Realising REDD in Africa: Risk, feasibility and supporting policy

Tony Knowles

Promotnrs:

Prof. Albert S. van Jaarsveld

Dr. Robert J. Scholes

Dr. Allan Ellis

Dissertation presented for the Degree of Doctor of Philosophy at the University of Stellenbosch

Cdbd l adq 1/01

DECLARATION

By submitting this dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the owner of the copyright thereof (unless to the extent explicitly otherwise stated) and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

Signature: .... Date: 25-10-2011.

This dissertation is dedicated to my parents, Ron and Pauline Knowles, and all those who sat around the dinner table at St. Frusquin Street on Sunday afternoons

Abstract

Responding appropriately to anthropogenic climate change presents a considerable challenge to humankind. Projected changes in climate are anticipated to affect the world's natural systems, human health and economies in many ways. Consequently, there is an urgent need to implement climate change mitigation and adaptation measures that are appropriate and efficient.

This dissertation focuses on aspects of risk and feasibility associated with land use based climate change mitigation. First, it reviews policy, implementation and incentive issues that are key to promoting permanence and reducing the risk of leakage associated with reducing emissions from deforestation and forest deforestation (REDD[1]) in sub-Saharan Africa. Secondly, it assesses the transaction costs associated with the implementation of avoided deforestation and reforestation activities and their effect on the financial feasibility of ventures located in woodland and rangeland systems. Thirdly, it explores the potential impact of biophysical risk factors (such as fire) on the outcome of REDD activities in two chapters. The first risk chapter introduces the notion of biophysical risk and reviews the risk of fire to REDD activities located in important African vegetation types. The second chapter on risk uses the Century Ecosystem Program and published climate projection data to assess the effect of projected changes in temperature, rainfall and atmospheric carbon dioxide on the outcome of REDD activities.

The results indicate that, among the biophysical risk variables assessed, fire may not present a major risk to REDD activities located in African woodland, savanna and grassland systems. In contrast, fire may present a significant risk in moist forests where unprecedented dry periods may allow fire to occur in a system where it has previously been absent. The analysis of the affect of climate change found that changes in climate are generally predicted to lead to an increase in carbon stocks and

[1] Regarding appropriate terminology, the acronym 'REDD' stands for Reducing Emissions from Deforestation and Degradation. More recently, the acronym 'REDD+' has emerged which stands for avoided deforestation and degradation activities as well as forest restoration, rehabilitation, sustainable management, afforestation and reforestation. Although a formal definition of REDD has not been accepted into the UNFCCC text as yet, it is generally accepted to denote all forest, woodland and rangeland activities that lead to a net reduction in the concentration of atmospheric GHGs. The acronym REDD is therefore used in this dissertation unless a specific chapter, for example Chapter 4 that focuses on the financial feasibility of activities, requires a clear, separate consideration of REDD and reforestation activities.

sequestration rates for the vegetation types assessed. Exceptions do occur, such as the modeled effect on nutrient-rich savannas, which require further investigation.

The analysis of transaction costs associated with REDD activities illustrated that such costs may inhibit the feasibility of smaller-scale activities, especially in ecosystems outside of moist forests with relatively low carbon stocks and associated revenues. Whereas the proposed creation of national-scale capacity may reduce some transaction costs to a certain extent, there is a clear need to better understand the true cost of REDD activities.

In terms of required supporting policy and implementation capacity, it is noted 1) that multi-criteria land use planning is particularly important in reducing permanence risk, 2) that the scope of recognized land use activities that reduce atmospheric GHG needs to be expanded if the benefits of REDD are to be fully realised and 3) that informal land tenure may not require transformation prior to successful, sustainable implementation. A review of the appropriateness of community-based forest carbon monitoring found that such an approach presents significant cost savings while providing local employment and incentive opportunities. Exposure to such initiatives to date indicated that the quality of data collected is adequate and sufficiently robust to fulfill project and national-scale reporting and verification requirements.

Abstract (Afrikaans translation)

Antropogeniese klimaatsverandering hou daadwerklike uitdagings vir die mensdom in. Huidige voorspellings dui daarop dat klimaatsverandering natuurlike sisteme, gesondheid en die ekonomie op 'n verskeidenheid van vlakke gaan beinvloed. Daar is dus 'n dringende nood aan korrekte en effektiewe aanpassings- en mitigasie maatstawwe wat geimplementeer kan word.

