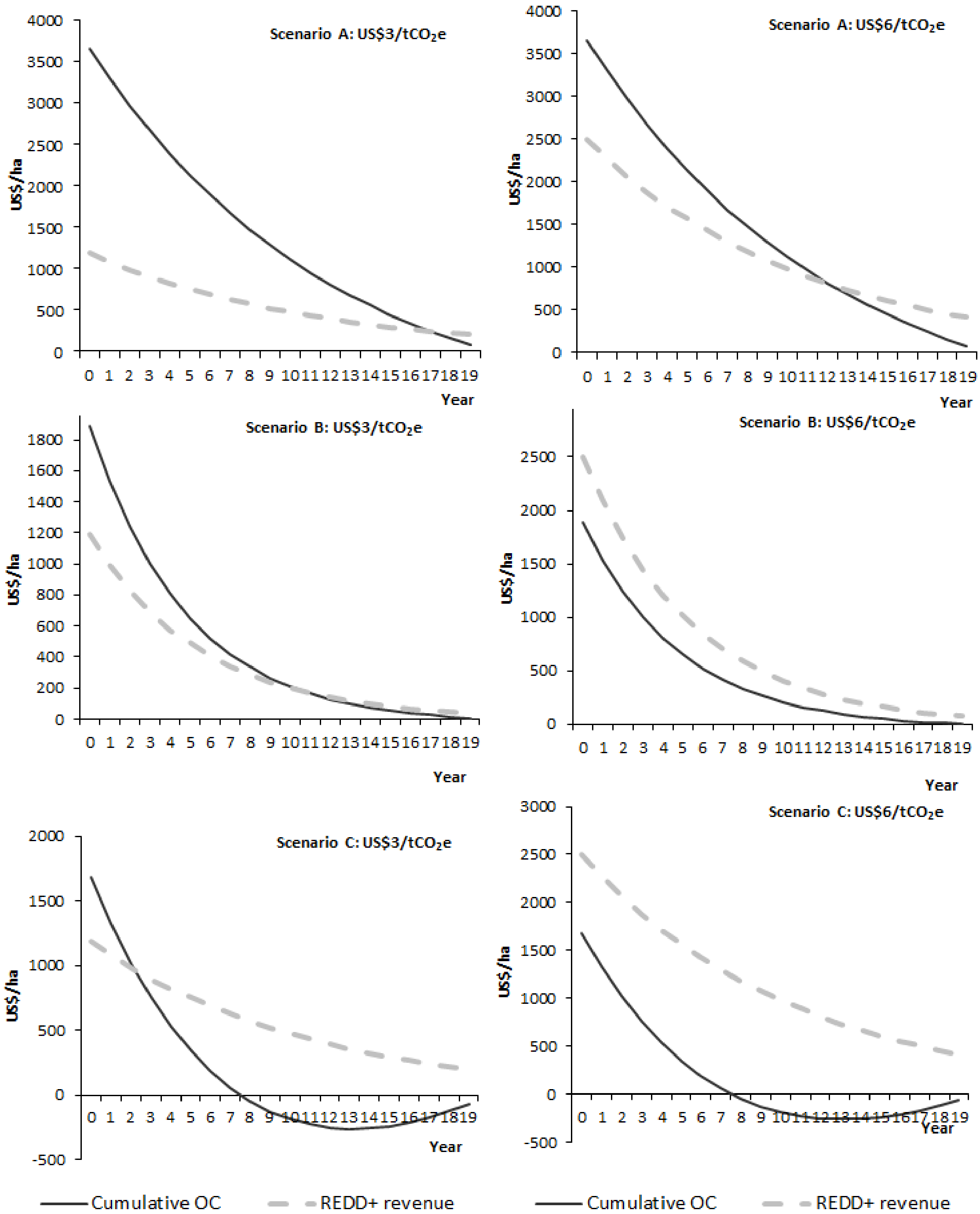

Figure 19. Cumulative opportunity costs and REDD+ revenues over time.

The cumulative opportunity costs (OCs) of forest conservation (US$/ha) for a hectare of land taken out of production in a given project year are presented as the bold line for the three scenarios representing differing assumptions of sustainability of the resource base, efforts to intensify agricultural production, and adding value to forest products. The potential REDD+ revenues receive for the hectare conserved (US$/ha) is presented by the dashed line, for the three future scenarios at two voluntary carbon market prices; US$3/$tCO_2$e and US$6/tCO_2e.

7.4. Discussion

The Bale REDD+ Project intervention objectives in the BME are not quantified within project documents, which state only qualitative strategies to achieve these reductions in forest losses (BERSMP, 2006). Scenario modelling of the OCs of forest conservation illustrates that this linguistic uncertainty, combined with a lack of information on how the productivity of land use will change over time, leads to substantial uncertainty in annual, cumulative and total OCs of forest conservation. This difference is most clearly seen in the optimistic scenario A and pessimistic scenario B. The cumulative OCs per hectare of forest conserved in the first year of the intervention – but for the 20-year period – differs by US$1,769/ha between the scenarios, with total OCs in scenario B 28% of those in scenario A. These differences are a result of simple changes in the assumptions about intensification of agricultural production, the degradation that results from land use practices, the ability to increase the value of forest products and the discount rate applied.

The highly positive annual OCs in Optimistic scenario A can be attributed the gains in agricultural productivity that outpace gains in forest product incomes, with neither land uses leading to degradation of the resource base. Agricultural intensification in Ethiopia, however, has persistently failed and this failure is attributed to a narrow focus on technology, with factors such as access to agricultural credit, incentive structure, institutions, governance and risk behaviours side-lined (Gebreselassie, 2006). Tenure insecurity, weak extension services and limited use of fertilisers, improved seeds and pesticides only exacerbate this. While the CFM project will give a form of tenure that has the potential to affect technology choice and utilisation within agriculture and influence access to long-term credit (Deininger et al., 2003), it may be that the implementing agency has limited influence over agricultural intensification in the project area.

The development of forest products, in contrast, may be more within the control of the conservation implementer. Forest product development has been set in motion by the BERSMP and is less influenced by the political economy. It has been successfully achieved in other regions of Ethiopia (BERSMP, 2008). In the explorative Zero-OC scenario C, it can be seen that a 12.6% increase in the prices of forest products per year can lead to zero total OCs of forest conservation at the end of the 20-year implementation period providing an economic incentive for forest conservation without carbon revenues. At year 13 the annual OCs become negative and by project year 20, income from a hectare of forest exceeds that of crops by US$66/ha. Given that after this point standing forest becomes a more economically viable land option, it becomes more likely that despite the time horizon of the project coming to an end, incentives for forest conservation should continue. It is noted, however, that this will require consistent increase in the value of low-impact forest uses.

Efforts are already underway to increase revenues from forested land. Improvements in harvesting, drying and packaging could make forest coffee suitable for export and specialist marketing. Improvements in the management, harvesting, processing, and marketing of honey is also possible. However, consideration is necessary of whether this annual price increase is realistic, as increasing at this rate to the end of project horizon requires a final price 11 times the current prices of forest products. While this might be possible for forest coffee as evidenced by high international demand for premium coffees (Wren, 2007), it may not be possible or maintainable for other forest products such as honey which would rely largely on national product demand. This is further complicated by the differences found in forest income per hectare by village location; coffee and honey are not ubiquitous across the BME (see Chapter 6). The production of forest coffee may also require a degree of excludability, and therefore a clearer understanding of the potential of areas for low-impact forest products is required. In some areas, the calculation of OCs of agricultural land, net of low-impact forest products may therefore be inappropriate. The Bale REDD+ Project implementers in the BME should, therefore, consider in more detail the extent of their influence on changes in productivity that impact on OCs of forest conservation.

The theory of PES predicts that local stakeholders, that are the environmental service providers, will be willing to enter into a contract if the OCs of forest conservation can be overcome (Ferraro, 2008). Thus for REDD+ to be feasible at the case study site, REDD+ revenues should overcome the present value of foregone agricultural production which are the cumulative OCs of forest conservation estimated here. Under more conservative voluntary carbon market prices of US$3/t$CO_2$e, while REDD+ revenues go some way to internalise the positive environmental externality of climate regulation by forests, they are largely insufficient in both scenario A and scenario B to overcome the cumulative OCs of forest conservation until the latter years of the project. At a higher carbon market price of US$6/t$CO_2$e, REDD+ revenue per hectare is sufficient to overcome OCs at all time periods under scenario B.

Scenario C is designed such that total OCs are zero over the full 20-year lifespan of the project. It was therefore expected that REDD+ revenues would exceed OCs. It should be remembered, however, that in this zero-total OCs scenario C, positive OCs are still incurred. Without a REDD+ project, the positive total OCs incurred in the initial 8 years amount to US$57 million. With a REDD+ project REDD+ revenues could be used to overcome these initially positive OCs. REDD+ revenue could be used not only to overcome the positive OCs in scenario C, but these revenues can also go some way to cover the upfront costs of the REDD+ project that were not included in the financial analysis. These costs include the development of project design documents, establishment of CFM units and certification to third-party standards and have been estimated at US$3,225,000 (UNIQUE, 2010).

It is important to note that this financial analysis does not consider the mechanism by which carbon revenues could reach households. With no emission reductions purchase agreement made, there is no benefit sharing framework for REDD+ revenues in the BME. Therefore, it is unclear how much of the REDD+ revenue would reach the households and how much would be directed to other forest stakeholders such as the central or regional government. If large portions of REDD+ revenues are captured by governments, these will be less available to overcome the OCs to household. Furthermore, no information exists about the payment contract that would be necessary under the intervention which would define the timing of payments, length of contracts, and upfront investments required (Ferraro, 2008). The timing of REDD+ payments will be particularly important for the assessment of whether carbon revenues can overcome the OCs of forest conservation. In these scenarios the REDD+ revenues assumed to be realised in the year of avoided deforestation. However, it is likely that REDD+ payments would be received periodically, rather than annually. Given that most households in the BME are subsistence households with few saving opportunities, households may be unable or unwilling to take such a long-term view. A better understanding of their discount rates and the impact of risk on household land use decisions could revise these OC estimates as risk preferences have been shown to impact payment levels required in PES (Ferraro, 2008, Knoke et al., 2011).

With a focus on agricultural expansion as the primary driver of deforestation, the overharvest of high-impact forest products (fuelwood and timber) are omitted as a driver of deforestation. The Bale REDD+ Project intervention plans to meet the needs of households for high-impact products – such as timber and fuelwood – through a number of measures to produce or reduce the need for biomass from the natural forest. Thus these costs to households were excluded from scenario analysis. Further research to generate a model that assesses what fraction of deforestation can be attributable to each driver could also better estimate these costs. Grieg-Gran (2008) for example, assigns Brazilian deforestation to nine different land uses which account for between 1% to 63% of deforested area. With information on the changes in availability of agricultural land, the effects of forest policy on price feedbacks could also be included in the scenarios (see e.g. Busch et al., 2009, Fisher et al., 2011). Such land-market dynamics are commonly ignored in conservation planning (Armsworth et al., 2006). Here too it has been assumed that the price elasticity of demand is zero, or perfectly inelastic. More information would be required on area change to include an estimate of the price elasticity of demand, and to establish if the assumption of zero elasticity over or underestimates these OC estimates of forest conservation. This financial analysis of the OCs of forest conservation in the BME also excludes other environmental and social non-market values of the forest including watershed protection, biodiversity protection, social capital, tenure security and empowerment provided by CFM. These non-market benefits could reduce the net OCs of forest conservation. The inclusion of other non-market values and the OCs of land for other drivers of deforestation as they become available would also be a valuable addition to future research and development of these scenarios.

There are trade-offs in the number of simplifying assumptions made and the complexity of scenario modelling. Alternative methods such as general equilibrium models and economic optimisation models, for example, are better able to consider market feedbacks (e.g. Cattaneo, 2005). In these scenarios prices are not modelled stochastically, with the assumption that prices for products will change linearly and therefore do not capture uncertainty. However, alternative methods are highly data and capacity intensive, thus they can be more difficult to use as an applied conservation tool. A further limitation of these scenarios is that their calibration did not rely on workshops or other participatory methods such as survey or Dephi methods (Höjer et al., 2008). Incorporating more diverse views, particularly of stakeholders and experts in the process would broaden the perspectives included within these scenarios. Despite their limitations, these scenarios serve to instigate discussion on how the private OCs of forest conservation can be influenced by the Bale REDD+ Project implementers in the BME. Scenario modelling of the private OCs of forest conservation is able to highlight the multiple sources of uncertainty that exist, and explore the implications of these uncertainties. As a form of creative visioning, scenarios allow stakeholders to break from established patterns of thinking (Wollenberg et al., 2000). Wollenberg *et al.* (2000) also note how scenarios allow stakeholders to overcome their tendency to overestimate the probability of desirable events.

This analysis of REDD+ revenues challenges the speed and optimism with which many conservation practitioners have adopted REDD+ as a tool to finance forest conservation; REDD+ revenues on their own may not be sufficient to incentivise forest conservation. The expectations of REDD+ in Ethiopia and more globally have been high (Clements, 2010, R-PP, 2011). The aim of this scenario modelling is not obtain a single or 'right' answer. It is to highlight the need for further discussion of the intervention strategy and the potential impact of its objectives. A better understanding of conservation targets, the extent to which Bale REDD+ Project implementers are able to increase productivity of land uses, and to internalise the non-market values of forests, will all help in understanding motivations for resource use and the design of appropriate incentives for forest conservation. Improving this scenarios analysis with more participatory calibration and repeating this analysis as intervention implementation progress is made, and more information becomes available, will allow the review of the progress of the intervention over time. Scenarios thus allow for more adaptive management. By allowing futures to be anticipated, planned for, and adapted to, scenario modelling of the OCs of forest conservation could lead to more effective conservation.

Chapter 8: Household intention to cooperate in forest conservation

8.1. Introduction

8.1.1. Problem statement

CFM establishes a common property regime where members of a well-defined group of people determine collective regulations for resource use, membership, monitoring, and sanctioning procedures (Arnold, 2001, Baland and Platteau, 2003). Such a shift to a common property regime from often *de facto* open access, establishes rights to the use and management of forest resources. It also establishes institutional arrangements that manage and enforce these rights both within the members of a group and for exclusion of those that are not members.

There are growing calls for REDD+ to be delivered via CFM (Klooster and Masera, 2000, Murdiyarso and Skutsch, 2006, Agrawal and Angelsen, 2009, Hayes and Persha, 2010). Both require the bundle of rights and obligations to be recognised by a country's legal system and enforced by its government. However, the economic incentives that underpin REDD+ would also need to operate in the context of the institutions and legal context established by CFM. There would need to be clarification of carbon rights, for example, in addition to forest use rights, and discussions around the role of the collective versus the individual if REDD+ payments were to be made and distributed according to the theory of REDD+, where it operates as a local-level PES mechanism providing economic incentives for the provision of environmental services (see Chapter 2 for other options to fund and implement REDD+).

Historically, CFM implementation has not offered communities payments. Instead, it is assumed that the change in property rights regime leading to increased empowerment and improved governance mechanisms, as well as improved forest resource management, will shift incentives sufficiently to deliver the resource management outcomes (Agrawal, 2003). The behavioural theory of collective action also indicates that the cooperative management of a common pool resource is not only influenced by the income from direct extraction, but by the impact of reputation, trust and reciprocity on households' payoffs, which can be considered as their costs and benefits (Ostrom, 2000, Castillo and Saysel, 2005). There is, therefore, a divergence between dominantly economic incentives for PES and the incentives to engage in CFM.

Few studies have noted that the OC approach to payment incentives in PES ignores the logic of collective action (Kosoy et al., 2008, Muradian et al., 2010). Estimates of the OCs of REDD+ commonly assume that household's act as self-interested profit maximisers thus acting independently of their impacts on others. It is clear, however, that individuals are often influenced by their adherence to social and cultural norms (Baland and Platteau, 1996). These are their preferences for altruism, reciprocity, inequity aversion and conformity with the wider community rather than purely their selfish motivation. This, for example, has been demonstrated by Velez *et al*. (2009) for extraction from a common pool fishery in Colombia.

The incentives for households to engage in REDD+ via CFM are not only potential carbon payments and improvements in forest management regime, but the devolution of use rights, the institutions and social capital established through CFM; therefore, the empowerment to take more control over resource management. These incentives can impact on households' willingness to incur costs of collective action, such as use restrictions, monitoring, patrolling and attending community meetings, so long as others reciprocate and adhere to the institutions put in place (Vatn, 2010).

As well as the omission of the logic of collective action in incentives for cooperation in REDD+, estimates of the OCs of REDD+ have also commonly omitted the non-market environmental values of forest conservation. These include watershed protection, biodiversity protection and conservation of landscape beauty (Pearce and Warford, 1993, Davies and Richards, 1999). The omission of non-market environmental values is largely due to complexity in their measurement and subsequent valuation, with methods varying in theoretical validity and acceptance, data requirements and ease of application (see OECD, 2002, Pagiola et al., 2005b). Their omission is acknowledged by studies as having the potential to overestimate the payment to incentivise forest conservation behaviours (e.g. Naidoo and Adamowicz, 2006, Grieg-Gran, 2008).

The omission of the logic of collective action as well as non-market environmental values could help explain findings where local stakeholders participate in PES where the estimated OCs are not met (Wunder, 2005, Corbera et al., 2007b, Kosoy et al., 2007). Of course, participation despite negative OCs could also result where stakeholders lack information on the market value of services they supply or the experience to truly evaluate the contracts they are offered (Peskett and Harkin, 2007, Kosoy et al., 2008). In order to opt out of PES, stakeholders must also be free from external pressure and coercion which is not always the case (Grieg-Gran et al., 2005, Robertson and Wunder, 2005, Pagiola et al., 2005a, Bennett, 2008). However, the omission of economic values of environmental goods and services provided by forest conservation, and the logic of collective action, both existing and generated by CFM incentives and sanctions, could lead to inappropriately designed REDD+ incentives.

The literature on participation in PES has focused on a locals stakeholder’s ability and eligibility to enter into a scheme, rather than on their desire to participate (Pagiola et al., 2005a, Zbinden and Lee, 2005, Wunder, 2008, Pascual et al., 2010, Sommerville et al., 2010). The literature for participation, or cooperation, on a common pool resource is more substantial. Consensus on an enabling environment for sustained collective action is summarised by Agrawal (2001) as small, interdependent groups that are relatively well off, with adequate underlying technical and institutional capacity and high dependence on forests for their livelihoods (see also Baland and Platteau, 1996). Studies have explored determinants of CFM success with regards to forest condition (Agrawal and Chhatre, 2006, Andersson and Agrawal, 2011); resource appropriators access to forest (Adhikari et al., 2004, Naidu, 2011); and participatory processes (Adhikari and Di Falco, 2009). The literature focuses on the impact of heterogeneity in wealth, interest, and social factors, although the outcome of these factors remain debated (see Poteete and Ostrom, 2004, Naidu, 2009).

Studies of cooperation in CFM, however, rarely consider how households' perceive the resource system or intervention. Matta and Alavalapati (2006) is an exception that considers participants perceptions of a CFM intervention in India. They find that those who believe environmental problems to be of high concern in their village and those with greater levels of education rate the overall perceived performance of the intervention more highly; where rated on a scale of 1 to 5 where 5 is the highest. It is noted that in this study the perceptions of the overall performance of the CFM intervention depended on individuals' initial perceptions of environmental problems and what they perceived the benefits of CFM were and so each person has a different baseline. Therefore, overall success and sustained participation in the studied CFM intervention will be a function of the perceptions of participants who vary in knowledge, understanding and beliefs. As with other studies of cooperation in CFM, however, Matta and Alavalapati (2006) is an ex-post assessment and not applied for intervention design or planning. However, if the factors which improve cooperation in an intervention are known, they could be actively promoted (Cavalcanti et al., 2010).

The attitudes and perceptions of those local to interventions has been recognised as important for success in the conservation literature (Zanetell and Knuth, 2004, Davies and Hodge, 2006, Nkonya et al., 2008). Zanetell and Knuth (2004) in their analysis of participation in community-based fishery management, find an unwillingness of respondents to participate where insurmountable problems were perceived, whereby villagers did not believe that anything they could do would alleviate the decline of the fishery in Venezuela. Communities have also been found less willing to support community based management where they believe the capacity of community institutions to undertake these challenges to be weak (Masozera et al., 2006). In Uganda, a survey of attitudes to a community conservation programme seven years after the intervention began, found communities to be critical of the conservation intervention with largely unchanged behaviour and high level of illegal activities (Infield and Namara, 2001).

The PES literature, more recently, has also called for more attention to be paid to stakeholders' attitudes and perceptions in PES schemes (Corbera et al., 2007a, Kosoy et al., 2008, Petheram and Campbell, 2010). An understanding of a stakeholder's attitude towards forest management and the use of the resource base will allow more consideration of socio-cultural factors for cooperation that go beyond payment incentives that PES theory highlights (Ferraro, 2008, Wünscher et al., 2008).

8.1.2. Aims and objectives

The Bale REDD+ Project has been proposed and initiated by the Government of Ethiopia (Oromia Regional Government, Bureau of Agriculture and Rural Development, and the Food Security and Disaster Prevention and Preparedness Commission) and NGOs FARM-Africa and SOS Sahel Ethiopia: the Bale REDD+ Project implementers. The project area covers 900,000 ha including dry and moist tropical forest which is currently being lost at 4% annually. In order to reduce deforestation over a 20 year period, CFM will be implemented alongside promotion of fuel-efficient stoves and biomass briquettes and plans are underway to plant woodlots and manage fire outbreaks. Increasing agricultural production and the value of NTFP will also occur as part of the project. While CFM and REDD+ can both be undertaken as separate policy interventions, in the BME these are considered together: the Bale REDD+ Project undertakes REDD+ via CFM. Thus emission reductions do not have to be additional to that achieved through CFM, but rather are those generated by CFM (see Chapter 3 for a full description of the Bale REDD+ Project).

Chapter 5 indicates that, while uncertain, emission reductions and positive REDD+ revenues are feasible at the case-study site. The 20-year project could generate 180ktCO_2e, even when accounting for only above-ground tree biomass. In Chapter 6, high positive OCs of forest conservation are found from foregone agricultural production, net of low-impact forest product harvesting. Average OCs of agricultural production US$334/ha in the survey year and Chapter 7 indicated that REDD+ revenues may be insufficient to overcome these OCs.

This Chapter investigates the local attitudes to resource management and conservation at the case-study site in order to complement the empirical estimates of households' OCs of REDD+ via CFM. In addition, the cooperative intention of a household is also assessed using a voluntary contribution to secure forest resource benefits into the future through CFM. Multivariate regression is used to investigate households' strength of intention to cooperate under a proposed CFM regime. I add to knowledge by considering how ex-ante information on attitudes and perceptions can be used in intervention design to encourage cooperation between households. I then outline the implications of these findings for REDD+ via CFM at the case-study site.

8.2. Methods

8.2.1. The REDD+ via CFM, Bale REDD+ Project

Although the extraction fuelwood for sale and timber extraction is illegal, in the status quo, the forest resource is a *de facto* open access regime. This is due to a lack of enforcement of forest laws. Each user is capable of subtracting welfare from other users and forest use is presently reported to be unsustainable with deforestation rates at 4% annually (BERSMP, 2006, BMNP, 2007). In order to tackle nationwide forest losses, Ethiopia is supporting the roll-out of CFM; the policy and legal framework of which is driven by the 2007 proclamation for Forest Development, Conservation and Utilisation. The state retains the rights to land, but use rights for forests are agreed with communities.

A proposed Bale REDD+ Project in the south-east of Ethiopia will devolve management responsibilities to communities through CFM while generating emission reductions through avoided deforestation. It is therefore a REDD+ via CFM project. By implementing the CFM regime, communities will not have property rights to the land but rights of access, withdrawal, management and exclusion of certain forest products. They legally have claim to the benefit streams from low-impact forest products (climber, coffee, honey and bamboo), and must restrict their extraction of high-impact forest products, namely, fuelwood and timber (see Section 4.3.2.2.). The expansion of agricultural land will also be strictly controlled. These rights will be transferred through forest management agreements made with the Bale Forest Enterprise and CBOs, groups of about 30 households, and will come with duties such as attending meetings, monitoring and patrolling of a defined forest area. Thus the establishment of CFM is not a complete transfer for rights, but a partial transfer where the state retains the ownership of land. Although yet to be confirmed, the Bale Forest Enterprise is likely to retain the carbon rights for emission reductions generated and benefit-sharing mechanisms have yet to be discussed and negotiated (see also Chapter 3, Section 3.4).

Historically in the BME, forests were under traditional management systems and elders were responsible for the introduction locally agreed rules and norms of resource us (Wakijira et al., *in press*). While these traditional forest management systems have been eroded by central government over time, some do exist to manage other communal resources in the region such as mineral springs for livestock and for the harvest of forest coffee (Chiodi and Pinard, 2011). The existence of traditional councils exist can prove conduits for organised local level forest management (Wily, 2010), which is promising for the implementation of the Bale REDD+ Project as REDD+ via CFM.

The Bale REDD+ Project is still in a design phase, however. With the intervention yet to be implemented in survey villages, the description of CFM in the BME relies on predicted rules of the intervention. During implementation forest use rules will be established by user groups including; what to extract, to what level, and how to distribute that extraction and sanctions if rules are broken. The exact conditions placed on households will, therefore, be negotiated with CBOs and households through the process of CFM establishment. The description as given to survey respondents was, therefore, based on the available information on the intervention at the time. It is assumed that high-impact forest product harvest and expansion of agricultural land will not be allowed, while low-impact forest product harvest will be allowed under the proposed Bale REDD+ Project. The negotiation of forest management agreements will also clarify the transaction costs that households would need to incur under the new forest management regime. These include negotiation, monitoring and enforcement costs (Adhikari and Lovett, 2006). It is likely that in reality, some households will make greater contributions to cooperative efforts through monitoring and enforcement, or administrative duties for CBO groups (Adhikari and Lovett, 2006; Meshack et al. 2006). However, these transaction costs of CFM on households cannot be established at present and these costs were not estimated for any households in this study (see Section 4.3.2.2).

8.2.2. Households' environmental and forest management attitudes

The environmental attitudes held by individuals derive from their underlying values and beliefs, thus qualitative study of opinions and perceptions of conservation interventions allows unobservable values to be better understood (Kotchen and Reiling, 2000). At the case study site, environmental attitudes to forest management were elicited in agree/disagree statements. This included anthropocentric motivation for resource use and non-market values, including option value for future direct and indirect use, and non-use values that capture the value of the forest existence and bequest for future generations (Davies and Richards, 1999). In addition, statements explored the general context and trade-offs households make between environmental concerns and with other concerns of wellbeing and competing livelihood priorities (see Table 25). A series of open-ended questions then explored opinions of past, present and proposed forest management regimes in the survey villages.

A description of the Bale REDD+ Project intervention was then read out to survey respondents. This included CFM and the potential for emission reductions to be generated for which international payments might be available (see Appendix 1). Following this, respondents were asked if they would take part if the project came to their village. In agreeing to take part it is assumed that households undertake a calculus of the costs and benefits they perceive they will incur (Lubell, 2002). In order to assess their perceived net benefits of the scheme, households were then asked if their welfare and their income expectations would change in the year following the implementation of the intervention if it was to take place. This would include an internal calculation of the costs and benefits of restrictions on their use of land and forests in addition to what they perceive they might receive through REDD+ payments.

While households received a description of how community cooperatives could receive money for keeping the carbon in the trees through REDD+, no anchor of the scale of the revenues that would be available through REDD+ was provided as this is not yet determined by the project and there was a fear of raising expectations by the Bale REDD+ Project implementers. This makes the interpretation of the perceived income expectations complex, thus it is not taken here to be a welfare measure or analysed purely on its own. Households were also asked, however, what they would spend carbon revenues on if payments were made. Household responses were coded for community, private or a combination of community and private goods. Community goods are defined as those that benefit the community at large, for example using the money for roads, bridges, and clinics, whereas private goods are those that benefit the household, for example using the money for trade, to school children, or to purchase livestock.

Respondents were also asked if they believed other households would participate and follow the rules as outlined in the description of the Bale REDD+ Project. This question was intended to determine if respondents would trust other households and to assess if they believed other households would reciprocate in cooperating in the communally used forest area.