Hierdie proefskrif fokus op die risiko en lewensvatbaarheid van grondgebruiksgebaseerde klimaatsverandering mitigasie. Eerstens gee dit 'n oorsig van die beleids en implementasie dryfvere wat noodsaaklik is vir die bevordering van permanentheid en die verlaging van die risiko van lekkasie wat geassosieer word met verlaagde emissies vanwee degradasie en ontbossing (VEDO[2]) in sub-Sahara Afrika. Tweedens analiseer dit die transaksiekoste wat geassosieer word met die vermyding van ontbossing en herbebossing en die effek daarvan op die finansiele lewensvatbaarheid van sulke aktiwiteite in bosveld en weiveld. Derdens ondersoek die proefskrif die biofisiese risiko faktore (soos vuur) op die uitslag van VEDO aktiwiteite in twee hoofstukke. Die eerste hoofstuk word ingelei deur 'n ontleding en verklaring van biofisiese risiko en gee dan 'n oorsig oor die risiko van vuur op VEDO projekte in belangrike plantegroei-tipes in Afrika. Die tweede hoofstuk maak gebruik van die Century Ekostelsel Program om die impak van voorspelde veranderings in temperatuur, reenval en atmosferiese koolstofdioksied op VEDO aktiwiteite te evalueer.

Die resultate dui aan dat onder die biofisiese risiko faktore wat ondersoek is, vuur nie so 'n belangrike risiko inhou vir VEDO projekte in die bosveld, savanna en grasveld plantegroeitipes in Afrika nie. In teenstelling hou vuur 'n groot risiko in vir nat woude waar ongekende droeë tydperke kan veroorsaak dat vuur wel mag voorkom

[2] Wat betref korrekte terminologie, die afkorting VEDO staan vir Verlaagde Emissies vanwee Degradasie en Ontbossing. Meer onlangs het die term VEDO+ sy verskyning gemaak en dit staan vir vermyding van ontbossing en degradasie sowel as woud-restorasie, rehabilitasie, volhoubare bestuur, bosaanplanting en herbebossing. Alhoewel die VNRKKV teks nog geen formele definisie vir die term VEDO aanvaar het nie, word dit algemeen aanvaar dat dit alle woude, bosveld en weidingsaktiwiteite insluit wat tot 'n netto verlaging in die konsentrasie van atmosferiese kweekhuis gasse lei. Die afkorting VEDO word dus deurgaans in hierdie proefskrif gebruik behalwe in gevalle soos in Hoofstuk 4, wat op die finansiele lewensvatbaarheid van aktiwiteite fokus, waar 'n duideliker en afsonderlike oorweging van VEDO en herbebossing aktiwiteite vereis word.

in ‘n stelsel waar dit voorheen afwesig was. Die analise op die effek van klimaatsverandering het bevind dat veranderinge in klimaat tipies sal lei tot a toename in koolstof voorrade en verhoogde sekwestrasie tempos vir die plantegroeitipes wat geevalueer is. Daar was egter uitsonderings, soos byvoorbeeld die gemodeleerde impakte op nutrientryke savannas wat verdere ondersoek benodig.

Die analise ten opsigte van die transaksiekoste wat gepaardgaan met VEDO aktiwiteite illustreer dat sulke kostes dalk die lewensvatbaarheid van kleinerskaal projekte mag benadeel, veral in ekostelsels anders as nat woude met relatief lae koolstof voorrade en geassosieerde inkomste. Die voorgestelde skepping van kapasiteit op ‘n nasionale vlak mag dalk transaksie koste verlaag tot ‘n mate, maar daar is duidelik ‘n behoefte om beter insigte te verkry oor die ware kostes van VEDO aktiwiteite.

Wat betref die vereiste ondersteunende beleid en implimentasie kapasiteit is daar bevind dat 1) multi-kriteria grondgebruik beplanning uiters belangrik is in die verlaging van permanentheidsrisiko, 2) die omvang van erkende grondgebruiks aktiwiteite moet uitgebrei word om ten volle voordeel te trek uit VEDO, 3) dat informele grondbesit dalk nie transformasie vereis voor suksesvolle, volhoubare implementasie nie. ‘n Oorsig oor die aanvaarbaarheid van gemeenskapsgebaseerde woudkoolstofmonitering het gevind dat so ‘n benadering tot groot kostebesparings lei terwyl dit ook plaaslike werkskepping bevorder en dien as dryfveer vir projekte. Blootstelling aan sulke inisiatiewe tot op hede dui aan dat die kwaliteit van die data wat ingesamel is voldoen aan projek- sowel as nasionale vlak verslaggewingsvereistes.

Acknowledgements

First, I thank Albert van Jaarsveld for providing me with the freedom and privilege of writing a dissertation on a subject of my own choice. I am deeply grateful for his patience, guidance and support through one of the more difficult periods of my life when my Mom passed away. Albert exposed me to the Millennium Assessment and introduced me leading thinkers and conservation organisations that initiated a career beyond my dissertation work.

When I asked Bob Scholes to be my co-supervisor at a meeting in Stockholm, I could never have anticipated the profound influence he would have on my work, career and the way I think about systems and concepts. Bob is a true mentor who has shared a contagious enthusiasm for science and investigating new ideas, while patiently schooling me in the fundamental basics of scientific and professional discipline. This dissertation would certainly not have been possible without his guidance and support for which I am deeply thankful. The time spent with his research group at the CSIR was also a real treat. I must express my thanks to Helen de Beer and the rest of the group for their hospitality and shared enthusiasm for what we do.

My deepest thanks go to Allan Ellis who kindly accepted the unglamorous task of adopting a PhD student late in their studies. The completion of this dissertation would have been extremely difficult without Allan's support and supervision.

I am also grateful to informal supervisors who have provided me with much of their time and advice. Especially James Blignaut for introducing me to ecological economics, Russell Wise for teaching me the basics of financial modeling, Neil Krige who allowed a biologist to attend his investment finance courses, and Bill Parton who provided tuition on the Century Model and shared his enthusiasm for modeling systems.

To mates who have shared their stoke for investigating interesting ideas, mutterings about admin and Chelsea buns at lunch time. Particularly Wayne Delport, Wendy Foden, Sandi Willows-Munro, Barend Erasmus, Jen Jones, Aimee Ginsburg, Mark Keith, Frans Radloff and Sally Archibald. I am especially grateful to Ilana van Wyk, Sandi Willows-Munro, Leon Theron and my Dad for proof-reading draft chapters. Through his assistance with client work, Leon has also provided me with the precious time required to complete my dissertation over the last 18 months for which I am sincerely thankful.

The South African National Research Foundation, the Millennium Assessment, and Ernst and Ethel Eriksen Trust are acknowledged for their financial support. I am also indebted to the Department of Botany and Zoology and are especially grateful to Belinda Reyers for her support in the early part of my PhD while managing a delinquent group of students.

My special thanks go to Ilana van Wyk for her love, encouragement and unrelenting belief in me. She also opened my mind to the social and political aspects of global change which is reflected in many of the ideas presented in this thesis.

Lastly, to my parents Ron and Pauline Knowles and my sister Tanya for their unconditional love, encouragement and support throughout my studies. This dissertation is submitted in lieu of a good report.

Table of Contents

List of Acronyms

AFOLU	Agriculture, forestry and other land use
AR	Afforestation / reforestation
CAIT	Climate Analysis Indicators Tool
CAM	Crassulacean acid metabolism
CBFM	Community-based forest carbon monitoring
CBNRM	Community based natural resource management
CBO	Community based organization
CCBA	Climate, Community and Biodiversity Alliance
CCBS	Climate, Community and Biodiversity Standard
CCSR	Centre of Climate System Research
CDM	Clean Development Mechanism
CO2	Carbon dioxide
[CO2]	Atmospheric carbon dioxide
COP	Conference of Parties
CS	Carbon sequestration
CSIRO	Commonwealth Scientific and Industrial Research Organization
DBH	Diameter at breast height
DRC	Democratic Republic of the Congo
ERU	Emission reduction unit
FAO	Food and Agriculture Organization (United Nations)
FCPF	(World Bank) Forest Carbon Partnership Fund
FSC	Forestry Stewardship Council
GCM	General Circulation Model
GHG	Greenhouse Gas
GPS	Global positioning system
GrADS	Grid Analysis and Display System
ha	Hectare
HFC	Hydrofluorocarbon
HFLD	High forest cover with low rates of deforestation
IPCC	Intergovernmental Panel on Climate Change
IPCC-DDC	IPCC Data Distribution Centre
IT	Information technology

K:TGAL	Kyoto: Think Global, Act Local
KP	Kyoto Protocol
MODIS	Moderate Resolution Imaging Spectroradiometer
MRV	Monitoring, Reporting and Verification
N	Nitrogen
NAMA	Nationally appropriate mitigation actions
NEP	Net Ecosystem Productivity
NGO	Non-Governmental Organisation
NPP	Net Primary Productivity
NTFP	Non-timber forest product
PDA	Personal digital assistant
PDD	Project Design Document
PES	Payment for Ecosystem Services
PET	Potential evapo-transpiration
PGIS	Participatory geographical information systems
PPM	Parts per million
REDD	Reduced emissions from deforestation and degradation
REDD+	REDD + forest restoration, rehabilitation, sustainable management, afforestation and reforestation.
REL	Reference emission level
SD-PAM	Sustainable development, policies and measures
SSA	Sub-Saharan Africa
tCO2e	Ton of carbon dioxide equivalent
UN	United Nations
UN-REDD	United Nations REDD program
UNEP	United Nations Environmental Program
UNFCCC	United Nations Framework Convention on Climate Change
UV-B	Ultraviolet B
VCS	Voluntary Carbon Standard
VER	Verified emission reduction
VFT	Vegetation functional type
WRI	World Resources Institute

Chapter 1

Introduction

Following the publication of the Intergovernmental Panel on Climate Change's (IPCC's) 2007 reports, it is now widely accepted that anthropogenic activities are leading to changes in the earth's climate beyond that which would naturally be expected (Denman et al. 2007). The reports also drew attention to the relative importance of greenhouse gas (GHG) emissions generated through land use change. Together they account for one fifth of annual global GHG emissions (Denman et al. 2007). If human-driven climate change is to be stabilised at a level that prevents dangerous anthropogenic interference with the earth's climate, it is crucial that GHG emissions generated through land use, particularly through deforestation and ecosystem degradation, are reduced considerably.