Table 25. Attitudinal statements towards environmental values and livelihoods. Agree/disagree statements to which households responded in the semi-structured survey and a description of the values they relate to.

Value elicited	Statement	Description
anthropocentric	*"If an area of forest is not being used by people it is not a problem if the forest area gets smaller"*	The anthropocentric statement considers the instrumental nature of household environmental values
selfish	*"Even if my household does not use a part of the forest, I would participate in this forest management"*	The selfish statement highlights whether households consider the wider community or are motivated purely by private returns
option	*"I do not think about my household's use of forest in the future, it is enough to think only about my households use of the forest now"*	Option value concerns goods and services that are not used at present, but have the potential to be used directly or indirectly, in the future
bequest and altruistic	*"There is a responsibility for me to manage the forest well now so that my children and future generations can benefit from forest in the future"*	The availability of goods and services to be used by future generations is referred to as bequest value and by current generations as altruistic value
existence	*"One management objective for forests in the Bale Mountains should be to support wildlife that lives there"*	The satisfaction in knowledge that services merely exist is classified as existence value
trade-offs	*"Forests should be managed only if this does not negatively affect people's livelihoods"*	Trade-off and general context statements highlight the prioritisation of forest conservation against other livelihood needs and well-being concerns
(a) general context	*"People have more important things to worry about than good management to maintain the forest"*	
(b) general context	*"People only cut down the forest because they have no other way of supplementing their livelihoods"*	

8.2.3. Households' intention to cooperate with the Bale REDD+ Project

Cooperation in this study is defined as not only entry into the CFM scheme, but also abiding by the intervention rules and undertaking pro-conservation behaviours. Within a household's cooperative intention, it is assumed that households will consider their OCs in addition to the value that households place on the devolution of use rights, on the institutions and social capital established, or on empowerment provided under a new CFM regime, and any transaction costs that CFM will impose on a household.

A proxy of the strength of a household's intention to cooperate was elicited with the question: '*to secure benefits into the future, would you be willing to give up some of your yearly income to your CFM cooperative so that they could better manage the forest?*' and a follow-up question that elicited the voluntary contribution amount. This question is relates to a contribution towards sustainable forest management as defined in interviews as; maintaining the forest area and quality so as to provide benefits and income into the future. It was elaborated that high-impact forest products would need to be harvested without detriment, as would any grazing of livestock in the forest, but low-impact forest products would not be affected under the proposed sustainable forest management regime. It was also made clear that this would entail the creation of a community based group with full responsibility for the outcome agreed with the authorities. As traditional forest management used to be commonplace in the Bale Mountains (see Chapter 3), this is a management regime that the respondents were able to relate to and understand.

The voluntary contribution is a hypothetical question, but where respondents reported future behaviour is assumed to illustrate their preferences or values. The hypothetical nature of the question means that no real economic commitment of the individual is required and this may lead to hypothetical bias in the response through warm glow effects (Andreoni, 1989). The voluntary contribution is also not incentive compatible and therefore is not used to estimate a welfare measure in this study (Champ et al., 1997). A result of the hypothetical and ex-ante question, individuals may also lack familiarity with the operation of the proposed community based organisations (CBOs); the small groups of households that sign forest use agreements. In light of the Bale REDD+ Project being implemented through CFM and the possible, but uncertain, nature of payments to communities from REDD+, it is clear that there respondents will be confounding a multitude of incentives. These include potential monetary and in-kind incentives from REDD+ as well as the aforementioned benefits from cooperation under CFM. It is therefore, not possible to distinguish respondents attitudes towards REDD+ separately from those towards CFM. The value is also not considered a measure of willingness-to-accept. The question was made as real as possible, however, with a description of the intervention and by indicating a clear form and frequency of payment. The CFM cooperative was also described to respondents as an honest, credible and reliable organisation with committees, rules and responsibilities (see Appendix 1).

Acknowledging the limitations of the voluntary contributions approach, it is assumed that a household's responses will reflect their future behaviour. The magnitude of the contribution elicited is assumed a quantitative indicator of behavioural intention to cooperate in the conservation intervention. Where a positive contribution exists, it indicates that the household is willing to move from the status-quo to the CFM regime (as outlined in Section 8.2.1) and it demonstrates a willingness to adhere to the rules as explained.

8.2.4. Econometric analysis

The determinants of cooperative intention, using a respondent's willingness to pay into the cooperative CFM group in order to contribute towards sustainable forest management as a proxy, were analysed using an ordinary least squares (OLS) regression (see Chapter 4). Explanatory variables were selected based on *a priori* assumptions of their impact (Table 26). These variables include socio-economic factors that might influence the voluntary contribution including income from agriculture, low-impact forest products (coffee, honey, climber and bamboo), and high-impact forest products (fuelwood and timber); and the households income expectations under the Bale REDD+ Project intervention. Demographic factors included were household size, years in village and the education of the household head.

The voluntary contribution, high-impact forest product income and crop income were log transformed. This log transformation allows for a non-linear relationship between the dependent and independent variables. It also normalises the residuals and reduces potential outliers with the dependent variable more likely to be normally distributed. Two zero values for the voluntary contribution were assigned the next smallest value of US$0.15 before log transformation. A dummy variable was generated for the presence or absence of low-impact forest product income, as 58 households had zero values. No strong colinearity between and of the independent regressors was observed (Table 27).

Table 26. Explanatory variables of intention to cooperate in the proposed Bale REDD+ Project intervention.

Description, hypothesised impact and justifications for explanatory variables used to explore a proxy for a household's intention to cooperate in the Bale REDD+ Project intervention through their willingness to give up a proportion of their income.

Explanatory variables	Explanation	Hyp. impact	Justification
lifp_yn	*Dummy variable for income from low-impact forest products (bamboo, climber, coffee and honey); 1= income greater than US$0, 0= no income in the survey year*	+	It is predicted that if HH derive income from high-impact forest products, their contribution towards continuing to receive these benefits into the future will be higher (Baland and Platteau, 1999, Agrawal and Chhatre, 2006).
log_hifp	*The log of the income a household derived from high-impact forest production (timber and fuelwood) in the survey year*	-	It is predicted that households would be less willing to contribute towards forest conservation given that they will have to undergo restrictions in their use.
log_crop	*The log of the net income a household derived from cultivated land in the survey year*	-	With higher income from agriculture, an alternative source of income to forests, HH are expected to have a lower contribution.
income_expectation	*The amount by which the HH thinks their income will change under the CFM intervention*	+	The higher the expected income change in the first year following the intervention, the higher a household's contribution is hypothesised to be. It is recognised that more research is required to understand the determinants of this variable, however.
HH_size	*The total number of people in the household*	-	With more mouths to feed, it is predicted that the contribution will decline as other needs dominate use of income.
education	*The number of years of education of the HH head ranging from 0 to 13 years*	+	Education is expected to increase pro-conservation behaviours as well as improving knowledge and skills to extract forest products more sustainably and cultivate land more intensively (Godoy and Contreras, 2001, Adhikari et al., 2004).
village_years	*The number of years the HH head has lived in the village*	+	Experience through age and knowledge during length of residence is also expected to increase pro-conservation behaviour and thus the contribution.
Agarfa	*Location dummy variable; 1= Agarfa, 0= not Agarfa*	Dummy variables for location were included in the model to control for village and forest fixed effects.	
Goro	*Location dummy variable; 1= Goro, 0= not Goro*		

Table 27. Correlation matrix of independent variables.
Correlation between right hand side variables used to predict household intention to cooperate in CFM used to assess the risk of colinearity.

	lifp	log_hifp	log_crop	income_expectation	HH_size	education	village_years
lifp	1						
log_hifp	-0.13	1					
log_crop	0.01	0.09	1				
income_expectation	-0.05	0.10	0.21	1			
HH_size	-0.06	0.15	0.22	0.10	1		
Education	0.08	0.12	0.02	0.23	-0.09	1	
village_years	-0.25	-0.10	-0.17	-0.04	0.07	-0.31	1

8.3. Results

8.3.1. Exploring attitudes to forest conservation

Perceived changes in the forest resource base

Of 237 household surveys, seven households were dropped where crop production was absent or failed in the survey year. A further two households were dropped due to apparent misreporting of yields leaving 228 surveys. Households reported travelling on average 6km to gather forest products, with two households travelling as far as 16km. 90% of households reported that the distance they travelled this year was more than last year by an average of 0.78km, but ranging up to 2km. 98% of households reported that they are likely to need to travel further next year with only two households reporting they would travel the same distance next year.

Given these results, it is unsurprising that 87% of respondents do not believe that the current level of forest use by their household and others in their village is able to continue into the future; 1% report that they do not know, while only

11% believe current levels of forest use are sustainable. These changes were attributed to the decline in the forest area as well as forest clearing without any planning, replanting or conservation measures. Underlying causes are reported as increasing family sizes, immigration and lack of alternatives, while the proximate drivers of deforestation are identified by the respondents as clearing for agricultural land, increasing fuelwood demands, an increase in grazing in the forest and an increase in the number of landless people. It is also mentioned that more people are gathering forest products to sell as well as to use within their households in recent years. Box 1 presents some quotes from the respondents when asked about the sustainability of the forest resource base. Overall, it is clear that the forest is declining and those that do think forest use is sustainable, largely do so on the proviso that the government intervene.

Box 1. Survey responses to the question "*Do you think that the current level of forest use by your household and others in this kebele can continue into the future? Why?*"

Selected quotes from those who think forest use is not sustainable:

– I can say forests will last for two years here after as it is badly being cut down for different uses.
– As it is extremely deforested it will not last for two years.
– If this situation continues it will not pass to the next generation because the cutting of forest is increasing. Forest conservation is not known in this area. There is even not enough forest for this generation.
– Because most people are cutting down the forest without thinking about the next generation. If the government controls the forest and teaches the community about the forest benefit the forest management will be changed for the next generation.
– Because the number of families are increasing and so the landless people increase. Even the students stop their education and start cutting down the forest for selling. For fuelwood other people are coming from other kebeles and for these reasons the forest will not pass to the next generation.

Selected quotes from those who think forest use is sustainable:

– If the government control the forest from time to time, if the community get education about using the forest sustainably and if the community gets a payment, the forest will pass to the next generation.
– If the forest conservation is continuing, there will be forest for the next generation because the forest conservation in this kebele is good at the moment.
– Because we are going to conserve the forest so that it can be passed to the next generation.

Attitudes to the environment and other values derived from the forest

There was very little variation observed in the attitudinal survey data, which illustrated that respondents generally disagreed with anthropocentric motivations of forest use and with selfish motivations for forest use. Households also held a strong desire for forest to be available for future use (bequest, existence and option value). The general disagreement with the trade-offs statement implies that households are willing to accept a reduction in private benefits in aid of forest conservation. It is also clear from the general context statement responses that livelihood and income security were high on the household agenda, with households believing that forests are a way of supporting livelihoods when other livelihood generating activities are not available (Figure 20).

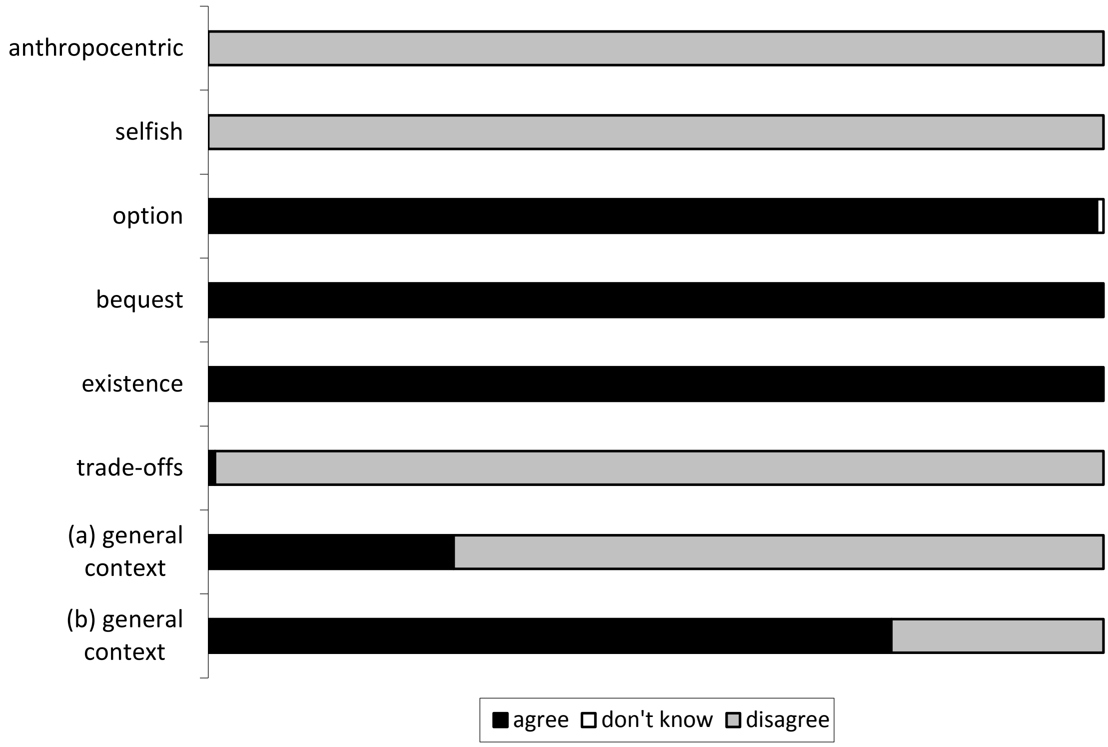

Figure 20. Responses to attitudinal statements of environmental values and livelihoods. Agree/disagree and don't know responses of survey respondents relating to underlying values (see Table 25 for a description of the statements).

Opinions on past and current forest management

Many respondents reported a much stricter forest management regime under the Derg regime (before 1991). Permission from village authorities was required for timber for house construction and the gathering only of deadwood for fuelwood was permitted. Most report that this strict management was 'better' due to a combination of government control and traditional management by village elders. A number of respondents, for example, reported that they were required to plant fast growing eucalyptus under the Derg regime and sale of forest products were restricted. Elders reportedly taught people about tree growth and how to harvesting biomass but conserving the whole tree. However, there are also reports that restrictions were not always adhered to, although forests were able to meet livelihood needs as, before 1991, there were fewer forest users and fewer forest uses; deadwood was much more abundant, young men were recruited for national military service, and livestock ownership was more prevalent.