The establishment of a robust scientific basis for addressing GHG emissions generated through land use led to the adoption of supporting policy in the United Nations Framework Convention on Climate Change (UNFCCC). Key decisions adopted at the last number of Conference of Parties (COP) meetings to the Convention, from the Bali Action Plan, to the Copenhagen Accord, to the decisions adopted at the 16th COP in Cancun in December 2010, clearly include and support the implementation of activities that reduce GHG emissions generated through deforestation and degradation (REDD). In turn, the broader adoption of land use based climate change mitigation activities within the UNFCCC has led to the allocation of considerable financial resources to the development of REDD and the initiation of the UN-REDD and the World Bank Forest Carbon Partnership Fund (FCPF) programs aimed at establishing national-scale implementation capacity and supporting policy.

Further to its contribution to mitigating climate change, the importance of avoiding deforestation and maintaining intact ecosystems, in terms of providing ecosystem services to humankind, has been brought to the fore by the Millennium Ecosystem Assessment (Scholes and Biggs 2004) and more recently the Economics of Ecosystems and Biodiversity study (TEEB 2010). The resilient and sustained provision of ecosystem services also forms the basis for 'ecosystem-based climate change adaptation', which is emerging as an important yet practical means for human societies and biodiversity to adapt to the potential detrimental effects of climate change (Vignola et al. 2009).

Despite these significant developments in science and policy at a global scale, significant questions remain regarding the nature of the financial feasibility of REDD activities and the risk of expected GHG emission reductions not being achieved. In addition, there is a need to understand how policy and institutional frameworks influence risk and feasibility in REDD activities and how they may be structured at a national- and sub-national scale to adequately support the implementation of REDD activities over 20-30 years or longer. This thesis aims to assess such questions regarding risk and feasibility, and how policy and implementation options should be structured.

These questions of risk and feasibility are perhaps easier to conceptualise if one compares land use based climate change mitigation activities to an industrial alternative. In an industrial sector project the emission reduction activity may be confined to a single location within a factory or industrial plant. The emission reduction activity usually occurs through a well-defined engineering process where technological and operational risks are relatively straightforward to estimate and manage accordingly. The owner of the factory is a registered company under the host-country's constitutional law. Furthermore, the owner typically has considerable assets and interests located in the host-country and would form an adequate legal counterparty in case of default.

In comparison, REDD or reforestation activities typically occur over vast spatial areas where many types of activities may be involved. In sub-Saharan Africa for example, REDD activities may include the halt of a logging concession, the establishment of a forest reserve in an area previously under deforestation pressure, or community-based forest management in a more populated, human-dominated landscape. Land use based activities are exposed to biophysical risk in the form of the effect of fire, drought or changes in climate to name a few. As the land on which the activity is based could be used for alternative purposes, REDD activities may also be exposed to vagaries of governance at a local and national level. Socio-economic pressure may lead to a change in land use priorities with resultant deforestation, ecosystem degradation and the associated release of sequestered carbon into the atmosphere.

These additional issues, which are essentially concerned with permanence and leakage risks[3], were among the main reasons why the recognised scope of land use based climate change mitigation activities in the first commitment period to the Kyoto Protocol (KP) were limited to reforestation and afforestation projects. Permanence and leakage risks in particular, led to the exclusion of REDD as a recognised climate change mitigation activity in the first commitment period of the Kyoto Protocol to the UNFCCC. The Kyoto Protocol, which requires developed countries to limit their GHG emissions, includes flexible mechanisms such as the Clean Development Mechanism that allows parties to implement GHG emission reduction projects in developing countries as a means of realising net GHG emission reduction commitments. Due to the permanence, leakage and sovereignty concerns, however, land use based climate change mitigation activities were limited to project-scale reforestation and afforestation initiatives. Whereas REDD is likely to be included in some form in post-2012 UNFCCC policy, a better understanding of the risk and economics of REDD is crucial if it is to reach its full potential.