Responses to questions on the current forest management in the area that the household uses reflect a decline in adherence to restrictions on use, advice from village committee members, and a loss of forest area. It is widely acknowledged amongst respondents that prohibitions exist for clearing forest and collecting some forest products for sale, but many respondents report the need of use for survival and therefore disregard of these rules. There are reports of competition between communities as well as within communities as a result of increasing family size, increasing number of families and increasing number of landless. Households report that they now gather forest products for sale, rather than predominantly for use within the household. However, a small number of respondents reported that current forest management is improving as a result of initiatives, by the government, to teach and implement CFM (Box 2).

Opinions of community forest management

Responses to an open-ended question of how forest management might change if communities were given legal rights to use the forest and forest products, gave overall support for the devolution of management and REDD+ via CFM. A number of households, however, stated that forest management would improve only if education, awareness, advice and supervision are also provided by the authorities. Some noted that the village could provide greater control and management as the committee and people in the village are more aware of who is deforesting and who is not. In contrast, some respondents were less enthusiastic about the transfer of rights, believing that handing over rights could result in worsening forest management, particularly mentioning that changes could not occur unless the landless and the poor who rely on the forest are provided with alternative opportunities (Box 2).

Following a description of the CFM scenario, respondents were asked if they and households in their village would join a CFM cooperative. The respondents showed high support for the proposed CFM intervention with 100% of respondents reporting that their household would join CFM. Furthermore, 79% of respondents believed that other households in the village would also join a CFM cooperative; the remaining 21% did not know if other members of the village would join a CFM group. When asked how certain they were that other members of a cooperative would follow the rules to maintain the forest area under CFM, 83% were certain, while 10% were uncertain and less than 7% of households did not know.

Box 2. Survey responses to questions on past, current and future forest management regimes

Selected quotes on opinions of past forest management:

- There was strict protection of forest in the past. We used only the deadwood for fuelwood and we had to have permission from the kebele authorities for house construction.
- Past forest management was much better than the current management. Protection of the forest was much stricter and communities were forced to plant eucalyptus around their home. The number of forest and forest product sellers was also much less than currently.
- Even if there was protection of forested we used to cut it for fuelwood and construction. In the past, however, there was more forest and less forest users. Most young men were recruited for national military service then, now it is being deforested by young men.
- The situation under the Derg was very good. At that time elders taught people how to keep the forest according to the traditional culture.
- In the past the forest was protected very well. During the Derg government the person who was cutting down the forest was punished. Even the person cutting down climber without permission was punished. For this conservation the weather condition was good during this time.

Selected quotes on opinions of current forest management:

- The current forest management is not good because of the increasing deforestation. Now we used the forest products for home and for selling, for everything!
- Now we have to cut down forest for fuelwood and timber as there is shortage of deadwood and fallen wood. By law it is forbidden but most communities do not obey it.
- We collect deadwood for firewood. For construction we are forced to cut down forest. But we do not openly cut down trees, we hide ourselves. Forest are being extremely unsustainably cut down.
- The past forest management was better than the current one because forest protection was respected and the communities used to obey the management.
- Currently, people are using forests for house building, fuelwood and selling forest products. Other people also come from other kebeles to cut down the forest.
- The government are now teaching us about the forest benefits because the climate change affects the crop product and brings about famine.

Selected quotes on opinions of giving community rights to the forest:

- If the community gets the legal rights and they are taught how to conserve and how to use and save the forest the management will change.
- The legal rights and education should be given in how to conserve or protect the forest. Representatives should be selected from the associations, and there should be checks and controls on how they are organised and how the forest is being conserved from time to time.
- If the forest is given to communities, the communities will control their resources and the management will be better. This is because people know each other and they control each other very well.
- There will be a big change because the committee and the people in the kebele recognise who is deforesting and who is not.
- There will not be any change. Because if we get the right, we will just cut it down.
- I don't think the forest management will change as most of communities in the kebele get benefits from forest and forest products and they only think for themselves.
- Unless landless and the poor who live on and rely on forest get a solution to their problem there will be no change.

8.3.2. Perceived welfare and income expectations under CFM

84 households had to be dropped from the sample due to enumerator misunderstanding of the perceived income change and voluntary contribution question. This reduced the sample size, but as the enumerator was employed over all survey villages, this did not lead to geographical bias. Nine further households were dropped where contribution data were missing from the enumerators' data books. Of the remaining 135 surveys, 99% of households reported that their welfare would increase as a result of CFM, 1% (two households) reported a perceived decline in welfare; where welfare is defined without time delimitation and not restricted to a household's change in income. The households that thought that their welfare would decline under CFM reported that they felt that restrictions would mean they would not be allowed to sell and use as many forest products as they needed. Only one stated that there would be too many interests under the cooperative so that he could not do as he pleased and would be influenced by other group members.

Households that thought their welfare would improve largely attributed this to increasing incomes from the forest through products such as coffee and honey. A number of survey respondents also noted that in cooperatives, members are able to generate more income working together than they would on their own, benefiting from cooperation as well as more experienced individuals, or indeed any job activities that might arise through the cooperative. One household believed that the formation of a cooperative would allow work to begin on a road and bridge such that their well-being would be increased in the future. Another noted that improving the forest condition will attract wild animals and therefore tourists, to the area.

Improved crop production, as a result of improved local climate regulation, and income from possible carbon payments were mentioned by respondents when explaining improvement in welfare. This raises concerns over high expectations under forest conservation, particularly for the return of good rainfall for crop growth in the immediate term. However, when asked to think five years into the future, as well as similar mentions of the climate returning to normal, respondents talked much more broadly about the benefits from forest conservation including; improved seedling survival, increasing forest quality and flowering leading to improved honey yields, watershed benefits, and the persistence of forests for the next generation to use. These speculative expectations of the short-run benefits of forest conservation will need to be managed from a policy intervention perspective.

Perceived income expectations of CFM were high in the period of one year after the implementation of the Bale REDD+ Project intervention. 99% of households perceived that their income would increase under CFM, and again two households believed their income would decline. These findings align with the above perceived welfare changes of households, which suggest households consider income very important to their welfare. The average income increase expected was US$286 ± 38. There was a large variation in the expected changes; households income expectations ranged from a loss of US$449 to a maximum increase of US$1,498 (Table 29).

When asked what money from payments for carbon storage could be used for 51% of respondents suggested only community goods. The most popular community good being the construction of a mill, followed by that of a school. 29% of respondents reported only private uses with most popular response being trading followed by livestock. 20% of respondents suggested a combination of private and community benefits could be funded. While the highest frequency response of trading appears an environmentally benign use of payments, the second most frequently reported private use of payment is livestock (Table 28). Where payments are designed, this indicates that secondary issues may well emerge. 40 respondents also suggested that the money should be reinvested in forest conservation.

Table 28. Survey respondents' reported desired use of carbon revenues by community and private goods.
The suggested uses of REDD+ revenues that could be received under the proposed REDD+ via CFM intervention, divided into community goods that benefit multiple households and private goods that benefit only the survey household, with frequency of response reported in brackets.

Community use of REDD+ revenue	Private use of REDD+ revenue
Mill (water and electric)(67) School (52) Irrigation (42) Bridge (41) Forest Conservation (40) Track (17) Transport (car/bus/lorry)(13) Road (13) Waterpipe (9) Clinic (8) Electricity (5) Industry (4) Welfare for elderly (3) Conserve wild animals (1)	Trading (106) Livestock (cattle, oxen and sheep)(58) Fertiliser (24) Harvest machine (farming equipment)(19) Family improvement (e.g. teaching children)(17) House building (11) Seeds (6) Plantation (4) Saving (2) Coffee for trade (1) Farmland (1) Modern beehive (1)

8.3.3. Households' intention to cooperate in CFM

8.3.3.1. Descriptive statistics

In response to the question '*to secure benefits into the future, would you be willing to give up some of your yearly income to your CFM cooperative so that they could better manage the forest*?', 99% of households were willing to contribute some of their yearly income. Only two households were not willing, stating reasons of not being able to afford it with one household stating that their income would decrease under the intervention, thus they would need support to pay into the scheme. The mean of the voluntary contribution into the cooperative CFM group was US$11± 4/year. The distribution of the contribution is highly skewed with median voluntary contribution of US$4.49 per year (Table 29; Figure 21). A significant difference by location is found at the 5% level (Kruskall-Wallis, K=8.140, df=2, p=0.0171**). The trend between survey locations followed that found for forest incomes and was opposite to that found for households' income expectations under the intervention: voluntary contributions were highest in the moist forest of Delo Mena where high value coffee grows and honey is a forest product option. The voluntary contribution was lowest in the woodland of Goro, where neither coffee nor timber, are viable forest products (see also Chapter 6).

Table 29. (a) Mean household income expectation under proposed CFM regime and (b) willingness to pay into the cooperative CFM group by location.
(a) The mean and median household income expectations (US$) in the first year of implementation of the Bale REDD+ Project intervention reported by survey village location with 95% confidence interval as well as median and range. No significant differences were found between villages: Kruskall-Wallis, K=0.931, df=2, p=0.6279; (b) The mean and median voluntary contribution in order to contribute towards sustainable forest management, a proxy for a household's intention to cooperation in CFM, is reported by survey village location with 95% confidence interval as well as median and range of values elicited.

Location	Mean	Median	Min	Max
(a) Income expectation under proposed CFM regime				
Agarfa (n=35)	280 ± 99	225	-449	1,124
Goro (n=23)	323 ± 120	225	15	1,498
Delo Mena (n=77)	278 ± 36	225	11	749
All (n=135)	286 ± 38	225	-449	1,498
(b) Voluntary contribution to the community cooperative for sustainable forest management				
Agarfa (n=35)	8.65 ± 3.79	7.49	0.00	52.43
Goro (n=23)	3.78 ± 1.60	2.25	0.15	14.98
Delo Mena (n=77)	14.42 ± 6.28	5.99	0.30	149.80
All (n=135)	11.11 ± 3.78	4.49	0	149.80

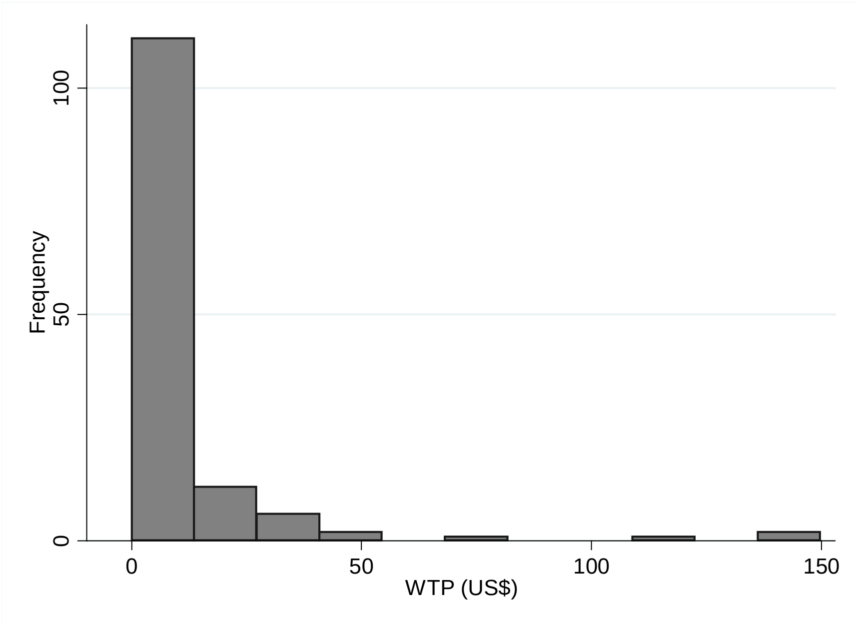

Figure 21. Histogram of household's willingness-to-pay into the into the cooperative CFM group.

8.3.3.2. Econometric analysis

An ordinary least squares regression model was employed to investigate the determinants of the voluntary contribution to the cooperative CFM group, a proxy for cooperative intention. Comparing the results to hypothesised impacts on the contribution level, many of the independent variables show the expected direction of the coefficient (Table 26; Table 30). The R^2 value indicates that the model explains 29% of the variation, thus there are other factors not observed here which impact on the contribution amount. A Ramsey RESET test was used to assess the functional form of the model to detect omitted variables, suggesting that the model was not miss-specified (Ramsey RESET, F(3, 121)=1.06, p=0.3673).

Table 30. Determinants of household voluntary contribution into the cooperative CFM group.
OLS regression results for predictors of household voluntary contribution towards sustainable forest management, a proxy for a household's intention to cooperation in CFM. Showing beta coefficients, robust standard errors in parentheses and significance where; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Determinants	OLS dependent variable: log_voluntary contribution
lifp_yn	0.538*** (0.0971)
log_hifp	0.374** (0.177)
log_crop	0.278** (0.140)
income_expectation	0.00109*** (0.000367)
income_expectation2	-0.000000629*** (0.000000233)
HH_size	-0.0583*** (0.0175)
education	0.00840 (0.0190)
village_years	-0.00300 (0.00428)
agarfa	0.463*** (0.114)
goro	-0.0840 (0.158)
constant	-1.215** (0.557)
N R^2	135 0.286

Forest income and crop income

Households that derive more income from the forest contributed more of their income for sustainable use of the forest into the future. This was true for low-impact forest products of honey and coffee, as hypothesised, as well as high-impact forest products that will be limited under the intervention. Given that the positive relationship between the voluntary contribution and high-impact forest product income exists despite of the proposed limitations of the intervention for high-impact forest product extraction, it could mean that forests are being significantly degraded that individuals believe that forests cannot continue to be used in the present way. This would agree with the qualitative statements of many respondents. Crop income was predicted to negatively impact the contribution amount, however, no such relationship was found. In fact, a 10% increase in crop income predicts a 2.7% increase in the voluntary contribution. This may be due to the small variation in agricultural incomes found at the case study site (Chapter 6). It may also be due to higher ability to pay due to the alternative income source. With no instrument available, it is not possible to disentangle these effects further without further research.

Income expectations

The income change that households expect in the first year of the Bale REDD+ Project intervention was found to be a non-linear inverse U-shape. It is first a positive determinant of the voluntary contribution, then after a point, a negative determinant. This relationship with cooperative intention shows a diminishing impact of expected income on the contribution amount after a turning point at US$833, which falls within the range of income expectations between US$ -449 and US$1498. Interpretation of the coefficient shows that while significant, the marginal effect of income expectations are small; a US$1 increase in income expectation increases the voluntary contribution by 0.07%. It is interesting to observe this finding even when the village level effects are controlled for, thus income expectations are not limited to locations where forest returns are highest. The findings of perceived income expectations are interpreted with caution as there are many factors that, themselves, will influence these perceived income expectations and in the absence of a clear instrument, this analysis is limited.

Demographic variables

As predicted household size is a negative determinant of the voluntary contribution, likely due to competing demands for a large household's income. An additional member of the household is predicted to reduce the voluntary contribution by close to 6%. It was predicted that the contribution amount would increase as a result of education as well as age and experience on pro-conservation behaviours as predicted in Chapter 6. However, no such relationships were found.

Location

The impact of location on the voluntary contribution amount illustrates that households in the dry forests of Agarfa have higher intention to cooperate through the voluntary contribution proxy. This is interesting given that descriptive statistics indicate highest contributions occur in the moist forests of Delo Mena. However, as income from forest sources have been controlled for, the locational differences may also capture village level differences in social factors such as trust, reciprocity and reputation known to impact on cooperation on a common pool resource (Ostrom, 2000, Castillo and Saysel, 2005). It may also represent different perceptions of the decline of forests in survey locations; households in Agarfa may be willing to contribute more if they perceive the decline of forest area to be of greater concern than at other survey locations.

8.4. Discussion

Households in the BME hold strong pro-conservation attitudes. It is clear, however, from general context statements that other livelihood concerns compete with those of forest conservation, for example food security, health and education. However, households appear willing to accept proposed Bale

REDD+ Project restrictions in return for improved forest management. On the whole, households are largely aware that forest use is unsustainable and some believe that current levels of forest use will result in complete forest loss in their village after only two years. Travelling further each year to collect forest products, households attribute forest losses to agricultural expansion and fuelwood collection, the reported drivers of deforestation in the region and country (BERSMP, 2006, R-PP, 2011). They also note that grazing in forests is increasing and inhibiting tree re-growth. Population growth in the BME is also seen as a clear driver of deforestation by survey respondents.

The pro-conservation attitudes and values held by households, in combination with the observed rapid decline in forest resource, could explain the fact that 100% of households were willing to enter into the CFM scheme as it was described to them. The overall disenchantment with the current status-quo forest resource management may also be driving the desire for change. Respondents reported much stricter forest management regimes in the past under the Derg military government and a decline in adherence to rules of forest use in the more recent past. Some respondents even noted that conflict is arising both within and between communities as a result of declining forest areas and breakdown of rules both traditional and otherwise. The potential for conflict supports the calls, by respondents, that education, awareness, advice and supervision are continuously provided by the authorities. The broad call from respondents for good regulatory oversight is also useful for the Bale REDD+ Project implementers. For CFM to be successful, communities need not only values, but the institutions to put these in place which implementers will be able to mediate and ensure persist over time (Gibson and Koontz, 1998).

The high positive response indicating that households' would take part in the intervention is an encouraging finding for the Bale REDD+ Project implementers. Individuals have been shown to be conditionally cooperative on the cooperation of others (Baland and Platteau, 1996, Fischbacher et al., 2001, Castillo and Saysel, 2005). Greater cooperation may well deliver more emission reductions. It is also promising to see that close to 80% of respondents believe that other households will cooperate in CFM and that they will reciprocate by following the rules of CFM. This could indicate high levels of social capital in the BME which can increase cooperation on a common pool resource and therefore longevity of the intervention (Ostrom, 2000, Castillo and Saysel, 2005). These could remain from systems of traditional forest management that existed in the Bale Mountains historically, but also representative of other cooperative natural resource management strategies that currently exist, in mineral licks for livestock, for example (Wakijira et al., *in press*). The intention of households to take part in the CFM intervention is also based on their calculation of the costs and benefits of doing so (Lubell, 2002). Of interest is that a high proportion of households expected both their welfare and income to increase as a result of the intervention.

The income expectations as a result of the intervention were high, however, and there are dangers of such high perceived expectations from a conservation intervention. Matta and Alvapatti (2006), exploring joint forest management in India, find less than 50% of surveyed households perceived any benefit of the forest conservation intervention five years after implementation and were unclear why they joined. Such disenchantment with a conservation intervention is likely to erode its success and longevity.

Many values are contained within a household's perceived payoffs of cooperating in the CFM intervention. In the BME, survey respondents attributed perceived changes in income to increases in agricultural yields due to improvements in the micro-climate, to increases in forest honey and coffee incomes, and the ability to organise into cooperatives increasing business opportunities and the likelihood of group led infrastructure improvements. Carbon payments were also mentioned by a number of respondents as contributing to increasing income. Particularly in light of the perceptions that local climate will change so rapidly, the inability of the CFM intervention to meet expectations in the BME could undermine its persistence. Withdrawal of cooperation will challenge the success and functioning of the CFM, but for REDD+ also threatens the permanence of emission reductions and may well lead to no overall climate change mitigation benefits.

A proxy for strength of cooperative intention, a household's voluntary contribution to the forest management cooperative for more sustainable forest management, was also impacted by a respondent's perceived income benefits from forests, greater income potential through being part of a cooperative and carbon payments. This finding is consistent with the literature that shows that perceived benefits of conservation interventions are important for both participation and cooperation (Matta and Alavalapati, 2006, Sommerville et al., 2010). The effect found here, although significant, has a small marginal effect on the voluntary contribution. It is noted also however, that there may be an element of circularity between households perceived benefits and their intention to cooperate that needs to be investigated further. The household incomes estimated from low-impact and high-impact forest products in Chapter 6, were also found to positively determine the strength of cooperative intent. This reinforces the prediction that cooperation is driven by the salience of the forest resource to households in the BME as found in the wider literature on cooperation (Baland and Platteau, 1999, Lise, 2000, Agrawal and Chhatre, 2006).

Larger households have significantly lower cooperative intention, but this may be due to the competing cash needs within the household. Education and age were both found to not be significant determinants of a household's voluntary contribution. This is surprising compared to other literature that recognises the positive impact of education and experience in pro-conservation behaviour (Godoy and Contreras, 2001, Adhikari et al., 2004). It is possible the impact of education has a more complex relationship than modelled here. For example, Lise (2000) finds that participation in CFM is enhanced by low average family education, but high survey respondent education. Similarly, the predicted effects of age and experience may be obscured by the fact that younger household heads may be more willing to adopt a new livelihood option and older household heads are more risk averse (Ellis, 2000).

As noted in Section 8.1.1., no anchor of carbon payments was provided to respondents. It was also not possible to determine the portion of perceived income to carbon finance as opposed to increase forest or other incomes. The differing perceptions between households mean that the results do need to be interpreted with caution. This is particularly so given the early nature of carbon finance in the Bale Mountains REDD+ project and the lack of benefit sharing discussions and negotiations which means it is unclear if payments will reach households or will distributed at community level. Despite this, it is interesting that over half of the respondents suggested that carbon revenues should be spent on common goods such as a mill and clinic. Such preferences for common goods have been found in Mexico by Kosoy *et al.* (2008) where communities received payments for biodiversity conservation and carbon on watersheds. This, however, complicates REDD+ payments making them less conditional on delivery of the emission reductions, largely irreversible and harder for households to assess if the benefits of the common good will overcome the costs of their cooperation (Gong et al., 2010, Pascual et al., 2010, Sommerville et al., 2010).

Households also stated that livestock might be purchased with carbon revenues. There is therefore a danger of reinvestment in environmentally degrading activities, which highlights the need to include rules for livestock grazing and to assess the impact of livestock on the forest resource base. This could prevent secondary issues from arising after intervention at the case study site. In Zimbabwe, for example, it was found that part of the revenues communities of the CAMPFIRE programme derived from wildlife conservation were invested in agricultural expansion and animal husbandry (Infield and Namara, 2001).

Due to the sensitivity of the data, surveys were unable to elicit total household wealth. The impact of wealth on cooperation in common pool resource management is known although consensus on the direction of the impact does not exist (see Poteete and Ostrom, 2004). Income from forests and agriculture, two of the three main livelihood sources in the region were included in the model and some wealth effects are captured within these variables. However, further research could better account for wealth effects on the voluntary contribution for example through household assets as proxies for wealth, or through expenditure data. It could also be assessed if households would make voluntary contributions in labour rather than cash given that two households could were unable to contribute a cash portion of their income and larger households are likely to have competing demands for cash. The household survey elicited respondents' thoughts on whether others would join the intervention and if others would follow the rules in order to indicate levels of trust and reciprocity. However, in light of low variation in responses to these questions, they were omitted from the regression model. Further research could go into more depth in capturing factors of social capital, including; relations of trust; reciprocity and exchanges; common rules, norms and sanctions; and, connectedness in networks and groups (Pretty, 2003). More attention to behavioural and social aspects could improve the application of these findings for on-the-ground policy design, and draw out village level differences beyond hypothesised ecological forest characteristics.

The limitations of the voluntary contribution approach to households' intention to cooperate in the Bale REDD+ Project intervention mean that the value itself is not interpreted here, but the determinants of the contribution are. Future research could elicit a more incentive compatible value with more effort dedicated to removing hypothetical and strategic biases. A further issue in interpreting households' intention to cooperate in the proposed intervention is that the description of the intervention was necessarily short. A full list of the costs and the benefits of the scheme were not provided to respondents such as transaction costs, or allowable limits for forest product harvesting as these were not yet available. It is possible therefore, that incomplete information results in the high levels of willingness to take part in the intervention. As Adam's *et al.* (2003) note that the calculus of costs and benefits will be dependent on an individual members knowledge and understanding of the collective action at hand. Furthermore, successful collective action is also linked to perceptions of fairness and cost sharing (Bardhan, 2000). This is, therefore, recognised as a major limitation of this Chapter's quantitative measure of intention to cooperate in REDD+ via CFM.

PES is based on a payment incentive changing the behaviour of land use managers to more conservation oriented behaviours. Chapters 6 and 7 indicated that REDD+ revenues in the BME may be insufficient to offset the estimated OCs of forest conservation. An integrated study of REDD+ via CFM in the BME, this exploration of households' attitudes and perceptions of the forest management intervention indicate, however, that 100% of households would enter into the proposed CFM arrangements. With clear concerns that the current levels of forest use cannot continue into the future, households may well be willing to incur a cost to move to a common property regime. This may also reflect a history of traditionally managed forests in the Bale Mountains that was reported to work more effectively at controlling forest product harvesting than central or regional government has been. It may also indicate that non-market benefits to the household are large. These non-market values could include the value that households place on the devolution of use rights, on the institutions and social capital established, or on empowerment provided under the new forest management regime. In both cases this would infer that OCs measures may overestimate the payment incentives required to generate the desired level of forest conservation.

It is also found, however, that instead of incurring costs of the intervention as Chapter 6 would predict, households are almost all expecting income gains as a result of CFM implementation. Although more research is required to disentangle the factors influencing perceived income changes and cooperative intention at the case study site, it can be seen that qualitative attitudinal data complements empirical cost estimates and can contribute towards intervention design and implementation. The high expectations for REDD+ mechanisms, by a number of forest stakeholders, have been noted elsewhere in the literature (Clements, 2010). Ethiopia's national REDD+ strategy clearly indicates that high expectations for the mechanism exist in-country (R-PP, 2011). REDD+ project development in the BME remains 'in the pipeline' despite four years of discussions. Eliciting households' perceived benefits of the proposed intervention provides an early indicator that expectations are high and improvements in information could ensure that these expectations are not unreasonable.

Chapter 9: Discussion and conclusions

9.1. Context

The potential for REDD+ to deliver climate mitigation benefits, to deliver finance to developing countries, and to contribute to meagre forest conservation budgets has resulted in a lot of excitement and discourse from academics, NGOs, government organisations, and civil society. Substantial finance has been pledged and transferred to build countries' readiness for REDD+. This flows despite uncertainty about the future financing mechanism for an international REDD+ mechanism. However, successful local-level forest management is required regardless of the financing mechanism and the scale at which REDD+ is implemented.

REDD+ can operate as a PES scheme, which can aid the delivery of direct, output based payments that are strongly addition to the business-as-usual baseline and conditional on continued service provision. One way to generate emission reductions from REDD+ is via CFM. Support for such approaches is growing in forested countries, but there has been little consideration of how REDD+ via CFM can be implemented on the ground if it is to operate as a PES scheme. The literature on PES, and limited studies of REDD+ via CFM, appear more 'PES-like'. This results in low conditionality of payments on environmental service delivery and uncertainties in whether carbon incentives overcome the OCs of land for ongoing provision of climate change mitigation services. The logic of collective action is also often ignored in PES, whereas it is central to incentivising CFM.

The proposed Bale REDD+ Project in the Bale Mountains of Ethiopia is been used to increase the understanding of how REDD+ can be implemented through CFM as a local-level PES scheme. This chapter first presents a discussion of the contributions to knowledge of this thesis, followed by policy recommendations and limitations to these findings.

9.2. Contribution to knowledge

Forest carbon accounting

The uncertainty in estimates of forest carbon stocks are understudied despite their challenge to the environmental effectiveness of a REDD+ mechanism. If estimated emission reductions result from a choice of methodology rather than changes in actual carbon stock, the integrity of REDD+ could be called into question. This thesis contributes to forest carbon accounting literature by illustrating that forest carbon stocks in the BME are insufficiently captured in biome averaged data, often applied where data on key forest variables and parameters, resources or capacity is scarce (Brown et al., 1989, Smith and Heath, 2001, Andersson et al., 2009). Such secondary data would underestimate forest carbon density in the BME by as much as 63% in the moist forest and 58% in the dry forest area.