This dissertation explores forms of risk and uncertainty that are particular to land use based climate change mitigation activities, particularly the potential effect of fire to REDD activities and the potential impact of projected changes in the climate itself. Few publications to date have focused on the impact of such 'biophysical' sources of risk and uncertainty on the outcome of REDD activities, which need to be considered to assess the efficiency of REDD compared to other climate change mitigation and land use options. For the purposes of this dissertation, the classic economic definitions of 'risk' and 'uncertainty' as defined by Knight (1921) are adopted, where risk is present when future events occur with *measurable probability* and uncertainty is present when the likelihood of future events is *indefinite or incalculable*.

In addition, the costs associated with land use based climate change mitigation activities are assessed. To date, the majority of analysis have solely focused on the opportunity costs of REDD (Stern 2006, Grieg-Gran 2006) with little consideration of

[3] The potential for carbon sequestered through land use change to be released into atmosphere during the course of a project's lifetime is referred to as permanence risk. Leakage occurs where the cause of deforestation or land degradation is not truly addressed but is diverted outside of the project area. For example, a timber concessionaire or producer of charcoal may merely move from inside the project area to a location elsewhere in the region.

transaction, development or implementation costs. Few have assessed the transaction cost of realising the carbon assets of REDD activities. The cost of documentation, monitoring, verification, as well as legal and brokerage fees should be more carefully examined. In addition to the costs incurred by an industrial or energy sector project, a land use based activity is likely to attract much higher costs associated with monitoring, scenario development, local- and national stakeholder engagement, and socio-economic and biodiversity survey costs. This study is one of the first to clearly include the cost of local- and national stakeholder engagement. Such engagement is currently required to obtain project certification through the Climate, Community and Biodiversity Alliance standard (CCBA) and is likely to be incorporated, if not expanded in future REDD policy and project implementation requirements due to growing pressure from indigenous people's rights advocacy groups and the need to acquire the informed consent of local residents.

An important aspect of considering the risk and feasibility of REDD activities, is how the structure of national policy and implementation frameworks affect the viability of such activities and how they could be structured to reduce the risk of delivery as well as improve cost efficiencies. This is a pertinent topic at the moment as many countries develop national-scale REDD programs through the UN-REDD and World Bank FCPF processes and consider where to place REDD activities, whether or not to reform land-tenure, which activities could be in the scope of recognised REDD activities, and where opportunities lie to employ local residents who have foregone previous livelihood practices that lead to deforestation. An initial review paper is presented as well as a chapter focusing on creating employment opportunities though community-based forest monitoring. These assess how national policy and implementation approaches may be structured to reduce risk and improve feasibility.

Climate regulation is one of many ecosystem services. Other services include water flow and sedimentation management, which require the management of ecosystems across vast spatial areas that may be populated to a certain degree and exposed to similar forms of biophysical and anthropogenic risk. While this dissertation focuses on REDD and reforestation activities in particular, the discussion and findings may be applicable to development and implementation of ecosystem service investments more broadly as well as ecosystem-based climate change adaptation interventions.

The structure of the dissertation thesis

This dissertation is comprised of five core chapters (four have been submitted as papers to peer-reviewed journals and the fifth has been published as a book chapter). Each is a stand-alone piece and is formatted in the style required by each journal. Together they have the objective of assessing the feasibility and risk associated with land use change mitigation ventures located in sub-Saharan Africa. As such, each chapter contains a review of the literature positioning the chapter within the current understanding and knowledge of the subject. To avoid replication, a single list of references has been compiled at the end of the thesis.