It is recognised that numerous advances in GIS technology are being made to improve forest area estimates, and the discourse on how to establish the BAU baseline of deforestation from which avoided deforestation is assessed is active (Achard et al., 2004, DeFries et al., 2007, Olander et al., 2008). Forested countries are also undertaking reinvigorated forest inventory and accounting. The thesis demonstrates that in the case of the Bale Mountains REDD+ project, the financial implications of using secondary instead of primary data on forest carbon stocks could lead to a two-fold difference in emission reductions; equating to close to US$39 million over the 20-year project lifespan.

More accurate accounting of emissions reductions can ensure that REDD+ delivers real climate change mitigation benefits and countries are adequately rewarded for the emission reductions generated. This thesis also highlights, however, that even where more complex carbon accounting is undertaken, forest carbon stock estimates still contain high levels of uncertainty arising from the complexity in the ecosystems themselves and through sampling, measurement and estimation error. The costs of increasing the precision of forest carbon stock estimates could, therefore, be high. Despite this, this thesis finds clear economic incentives for the investment of resources to reduce uncertainty in forest carbon stock estimates.

The opportunity costs of REDD+ via CFM

The OCs of land are proposed to be one of the largest costs of REDD+ and can be used to anchor the level of payment needed to achieve the desired level of forest conservation, and so emission reductions (Pirard, 2008, Wertz-Kanounnikoff, 2008, Pagiola and Bosquet, 2009, White and Minang, 2011). The assessment of the OCs of REDD+ are often undertaken at a scale inappropriate for the design of on-the-ground incentives for local-level PES, however, and there has also been little attention paid to the OCs of REDD+ via CFM where there are community property rights and community groups legally become environmental service providers.

The OCs from a hectare of agricultural land in the survey year was estimated at US$417/ha. If low-impact forest products - including bamboo, climber, coffee and honey – offset the OCs of agricultural land, net OCs are estimated at US$375/ha. Not all forest, however, is likely to have ended up as agricultural land. The OCs of land from which high-impact forest products are harvested – including timber and fuelwood – is estimated at US$28/ha. If low-impact forest products can offset the OCs of high-impact forest products, the net OCs are estimated at US$ -12/ha. In assessing the local OCs of land for a proposed REDD+ via CFM project this thesis adds to the growing body of evidence that shows how incorporating cost information in conservation planning can aid interventions design (Polasky et al., 2001, Polasky et al., 2005, Naidoo et al., 2006, Naidoo and Iwamura, 2007, Carwardine et al., 2008).

While the OCs of agriculture show the clear economic incentive to clear forest, the headline OCs of forest products, obscure forest product use heterogeneity across the Bale Mountains Eco-Region. The income from forest products was found to be largely determined by survey location which dictated the availability of forest products. In the dry forest and woodland survey locations, forest coffee does not grow and incomes from high-impact forest products exceed that of low-impact forest products. In the moist forest where forest coffee and forest honey is available, income from low-impact forest products exceeded that from high-impact forest products. The high value of forest coffee is responsible for the finding that the net OCs of high-impact forest products, offset by low-impact forest products for the whole Bale REDD+ Project region appears negative. This implies that economic incentives do exist to conserve moist forest as opposed to fuelwood and timber harvest, but it is likely that other barriers relating to the current open access regime and lack of law enforcement that prevent the realisation of these values leading to continued forest decline. These research findings clearly demonstrate that a payment incentive differentiated by location would be more efficient than one based on the average OCs of forest products for the Bale REDD+ Project. This is further exemplified by the fact that only 50% of the surveyed households were found to gather low-impact forest products and therefore OCs of land would not be offset by such forest products.

In addition to location, this research identified that more educated households derive higher income from low-impact forest products and younger households derived higher incomes from high-impact forest products. These findings suggest that heterogeneity in OCs of land go beyond location. In the first instance, the Bale REDD+ Project implementers could capitalise on the impact of education through outreach and awareness programmes funded either through core Project funds that current arise through donor support, or through carbon revenues. Secondly, supplementary payment mechanisms could be designed in the implementation of the REDD+ via CFM to protect the welfare of younger households. This might be implemented following eroded traditional forest management systems, for example, that redistributed forest coffee beans to families unable to collect their own as a result of old age, physical disability or illness under the pre-existing *Gada* systems of traditional forest management (Wakijira et al., *in press*). Or, it could provide preferential access to alternative fuel sources and fuel-efficient stoves to such households.

These findings also bring attention to the fact that all households rely on the forest resource for fuelwood. High-impact forest products were mostly consumed within the home whereas low-impact forest products were largely sold by households. The Bale REDD+ Project intends to establish woodlots, distribute fuel-efficient stoves, and to introduce systems to make biomass briquettes to reduce the demand for fuelwood from the standing forest. These would initially be funded or subsidised through the donors of the overall BERSM Programme, but over time, could be supported with REDD+ revenues. Payments to households may well even be made indirectly through fuelwood allocation, rather than direct cash incentives.

Depending on how access to the woodlots is established under the Bale REDD+ Project, and on the success in uptake of more fuel-efficient stoves and biomass briquettes, the OCs of high-impact forest products could decline over time. Payments to overcome these OCs may therefore be considered a hurdle payment to a more sustainable path of use after which the payment is no longer necessary. On the other hand, if the demand for products such as fuelwood cannot be met by woodlots - the establishment of which has been delayed – then households might return to high-impact activities due to lack of alternatives, thus eroding the REDD+ mechanism. Of course, both alternatives will need to bear in mind rising demand for such forest products through population growth.

Other studies of REDD+ via CFM have proposed more 'PES-like' interventions (Nepstad et al., 2007, Peskett et al., 2008, Skutsch et al., 2011). The inability to attribute land-change in the status quo and so service provision to a single individual highlights the complexity of implementing PES under a common property regime. With common property rights to many forest products, households will have overlapping access and therefore costs and benefits on a given unit of forest land. PES incentive payments are, therefore, hard to link to a certain hectare of land enrolled or a given individual. Furthermore, given the lack of land-use change models for the Bale Mountains Eco-region, it cannot be predicted if land would be converted to agriculture or deforested through high-impact forest product harvesting. If all households were to receive payments for their potential OCs of land, therefore, it would impact on the effectiveness and additionality of emission reductions and payments would be made without emission reductions being generated. The conditionality and the efficiency that PES was initially proposed to deliver would be challenged (Ferraro and Kiss, 2002, Ferraro and Simpson, 2002). Similarly, the conditionality of the payment would be eroded and efficiency reduced. Possibly so much so that REDD+ no longer becomes a cost-effective climate change mitigation option.

CFM approaches to REDD+, therefore, can break the pure economic models required by PES in theory. This thesis suggests, however, that the attribution of the OCs REDD+, particularly of agricultural production, could be improved through benefit sharing mechanisms designed by the forest Community Based Organisations (CBOs) themselves. In this way, REDD+ via CFM could still be efficiently implemented with the conditionality and the efficiency that PES was initially proposed to deliver. It is noted, however, that this thesis goes some way in considering only one way to implement REDD+. By focussing on REDD+ as a PES scheme, it does not consider alternative policy options that include, for example, the imposition of private property rights, or improvements in central government control and law-enforcement of forest regulations, although these have so far proven ineffective in the BME. As Ethiopia is pursuing a national REDD+ strategy it is likely that in the longer-run a suite of measures will be employed to achieve emission reductions from forestry activities.

In the Bale Mountains Eco-Region, CFM could build on the traditional forest management practices than are reported to have eroded since the 20th Century. Village elders used to be responsible for both the day-to-day jurisdiction of forests, as well as the introduction of rules and norms of resource use (Wakijira et al., *in press*). Groups of village elders still exist at kebele level and are still called upon and respected for dispute settlement; all permissions to undertake the surveys in this thesis, for example, went through the village elders in addition to the local government authorities.

The implementation of REDD+ via CFM in the BME will require the establishment of institutions to enable cooperation to occur. It is so far planned that a subset of identifiable forest users will be formed as a CBO group which is envisaged to be no more than 30 households. The forest group will be allocated between 300 and 500 hectares and formalised in contracts signed between the CBO and local government forest agency that will also formalise their duties and responsibilities: including maintaining forest area, excluding those not within the group and regulation of use from those within the group. The local forest agency will undertake assessments of forest cover, and are expected to enforce sanctions for non-compliance to both CBO and non-CBO members. However, where rules are broken, individuals will first appear in front of village elders to be sanctioned and it is likely that only repeat offenders will be referred to the government authorities.

Given the strong history of traditional forest management within the Bale Mountains Eco-Region, it is possible to envisage the REDD+ via CFM system working. It is recognised, however, that new functions will be added to such institutions. For example, more stringent monitoring and reporting of emission reductions are likely to be required, and there may need to be greater oversight from the local government agencies to ensure leakage does not occur from one forest area to another and that emission reductions are permanent. The potential revenues from carbon may also require the consideration of the capacities of CBOs to manage finance, and the local government may act as an intermediary to also absorb some of the risk and liability of non-permanence of emission reductions. Additional support to develop such new functions will be necessary, but with confidence in appropriate benefit-sharing mechanisms at the community level and oversight from implementing agencies to avoid possible social risks such as the capture of benefits by village elders or powerful members of society and marginalisation of poorer social groups, local-level knowledge could allocate payments in line with costs.

Scenarios of the OCs of forest conservation over time

The Bale REDD+ Project is typical of conservation interventions that lack explicit goals and targets (Margules and Pressey, 2000). While the Bale REDD+ Project proposes the commercialisation of forest products, the promotion of agricultural intensification, for example, there are no quantitative estimates of impacts that can help understand how the OCs of land will change over time. The scenario modelling of the OCs of land over the 20-year REDD+ project period, using different assumptions, shows that small changes in the assumptions of productivity of land uses significantly impact on the annual OCs, cumulative OCs and the total OCs of the intervention. Although the scenario storylines differ as a result of paucity of data on productivity trends and degradation of land and forest, the scenarios show that the Bale REDD+ Project implementers should consider more carefully their conservation goals and targets.

The total OCs of agriculture as a result of the Bale REDD+ Project were estimated as the discounted revenues of agriculture, net of low-impact forest products, for the total area of forest conserved during the 20-year project. The three scenarios generated from small changes in assumptions, saw the OCs range from US$121 million in an optimistic scenario to US$441 million in a more pessimistic scenario. It is also clear from the scenarios analysis that while the Bale REDD+ Project implementer’s sphere of influence might extend to the proximate drivers of deforestation, such as agricultural expansion and wood extraction, they are likely to be less influential to the underlying drivers of deforestation such as population growth, wider market factors, demographic trends, and/or institutions.

The consideration of economic incentives for conservation behaviours and efforts to alter them have been established in many conservation interventions

(Brandon and Wells, 1992). However, quantifying these targets and assessing if they are feasible is rare. The scenario backcasting from zero total OCs illustrates that the prices of low-impact forest products would need to be multiplied by 11 times their current value to be sufficient to generate a situation where forest conservation becomes an economically viable land use option. It seems unrealistic, however, to expect the current price of bamboo, climber, coffee, and honey to reach such highs. This appears unlikely even given the high demand for specialist coffee in international markets (Wren, 2007); between 1993 and 2013 coffee Arabica has risen in value from just over US$1.50 per kg to over US$4 per kg, it doesn’t match the required price rise in the scenario modelling in the Bale Mountains Eco-Region (World Bank, 2013). Given also earlier findings that show household incomes from forest products are highly spatially explicit, such a price increase may not be feasible across the BME.

The scenario analysis also illustrates that REDD+ revenues may be insufficient to overcome the OCs of agricultural land at the case study site. Of the market mechanisms that could be used in conservation, carbon trading is thought to have the greatest potential to capture positive externalities to the degree required to make forest conservation economically viable (Richards, 1999). Carbon is also often found to be the largest of the non-marketed environmental service values of forests (Pearce, 1997). However, the scenarios illustrate that the OCs of agriculture can be prohibitively high if carbon prices are relatively low and efforts to increase agricultural productivity and add value to forest products achieve little success. This depends on the scenario parameters, of course, but may challenges the speed and optimism with which many conservation practitioners are adopting REDD+ as a tool to finance forest conservation, including in Ethiopia.

Overall, the findings demonstrate that the application of scenarios can be a useful tool in conservation planning. Scenario modelling is not highly data intensive and allows conservation implementers to break from established patterns of thinking and critically evaluate their plans. The scenario modelling undertaken does not provide a single answer for the OCs of REDD+ over time, but highlights the need for further discussion of the intervention strategy and the potential impact of its objectives.

Attitudes and intention to cooperate in REDD+ via CFM

This thesis also demonstrates that implementing REDD+ via CFM can defy the purely economic logic of PES. All surveyed households stated a willingness to take part in the proposed REDD+ via CFM intervention accepting the forest use restrictions that this places upon them; modelled here as limitations on expansion of agricultural land, high-impact forest product harvesting, and duties and responsibilities resulting from the cooperative forest management agreement. The thesis, therefore demonstrates how an understanding of local environmental attitudes, perceptions of forest management and cooperative intention could aid the identification of household incentives to provide collective action beyond the analysis of the OCs of a hectare of forest conservation.

Households in the BME were very aware of the declining forest area and condition, as well as both the proximate and underlying drivers of deforestation at the case study site. Their concerns about the resource base were held alongside a general disenchantment regarding current forest management and, overall, a preference for the traditional forest management arrangements that had existed previously in the region. The complete willingness of survey respondents to enter into the Bale REDD+ Project might be interpreted as the perceived benefits of cooperation overcoming any perceived OC of foregone land use and transaction costs due to time spent in meetings. This may include, for example, the benefits of non-market values of water-shed protection, cultural forest values, or recreational value, in addition to any value households place on empowerment, the use rights to the forest, or the value of being part of a cooperative.

While this research was unable to further deconstruct the perceived benefits into their component parts, it is clear that households perceived that their incomes would increase under the Bale REDD+ Project intervention. This increase was attributed to higher production of low-impact forest products and as a result of the ability to trade as a cooperative group, which may be reasonable. However, perceptions of future incomes were also high due to expectations of higher crop yields due to improved rainfall and local climate regime. Additional uncertainty was added by the potential for carbon payments, for which no anchor value was provided. Despite these limitations, this finding highlights a possible danger of households' high income expectations of the Bale REDD+ Project intervention.

The need to manage expectations of the Bale REDD+ Project is particularly acute given the uncertain nature of if, and how, carbon finance will reach the community. The lack of an emission reductions purchase agreement and adequate assessment of benefit-sharing mechanisms, it remains unclear how much of carbon revenues will be absorbed by central or regional government, and how much will reach households. This is exacerbated by uncertainty in carbon rights and experience of handling carbon revenues from forestry in Ethiopia. This will have impacts on the longevity of an intervention, and the permanence of emission reductions as it may be less likely that household expectations are met by the proposed intervention if less than the modelled carbon revenues are available to communities. Furthermore, discussions on who bears the liability for the non-permanence of emission reductions, and how a risk buffer would be managed must be worked out by the Bale REDD+ Project. Although in Chapter 5 it is estimated that a 65% buffer for these risks is set aside, the implications for the CBOs if emission reductions are not generated for either natural or anthropogenic reasons must be further discussed.

This ex-ante study of attitudes and perceptions towards a conservation intervention is rare. It demonstrates the potential importance of the logic of collective action and how they might impact on the economic incentives required for PES to operate in theory. It also demonstrates that ex-ante studies can inform conservation planning and may well lead to better implementation and longevity of the Project intervention in the BME in the longer-run.

9.3. Policy recommendations

With strong pro-conservation attitudes, the barriers to households undertaking forest conservation measures in the BME appears to be their reliance on forest products for subsistence use, particularly on fuelwood, and the *de facto* open access regime. The additionality of emission reductions in the BME can, therefore, be justified on the basis of clear policy barriers to more sustainable forest management if the status-quo was to continue. The proposed implementation of REDD+ via CFM can be considered a realistic intervention given that resource appropriators are dependent on the forest resource for a large portion of their livelihoods and share a common understanding of how resource use affects others. The details of implementation of the intervention, however, are vague and the findings of this thesis suggest three major policy recommendations that will aid in the implementation of REDD+ via CFM at the case study site.

Recommendation 1: Undertake a detailed forest inventory

The economic implications of forest carbon accounting methods were shown to be substantial for the BME REDD+ project. The investment of resources in a detailed forest inventory is therefore justified in the BME. This will increase the environmental integrity of a REDD+ project but will also improve investor confidence in the emission reductions generated. The development of land-use and land use change models will also help to understand where deforestation is taking place and to what forest land is converted too; agriculture, pasture or for timber and fuelwood that can better model carbon losses and, therefore, emission reductions, under the Bale REDD+ Project.

Recommendation 2: Develop quantitative objectives for the Bale REDD+ Project and repeat scenario analysis

The scenario analysis highlights that the qualitative goals of the Bale REDD+ Project intervention and the lack of data on trends in land-use change result in uncertainty over whether REDD+ revenues will overcome the OCs of forest conservation. The Project implementers should work to quantify better their objectives to intensify agriculture, efforts at which have persistently failed across Ethiopia. Similarly, while adding value to low-impact forest products is likely to be more within the influence of Project implementers, considerations of the scale at which this can be achieved both by product and by location is necessary.

In the BME, there is currently an adaptive management approach to the Project intervention. For the Forest Management Agreements signed with communities, two or three years may be required before changes in the forest condition can really be observed. The implementers expect ex-post adjustments of policy to be required. Repeating the scenario modelling exercise as the intervention progresses will allow the review of the intervention in accordance with this adaptive management approach. These scenarios should also be built further through participatory approaches to incorporate more diverse views from a variety of forest stakeholders. Breaking the mould and also undertaking further ex-ante assessments of policy objectives at the case study site could allow proactive steps can be taken to influence OCs of forest conservation and to predict and plan for future eventualities.

Recommendation 3: Build and use OC information in REDD+ via CFM incentive design

Forests are an important part of household's livelihood strategies in the BME. The Project intervention should be working to both minimise and overcome the OCs of forest conservation through the options available to them and their sphere of influence. The OCs of high-impact forest products are experienced by all households at the case study site as modelled here. The conservation implementers' plans to develop woodlots and to expand the distribution of more fuel-efficient stoves, may mean these are temporary OCs of REDD+. However, with 99% of households gathering fuelwood and woodlot establishment delayed, plans to meet household's biomass needs should be advanced rapidly if leakage of emission reductions is to be prevented and avoided deforestation targets are to be met.

Considerations on how REDD+ revenue is to be shared also need to be made with consideration of the project's ability to attribute costs at fine-scales. The Bale REDD+ Project implementers should consider the promotion of CBO and kebele-level engagement in benefit sharing mechanism design. Regulatory oversight, however, will be necessary to assess possible risks of elite capture and for equity and conflict resolution. Such oversight was requested by households in survey findings.

Bale REDD+ Project implementers should also consider that where payments are made, differentiation by forest type can improve the efficiency of payments. With lower OCs of high-impact forest products and lower net OCs of agriculture, the redistribution of REDD+ revenues from moist forest areas to dry forest or woodland areas may also be possible. With younger households found to derive higher income from high-impact forest products, and anecdotal evidence that younger households have limited access to agricultural land, particular attention to the costs incurred by this demographic group should be made in the intervention design. Bale REDD+ Project implementers could consider how the intermediary organisation that will receive carbon finance, might operate and make such allocation and redistribution decisions. In doing so, it is clear that the expectations that households hold for the proposed Bale REDD+ Project intervention are mediated and managed appropriately.

A caveat of these recommendations is that the use of estimates of the OCs of in REDD+ via CFM design must be complemented with an understanding of the attitudes and perceptions of communities local to the intervention. The logic of collective action is commonly omitted in designing PES scheme incentives. However, the urgency of the situation recognised by the survey respondents and faith in restoration of traditional community forest management structures, suggest that they will play a role in incentivising REDD+ via CFM in the BME.

These research findings also have wider implications for REDD+ via CFM:

- At present there is no standardised method to assess or communicate uncertainty in emission reductions accounting and the principle of conservativeness remains the dominant approach (Mollicone et al., 2007, Grassi et al., 2008). Although this enables REDD+ to deliver real and permanent emission reductions, adopting the principle of conservativeness with zero uncertainty leaves policy makers without a confidence interval. The implication could be lost opportunities for climate change mitigation.

The uncertainty in forest carbon accounting should therefore be quantified, reduced and communicated more appropriately. Tools such as sensitivity analysis could be employed to identify which components impact the most on total uncertainty which can then be prioritised for further research. Decision-makers need to initiate discourse on the level of uncertainty that is acceptable for a performance-based mechanism such as REDD+. With many national forest inventories in developing countries non-comprehensive and limited resources for new field measurements, a portion of REDD+ readiness finance should be earmarked for improving national forest inventories so that emission reductions are credible.

- The implications of the apparent shift towards 'PES-like' implementation of REDD+ via CFM needs to be considered in view of the need for REDD+ to deliver real, permanent, and verifiable emission reductions that are additional to the status-quo and conditional on service delivery. Given the complexities of REDD+ via CFM with regards to attribution of costs, more community-level benefit-sharing mechanisms might be explored in both theory and through demonstration and pilot REDD+ activities.

A shift away from assessments of the OCs of agricultural land, towards consideration of other OCs of land, such as subsistence forest products, is also necessary. There are growing calls for REDD+ to 'do no harm' to forest communities (CCBA, 2008, Griffiths, 2009). These are embodied in the UNFCCC text safeguards that are intended to avoid negative impacts on the poor, on biodiversity, on food security and national sovereignty. This can be interpreted as going beyond ensuring that households' costs of forest conservation are to be overcome, but more consideration of how such safeguards can be operationalised is needed.

- Moving to common property regimes, households may be willing to incur costs of a property regime change. The logic of collective action implies that estimates of the OC of land can overestimate the payment incentives required to generate the desired level of forest conservation. Opportunities for climate change mitigation may be lost if schemes are not implemented due to apparently prohibitive OCs of REDD+. More research is required on how collective action influences households' incentives to cooperate in REDD+ via CFM.

9.4. Limitations and recommendations for future research

This research has emphasised the uncertainty that exists in forest carbon stock accounting for REDD+ projects. The focus of the discussion is on uncertainty from sampling error and in market variables. It is acknowledged, however, that there are other sources of uncertainty arising in carbon stock estimation, including measurement and estimation error. The complex forest carbon stock assessment undertaken here relies on allometric equations that arise from Asian and Latin American tree datasets. Destructive sampling to generate allometric equations specific to Ethiopia or East Africa could be generated through further research. Furthermore, the primary data estimate of forest carbon stock omits other carbon pools which will also contain uncertainty in their measurement and need to be further investigated.

In estimating the potential emission reductions and revenues of a REDD+ project in the BME, a fixed historical deforestation baseline of 4% is applied and it is assumed that the conservation objective meets its stated avoided deforestation targets. Further research into developing a BAU baseline of deforestation could provide an improved estimate for emission reductions and, therefore, REDD+ revenues. The emission reductions accounting also ignores emission reductions that may be generated from reduced degradation, forest conservation, sustainable management of forests and the enhancement of forest carbon stocks; all included within the definition of REDD+. Avoided forest degradation, in particular, may be relevant for CFM (e.g. Chhatre and Agrawal, 2009). Further research into the causes of degradation and consequences of CFM on forest quality could lead to better emission reductions accounting in the proposed BME REDD+ intervention.

As a result of the logistical and permission constraints encountered during data collection, a major limitation of this research is that the number of survey locations was limited. With three survey locations with three different forest types the effects due to forest and the effects due to village characteristics cannot be separated. Expanding this research into other survey locations could reinforce these findings and provide a better understanding of what is driving the differences observed between survey locations. Future household surveys could also increase the sample size, and address the seasonality of forest product harvesting and of market prices. The omission of female headed households in this study is also an important area where further investigation should be pursued. Cultural barriers in the BME prevented enumerators approaching females within their households. Per capita income of female headed households in Ethiopia can be as low as 60% of the per capita income of male-headed households (Yemiru et al., 2010). This indicates that possible welfare risks to female headed households as a result of REDD+ via CFM in the BME need to be further investigated.

This thesis is predominantly a financial analysis of REDD+ via CFM with a focus on the environmental service of climate change mitigation. A number of other non-market values that households derive from the forest ecosystem are not assessed. Given the focus in the international REDD+ debate on the need to secure such co-benefits, further investigation on how these impact on costs is necessary. Although these values are complex to calculate at a household level and ecosystem services can be complex to quantify, methods are available that could investigate further how non-market benefits impact on the incentives required to generated the desired level of service provision for REDD+ (e.g. OECD, 2002, Pagiola et al., 2005b).

The financial analysis undertaken here was also not able to consider the impact of the intervention on household's income from livestock. Grazing is currently allowed and occurs both in and outside of the forest. Under the proposed Project intervention, this may change through measures such as rotational grazing agreements which prevent forest re-growth being destroyed by livestock. Research in the BME is underway to assess the deforestation impact of livestock. The development of this research could include the OCs of the Project intervention resulting from grazing land restrictions.

In order to model OCs of REDD+ in the BME, land uses were simplified into agriculture, low-impact forest production and high-impact forest production. More research could develop these land categories to explore how OCs may be affected by land characteristics such as soil fertility, altitude and/or land use practices. The assumptions of the non-rival nature of low-impact forest products with the intervention and the rival nature high-impact forest products with the intervention require more attention. There is anecdotal evidence that honey yields are declining in the highland regions, with flowering species suffering from degradation. Forest coffee although a natural plant is encouraged through thinning of the forest canopy and harvest of forest coffee may be excludable. A better consideration of the sustainability and excludability of these low-impact forest products, with a view to understanding if household production can feasibly be extended outside of the current production boundary, is necessary.

Similarly, the harvest of timber and fuelwood – both high-impact forest products – may be allowed within sustainable limits; defined as a level of extraction below the mean annual increment of biomass. An ex-ante study of REDD+ via CFM in the BME, the by-laws of CBOs that will define the use and management of forests are yet to be established. Designed through participatory processes, early experience in the BME shows that CBOs are limiting the harvest of fuelwood through restrictions on the number of days that households have access to the forest. As by-laws emerge and measures to reduce and remove the fuelwood and timber demands on natural forest areas are put in place, the estimate of the OCs of high-impact forest products could be adjusted appropriately.

The transaction costs that households may incur as a result of the implementation will also become more obvious as REDD+ via CFM is rolled out across the BME. Such information as it emerges could also be included in the analysis of the impacts of REDD+ on households. It is noted in Chapter 4 that transaction costs will also be incurred by household participating in the scheme including negotiation, monitoring and enforcement costs and that these costs may not be evenly distributed over socio-demographic groups (Meshack et al., 2006, Adhikari and Lovett, 2006). In the BME, these transaction costs are likely to entail weekly meetings and the patrolling of forest in critical harvest times. While emerging by-laws state that all members should contribute to patrolling, the committee members appear to be taking on this burden in the few established groups. The time burdens that this places on households may well influence their payoffs and therefore cooperation in the intervention. Analysis of these costs are important, as while this thesis assumes the OCs of land to be the highest costs incurred by households, this assumption has not been tested.

An improvement to this thesis would be to estimate the shadow prices of products for both subsistence and sale on markets, instead of applying market prices that are likely to have led to the overestimation of OCs of land. Rural households will operate within market imperfections and constraints and will have differing transaction costs to access and participate in markets (Sadoulet and de Janvry, 1995). Utilisation of non-separable household models where production and consumption behaviour is linked, would improve this research. Investing resources in estimating the shadow prices of products for each household would also give a better estimate of OCs of land. This is particularly true as many products were consumed at home and shadow prices are generally lower than market prices.

The scenario analysis required further assumptions to be made in order to model the OCs of REDD+ over time. This included the assumptions that: all deforestation results from conversion to agricultural land use; once transformed land remains under that use; and, that households will generate income on this land to the same level of a household's existing income from that use. These assumptions resulted from a lack of data on deforestation dynamics at the case study site. NGOs in the BME are using GIS imagery to analyse land use and trends in land-use change. As more data become available on the proportion of deforestation from each land use, models of deforestation risk could be developed. A better understanding of which households are expanding agricultural land, where, and on what type of land this occurs could explore whether newly converted land is more marginal, or if one-off benefits and costs of land conversion exist. Scenarios could be improved through the inclusion of price feedbacks from the changing availability of land (e.g. Busch et al., 2009, Fisher et al., 2011). Data on population growth and its impact on land-use change may also focus attention on emerging families and their access to CFM that are largely unacknowledged in the intervention as it stands.