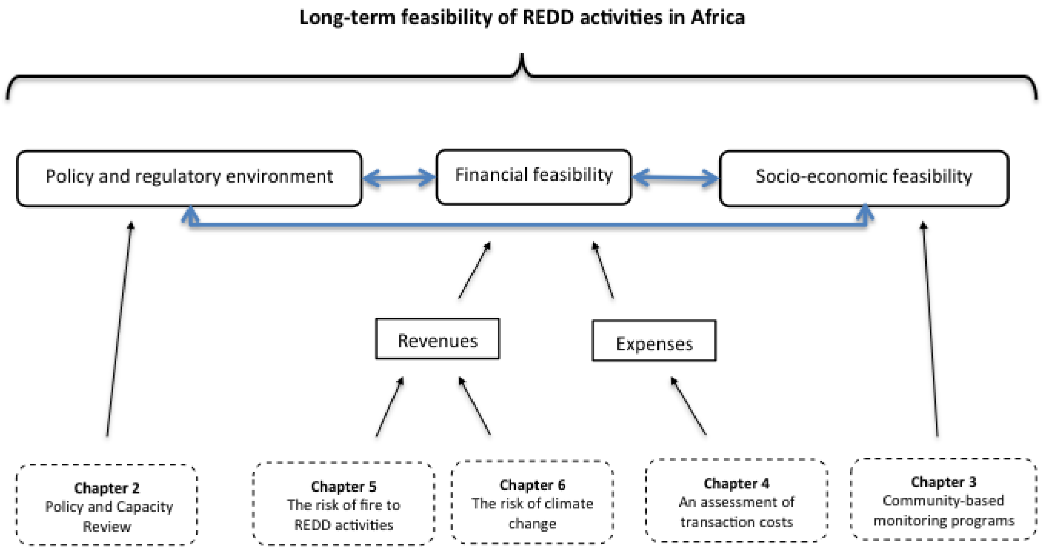

Figure 1: A conceptual framework of thesis

The first paper of the thesis considers policy and implementation issues that are particularly important and specific to the realisation of REDD activities in the ecological, socio-economic and land tenure context of sub-Saharan Africa. It considers potential means of reducing permanence and leakage risk through multi-criteria national land use planning, the creation of appropriate incentive mechanisms, and establishing adequate governance throughout the supply chain of forest commodities. In addition, it considers the manner in which established land use based enterprises in the region have previously dealt with informal land tenure while remaining viable over decadal time-scales. Such considerations are crucial if REDD and 'ecosystem-based climate change adaptation' are going to be widely accepted and realised in the region.

The second paper expands on the theme of creating local incentives and explores the applicability of 'community-based forest carbon monitoring' as a means of cost efficiently monitoring changes in terrestrial carbon stocks while creating local employment and a sense of local ownership and value in forest systems.

This is a challenging issue for the developers of REDD activities who need to address the true drivers of deforestation. In the context of sub-Saharan Africa, this generally requires the creation of alternative livelihoods in remote areas that replace prior activities such as logging, the production of charcoal and cash crops. Monitoring and policing a forest may be able to provide some employment and income opportunities that are sustainable over the lifetime of the project. While monitoring and policing requirements certainly won't employ everyone, it may at least employ some local residents and lead to a local sense of ownership which in turn may increase the likelihood of the project persisting over its expected lifespan and decrease associated permanence risk. It may also decrease leakage risk as it employs individuals who would otherwise need to pursue alternative livelihoods elsewhere which are likely to include either small-grower farming, timber extraction or charcoal production. Chapter 3 explores these themes and assesses whether community based monitoring provides appropriate and adequate data for project- and national scale GHG accounting purposes.

In addition to the establishment of appropriate supporting policy, capacity and incentives for implementation, a key issue is the financial feasibility and attractiveness of REDD activities as a land use choice. In the third paper, the focus is on the financial feasibility of land use based climate change mitigation activities located in southern Africa. Whereas REDD may make good sense from a climate regulation perspective and may maintain ecosystem services, is it financially viable?

Previous analyses of the economic feasibility of REDD activities have generally considered opportunity costs alone. Other expenses such as the cost of project development, management and operations, documentation, monitoring, local and national stakeholder engagement, legal fees, marketing and buyer engagement, verification and brokerage fees, have generally not been considered. The cost of stakeholder engagement and expenses incurred through developing and implementing alternative livelihood practices are especially neglected in these analyses.

Although seldom explicitly stated, there is a general underlying assumption that transaction, development and operational costs are marginal. This assumption may be valid for large-scale REDD projects located in moist forests in the tropics where carbon revenues are considerable but it does not hold for smaller-scale ventures and projects located outside of the moist tropical forests where carbon stocks and associated revenues are comparatively low. Furthermore, REDD activities located outside of remote tropical forests, particularly in the populated landscapes of east and southern Africa are likely to require substantial additional stakeholder engagement, the development of alternative livelihoods that comprehensively address deforestation drivers, as well as operations management, that may come at a considerable cost. This chapter assesses the nature of these transactions costs.

Chapters 5 and 6 consider two forms of biophysical risk that may affect the projected outcome of REDD activities. In addition to conventional forms of risk that are usually considered, for example, operational, technological, or exchange rate risk, investments in land use based activities that require the maintenance of intact systems across vast spatial scales, are exposed to a particular class of risk that is related to exposure to biophysical factors such as fire, droughts, pests and changes in climate. In Chapter 5, biophysical risk is defined as: ‘the deviation from expected returns in an ecosystem service investment due to variability in prevailing biophysical conditions such as rainfall, temperature, fire or pests’.

Few studies to date have focused on the potential effect of biophysical risk to the GHG emission outcome of REDD ventures or similar activities elsewhere. Consideration of such risk is not only important to land use based climate change mitigation activities but ‘ecosystem-based climate change adaptation’ and investment in ecosystem services more broadly. Adequately estimating biophysical risk changes it to a known quantified risk factor that can be managed as a source of uncertainty to investors.

Chapter 5 is one of the first papers to consider the risk fire presents to REDD activities. Fire has been cited broadly as a risk factor that needs to be considered when assessing the feasibility of REDD activities (Pierce et al. 2004, Running 2006, Westerling et al. 2006, van der Werf et al. 2008, Conrad and Soloman 2009, Ethiopia 2010, Kenya 2010, Tanzania 2010), yet few reviews, computer models or empirical analysis exist on the subject.