While Chapter 8 highlights the importance of an understanding of the attitudes and perceptions of local stakeholders, there are clear limitations to the quantitative measure of behavioural intention, the voluntary contribution, and in disentangling the willingness of households to take part in the proposed intervention. Further research could explore this further, assessing in more detail how and why households believe their incomes will increase under the intervention. Factors such as social capital at survey locations including: relations of trust; reciprocity and exchanges; common rules, norms and sanctions; and, connectedness in networks and groups can also be explored in much greater detail as social capital is known to impact on cooperation (Pretty, 2003).

This thesis does not consider in various mechanism by which revenues could reach households due to time restrictions on material to be covered. Throughout, it is assumed that the revenues will be able to reach households after the costs of implementation are met. With no Emission Reductions Purchase Agreement for the BME REDD+ project, no agreements have been made on the benefit-sharing mechanism or the amount of revenues that will be absorbed by other stakeholders, such as central or regional government. Furthermore, the impact that risk and time preferences will play in the design of PES incentives and contract design are not considered here (Ferraro, 2008, Knoke et al., 2011). While the scenario analysis indicates that forest conservation could become an economically rational land use option, this could only hold if a household can take a 20-year view. The subsistence households of the BME may be unable or unwilling to take such a long term view. Time and risk preferences factors of households in the BME warrant further research as the REDD+ project progresses.

Finally, it is acknowledged that this thesis considers only forest stakeholders that live locally to the forest at the case-study site. These are considered here to be the most affected party from REDD+ development in the BME. However, REDD+ planning and implementation will need due consideration of other forest stakeholders including the private sector (e.g. small-scale wood enterprises and non-wood forest enterprises), urban dwellers that are dependent on forest products brought to markets, and local and central governments. A variety of forest stakeholders will incur OCs of REDD+ and should be considered in REDD+ via CFM intervention design.

9.5. Conclusion

This thesis is a rare ex-ante analysis of the incentives to deliver REDD+ via CFM and exploration of how such a mechanism might be implemented on-the-ground. I demonstrate that forest carbon stock accounting uncertainties are being overlooked, but have substantial implications for the environmental integrity of a REDD+ mechanism. I demonstrate that the focus on the OCs of agriculture in studies of REDD+ omit important OCs of subsistence forest products. This research highlights a drift towards more PES-like implementation of REDD+ via CFM that at first glance can appear better fitting with local realities. However, I suggest that the efficiency and conditionality of

PES can be maintained through placing more trust in community-level institutions to attribute costs and engage in incentive design and benefit-sharing. The application of scenarios to model the OCs of forest conservation over time illustrates that REDD+ revenues may not be sufficient to overcome the OCs of forest conservation if carbon prices are low and efforts to intensify agriculture and add value to forest products are limited in their success. Proactive measures can be undertaken by the Bale REDD+ Project implementers, however, towards overcoming and reduce these OCs. Finally, I show that generating an understanding of stakeholders' attitudes towards forest management and the use of the resource base will allow better consideration of socio-cultural factors for cooperation in conservation that will go beyond payment incentives that PES theory highlights. These themselves could help tip the balance for forest conservation even if the OCs of land appear prohibitive.

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Appendix 1: Household survey

[Record: enumerator initials, date and location]

[Read out] **"My name is** *(name of enumerator).* **I am here on behalf of Charlene Watson who is collecting information for her PhD studies in London, England.**

We are collecting information on the income that forests and cultivated land provide to your household and other households in the Bale area. This income includes products sold and consumed at home. This information will allow a better understanding of the land use in your kebele and the role that forests and forest products have in people's lives. The information and comments you provide in this questionnaire will also contribute to investigations into the opportunities and possibilities for future forest management in the Bale area. In particular, a scheme for communities to manage forest called participatory forest management which is supported by the government and by Farm Africa-SOS Sahel.

If you chose to take part in this questionnaire, your answers will not be shared with other members of the community or the authorities. Would you like to continue with the questions, it will take about 60 minutes?"

[If no, move on to the next household]

[If yes, write down time interview started]

Part 1 – Your thoughts about the forest management around your kebele

1. What did you think about the management of the forests that your household used in the past under the Derg regime (before 1991)?

   …………………………………………………………………………………………

   ……………………………………………………………………….………………

2. What do you think about the current management of the forests that your household uses?

   …………………………………………………………………………………………

   ……………………………………………………………………….………………

3.
   a. Do you think that the current level of forest use by your household and others in this kebele can continue into the future?

      yes/ no/ don't know

   b. Why?

   …………………………………………………………………………………………

   ……………………………………………………………………….………………

4. How might forest management change if communities were given legal rights to use the forest and forest products?

   …………………………………………………………………………………………

   ……………………………………………………………………….………………

5. We will read a statement out to you and would like you to decide if you agree, don't know, or disagree.
   a. If an area of forest is not being used by people it is not a problem if the forest area gets smaller.
   b. Even if my household does not use a part of the forest, I would participate in this forests management.
   c. I do not think about my household's use of forest in the future, it is enough to think only about my household's use of the forest now.
   d. There is a responsibility for me to manage the forest well now so that my children and future generations can benefit from the forest in the future.

e. One management objective for forests in the Bale Mountains should be to support wildlife that lives there.
f. Forests should be managed only if this does not negatively affect people's livelihoods.
g. People have more important things to worry about than good management to maintain the forest.
h. People only cut down the forest because they have no other ways of supplementing their livelihoods.

Part 2 – Forest use and land cultivated by your household

The next few questions help us compare the income that you get from forests and cultivated land. This includes forest products and crops that your household collected, used at home and sold. If you cannot recall for the last year (or 12 months), please give answers for a period of time that you can remember.

6. Which of the following products did your household collect?

Product	Amount collected last year (units)	Maximum time travelled from your house (hours)	Amount sold (units)
Timber			
Fuelwood			
Bamboo			
Honey			
Coffee			
Other.......			

7.

a. On average, is the time you travel from your house to collect these forest products *more/ the same/ less* than the year before this?

b. On average, how much more or less than the year before this? ....*hours*

c. Why do you think the time to collect changing?

…………………………………………………………………………………………

……………………………………………………………………….………………

d. Do you think the time you travel to collect forest products will change next year? *more/ the same / less*

8.

a. Please list the crops that your household harvested last year (or 12 months).

Crop type	Yield from first crop (Qt)	Yield from second crop (Qt)	Total amount sold each year (Qt)

b. Estimate your costs of inputs to crop production in one year?

Item	Cost (Ethiopian Birr)
Fertiliser	
Seed	
Equipment	
Paid Labour	
Other	

9.

a. What area of land did your household cultivate last year? ……….*hectares*

b. Is this *more/ the same/ less* than the year before this?

c. How much more or less the year before this? .... *hectares*

d. For what reason?

…………………………………………………………………………………………

……………………………………………………………………...…………

e. Will you cultivate *more/ the same/ less* land next year?

10. I would like you to take these 100 beans and to share them between the sources of income that your household receives. For example, if you receive no income from that source put zero beans on it. If you receive half of your income from crops, put 50 of your beans on it.

Forest Products	Crops	Livestock	Trading	Paid Labour	Money from Overseas	Other

Part 3 – A community based Participatory Forest Management system

Recent research in the Bale Mountains has shown that the forest area is decreasing very quickly. Forest is being cut down to make space for settlements, agriculture, grazing of livestock and to use the timber and fuelwood. If nothing is done the forest area will continue to get smaller. This means the benefits that forests provide to households over time will decrease.

One way to maintain the benefits that forests in Bale provide is to ensure that the forest is used sustainably. This means maintaining the forest area and quality through community-based forest management and by diversifying the ways that households can earn an income. This is similar to what is being proposed by FARM Africa-SOS Sahel. This has been carried out successfully in other areas of Ethiopia and it is called Participatory Forest Management.

Under Participatory Forest Management your household would become a member of a community cooperative. It would be an honest, credible and reliable organisation with committees, rules and responsibilities. The cooperative would have a legal agreement with the Oromia Forest and Wildlife Enterprise. It gives your cooperative the rights to use the forest and to use and sell forest products. Under this agreement, your cooperative is responsible to manage the forest according to a forest management plan agreed with the Oromia Regional State Forest Enterprise.

Under Participatory Forest Management, in order to meet household needs for fuelwood and timber your cooperative will have to sustainably manage the harvest from the forest and utilise woodlots. Livestock grazing will have to be managed so that it does not degrade the forest. The collection of forest products such as honey, climbers, coffee, and medicinal plants would not be affected as the tree cover is maintained and forest destruction is minimised.

It is possible that community cooperatives might also be able to receive payments if they are able to protect the forest from being cut down and maintain the area for 20 years. This is because trees store carbon and when they are cut down this is released into the atmosphere and the global climate changes. Governments, companies and households in places like England, are willing-to-pay to keep the carbon in the trees to slow down climate change. It is possible that payments could be made for carbon each year, only where the forest area has been maintained. The cooperative could then choose what the money is spent on. A community group that cut down forest would not receive any payment for carbon.

11. Do you understand <u>Participatory Forest Management</u> that was explained to you? *yes, all of it/ some of it/ no, none of it*

    [Explain again if they understand none of it]

12. Do you understand why your cooperative might be paid money to keep carbon in the trees? *yes, all of it/ some of it/ no, none of it*

    [Explain again if they understand none of it]

13. Do you have any comments about the forest management just described to you, its advantages and disadvantages to your household and community if it were to be organised within your kebele?

    ………………………………………………………………………………………

    …………………………………………………………..……………………

14. Do you think the <u>Participatory Forest Management</u> will affect some people or social groups in your kebele more than others? In what way and why?

    ………………………………………………………………………………………

    …………………………………………………………..……………………

15. Do you think that the households in your kebele would join a <u>Participatory Forest Management</u> Cooperative? *yes/ don't know/ no*

16. Would your household be willing to join a <u>Participatory Forest Management</u> cooperative (remember that this is not a binding agreement to join)?

    yes/ don't know/ no

17. How certain are you that other members of a cooperative would follow the rules to maintain the forest area under <u>Participatory Forest Management</u>? *certain/ don't know/ uncertain*

18. If your cooperative received payments from carbon, how should the money be used?

……………………………………………………………………………………………

…………………………………………………………………..……………………

19.

a. Under the rules of the Participatory Forest Management described to you do you think your household would be *better off / just as well off as you are now / worse off*?

b. Why?

……………………………………………………………………………………………

…………………………………………………………………..……………………

20. Do you think that the rules of the Participatory Forest Management described to you would lead to a change in the income that your household gets from forest products next year (this includes products used at home as well as sold)? *increase/decrease/no change*

[If they say 'no', move to section 4]

[If they say 'yes' or 'don't know' continue with this section]

21. If Participatory Forest Management were to be carried out in your kebele, what change in amount of income would your household experience if it had to follow the rules of Participatory Forest Management? Before you answer, please think carefully about the impact of the scheme for your household.

……………………….. *Ethiopian Birr*

22. If the respondent was not willing to give an amount of money, why was this?

……………………………………………………………………………………………

…………………………………………………………………..……………………

23.

a. Under Participatory Forest Management the forests will be sustainably managed so that they will provide benefits and income into the future. These benefits include climate regulation and the protection of the watershed. It also means that forests will be available for future generations to use. To secure these benefits, would you be willing to give up some of your yearly income to your Participatory Forest Management Cooperative so that they could better manage the forest?

   yes/ no

b. If yes, how much money each year would you be willing to give to the cooperative so that forest use will continue into the future? ....… *Ethiopian Birr*

24. If the respondent was not willing to give an amount of money, why was this?

    ……………………………………………………………………………………

    ………………………………………………………………………..……………

25. You stated that the income your household gets from forests next year might change under Participatory Forest Management. Thinking five years into the future, how do you think that Participatory Forest Management might affect this income?

    ……………………………………………………………………………………

    ………………………………………………………………………..……………

[Read out] **"Please remember at this point that carbon payments are not definitely going to be made to cooperatives under Participatory Forest Management and if it was, it may not be organised as described above."**

Part 4 – Household Information

This is the last few questions which are about the people in your household. Please remember, information will not be shared with the authorities.

26.

a. Are you the head of the household? *yes/ no*

b. What is your level of education? .... *grade*

27. How many years has your household lived in this kebele? .... *years*

28.

a. Is this household polygamous? *yes/ no*

b. When you answered questions on forest products and crops, how many houses did you report for? ….... *houses*

c. How many people live in this household? .... *people*

[Read out] **"Thank you for giving your time to complete this questionnaire. Your answers will help use compare the value that forests and cultivated land provide to the households in the Bale area. They will also contribute to investigations into the opportunities and possibilities for future forest management in the Bale area. Charlene will report the overall results of this questionnaire when she next returns to the Bale Mountains".**

[Write down time of completion]

Appendix 2: Market price survey

Product	Unit	Average price across six market survey sites (ETB/unit)
Bamboo	Donkey	49
Climber	Donkey	13
Fence	1 piece	9
Firewood	Donkey	26
Grass	Sack	8
Timber	1 piece	40
Honey	Kg	31
Oils	Kg	21
Sorghum	Quintile	341
Barley	Quintile	284
Bean	Quintile	526
Peas	Quintile	724
Maize	Quintile	379
Teff	Quintile	632
Wheat	Quintile	444
Lentil	kg	15
Oats	Quintile	304
Onion	kg	6
Spring Onion	Quintile	180
Potato	Quintile	176
Garlic (White onion)	Quintile	617
Cabbage (round)	Quintile	108

Product	Unit	Average price across six market survey sites (ETB/unit)
Carrot	Quintile	158
Sweet Potato	Quintile	198
Mango	1 piece	1
Avocado	1 piece	1
Banana	kg	7
Beetroot	Quintile	129
Pepper	kg	35
Chat	bundle	23
Coffee	kg	30
Pineapple	kg	7
Butter	kg	77
Ginger	kg	5
Absuda (black cumin)	Quintile	2454
Abish	Quintile	1475
Dinblata	Quintile	525
Shimbura	Quintile	533
Salt	kg	3
Gayyo	Quintile	435
Lemon	1 piece	1
Green pepper	kg	15
Orange	1 piece	2
Dog tooth	Quintile	422