The review, based on published literature, considers the risk of fire on the outcome of REDD activities in predominant African vegetation types. It explores the nature of fire risk and attempts to understand the impact of fire on the variability in expected returns through a combined consideration of the size and location of carbon stocks, fire regimes, the fraction of the net carbon pool released into the atmosphere through a typical fire event and the rate of recovery of released carbon.

The second paper to consider biophysical risk seeks to assess the effect of projected changes in climate and atmospheric CO_2 on the carbon sequestration outcome of reforestation and avoided deforestation activities using a computer simulation approach. Anthropogenic climate change may manifest itself in a number of ways including changes in mean annual rainfall and temperature, as well as change in the occurrence and intensity of extreme weather events. As two of the key determinants of plant growth are precipitation and ambient temperature, it was decided to assess the impact that projected change in climate may have on ecosystem productivity, associated carbon sequestration rates and standing carbon stocks.

Since the size of the carbon pool at the end of activity's lifespan essentially determines the number of emission reduction units (ERUs) awarded to the project and revenues, consideration of potential changes in rainfall and temperature is a crucial part of assessing the feasibility of land use based climate change mitigation activities. Such consideration is equally important to project- and national-scale activities, where it may be appropriate for national planners to include an assessment of the potential effect of projected changes in climate on a national portfolios of REDD projects prior to implementation.

In addition to changes in climatic variables, attention has also been paid to the potential effect of rising atmospheric carbon dioxide ([CO2]) on plant growth, biomass stocks and the relative abundance of C3 woody species compared to C4 grass species in rangeland systems (Bond et al. 2003a, Kgope et al. 2010). Certain studies, for example Kgope et al. (2010) indicate that increases in [CO2] may have a considerable effect on carbon sequestration rates and observed carbon stocks.

The key questions the paper therefore seeks to answer are: How do projected changes in [CO2], temperature and rainfall affect the outcome of REDD and reforestation activities? What is the effect of the projected change in each variable when modeled separately? And, do projected changes in climate present a significant risk to the outcome of land use based climate change mitigation activities?

The thesis is concluded with a summary and synthesis of emerging outcomes in Chapter 7.

In terms of geographical relevance, although much of the discussion and analysis is globally applicable, the principal area of focus is the wooded ecosystems of sub-Saharan Africa.

Goals of the study

The goals of this dissertation are:

- To review policy and implementation issues that are key to the successful realisation of REDD activities located in sub-Saharan Africa
- To review the applicability of community-based forest carbon monitoring programs in terms of cost-effectiveness, creating local value and employment opportunities, and meeting local and international monitoring, reporting and verification requirements
- To assess the effect of transaction costs and economies of scale on the financial feasibility of REDD and reforestation activities in sub-Saharan Africa
- To estimate the potential impact of projected changes in climate on land use based climate change mitigation activities in sub-Saharan Africa
- To assess the risk of fire to REDD activities located in predominant African vegetation types

Chapter 2

Realizing African REDD: a sub-Saharan perspective on reducing deforestation

The ideas and concepts presented in this paper emerged during the course of providing advisory services related to the development of REDD and reforestation opportunities in South Africa, western Zambia, the Democratic Republic of the Congo, Liberia and northern Mozambique over the past five years. They arose from the need to answer the question: How could land use based climate change activities and associated policy be structured in the context of sub-Saharan Africa to ensure their success over at least the required 20-30 year project period, if not in perpetuity?

The paper is my own work with input from Dr Albert van Jaarsveld, Dr Allan Ellis and Dr Bob Scholes on the draft text. The ideas are the result of a review of applicable climate change policy, economics and anthropological literature as well as discussions with many parties, ranging from senior ministers, to leading academics, to a gentleman transporting charcoal to market on his bicycle in southern Malawi.

Realizing African REDD: a sub-Saharan perspective on reducing deforestation

Tony Knowles, Robert. J. Scholes, Albert. S. van Jaarsveld and Allan Ellis

Abstract

The potential inclusion of incentives for action taken to reduce emissions from deforestation and forest degradation (REDD) in post-2012 international climate policy provides a significant opportunity for sub-Saharan Africa to contribute to global climate change mitigation efforts. During the negotiation period, a window of opportunity exists in which to structure REDD policy and implementation options in a manner that improves the likelihood that the greenhouse gas emission benefits are realized, while at the same time meeting socio-economic and biodiversity-related objectives.

Key policy issues are reviewed from a sub-Saharan African perspective, considering permanence, leakage and impacts on climate, livelihoods and ecological services. The integration of REDD with multicriteria land use planning, the scope of incentives that may be considered, and the balance between market- and fund-based approaches to payment for REDD are discussed. Three implementation issues are particularly pertinent in this region: land tenure; structuring appropriate incentives; and ensuring the integrity of the forest commodity supply chain.

The fundamental future challenges to the implementation of REDD in sub-Saharan Africa revolve more around development studies and emerging-nation economics than around forest management and climate system science. While international policy, efficient monitoring systems and knowledge of forest ecology are crucial and in need of substantial further development, the principle unresolved issues appear to relate to permanence and liability.

Introduction

As the world negotiates post-2012 climate change policy, the possibility of reducing greenhouse gas (GHG) emissions through avoided deforestation and forest degradation (Reducing Emissions from Deforestation and Forest Degradation - REDD) is being brought to the fore. Globally, GHG emissions generated through agriculture, forestry and more broadly, land-use (AFOLU), account for almost 20 percent of anthropogenic emissions (Gullison et al. 2007, IPCC 2007). In sub-Saharan Africa (SSA) in particular, 73 percent of human-produced emissions are generated through land-use activities (IPCC 2007, WRI 2008).

The importance and value of REDD in SSA goes far beyond reducing atmospheric GHG. While REDD provides an important opportunity for African nations to contribute to global climate change mitigation efforts, the potential social, economic and biodiversity benefits through improved forest governance and ecosystem service maintenance are also considerable (Guo et al. 2001, Scholes and Biggs 2004). Where 70 percent of the continent's people live in rural areas and are dependent on their immediate surroundings for a significant portion of their livelihoods, ecosystem integrity and services are closely linked to human well-being (Shackleton et al. 2007, Laye and Gaye 2008). Due to the strong, direct link between ecosystem services and rural livelihoods, the rural poor of SSA have been consistently identified as one of the most vulnerable groups to climate change (Thornton et al. 2006). Maintaining ecosystem structure and function can therefore potentially not only mitigate climate change but also contribute to fundamental climate change adaptation capacity. In a region where rainfall is projected to decrease over large areas and extreme events such as droughts and floods are expected to increase (IPCC 2007), managing ecosystem services such as water flow regulation and soil fertility is a crucial but practical means of adapting to climate change.

To ensure the opportunities and benefits outlined above are realized, REDD needs to be adopted in its broadest form while its implementation needs to be designed and structured in an environmentally and socially appropriate manner from the outset (Boyd et al. 2007, Myers 2007, Swart and Raes 2007, Cotula and Mayers 2009). Expected additional outcomes from REDD, such as the socio-economic, biodiversity, ecosystem service and climate change adaptation, need to be included in the design phase of REDD projects instead of being added on as the implementation progresses.

There is window of opportunity to structure policy in such a way that expected outcomes are realized. It also presents an opportunity to step back and reflect on the lessons learnt from Clean Development Mechanism (CDM) implementation in the first commitment period of the Kyoto Protocol (UNFCCC, 1998). The CDM was not a success in Africa, especially where land-use activities are concerned (World Bank, 2009), but the CDM process does provide valuable insights that should be capitalized on if REDD is to succeed.

Since the inclusion of the REDD within the 'Bali Action Plan' at the 13th Conference of Parties meeting in December 2007, significant progress has been made towards the development of REDD policy and institutional, legal and implementation options (Streck and O'Sullivan 2008, Angelsen et al. 2009, Streck et al. 2009). In addition, vast strides have been made towards implementation on the ground through the United Nations REDD program and the World Bank's Forest Carbon Partnership Facility. Our intention here is not to propose a completely new policy framework, but to provide a SSA perspective on key issues that are important to the success of REDD in the region.

First, we focus on issues which are generic to the implementation of REDD globally. Thereafter, we zoom in on three implementation issues that are more particular to the SSA region. The review considers REDD options in their broadest form, not only focusing on tall, closed tropical rainforest loss in West and Central Africa, but also on deforestation and ecosystem degradation in the more open, shorter, drier savannas and woodlands of East and southern Africa. The latter occupy a larger area and are arguably in the front line of degradation and deforestation.

1. A sub-Saharan African perspective on generic REDD issues

One of the main challenges in developing REDD policy and implementation options is its multi-disciplinary nature and the spread of driving forces across a range of spatial scales. Successful implementation requires consideration of, among other things, (1) determining realistic emission reduction goals, (2) calculating GHG emission reductions, (3) efficient monitoring, reporting and verification, (4) determining stakeholder rights and concerns as well as (5) compensation options, national land-use priorities, and national policies and measures (Angelsen et al. 2009, Streck et al. 2009). Here we review three issues that have been prominent in discussions on the global development of REDD, that are also important to the realization of REDD in the region, namely:

- The integration of REDD within a broader framework of multi-criteria national land-use planning
- Expanding the scope of permissible projects, in terms of target ecosystems, carbon pools and processes within ecosystems and the mitigation activities allowed
- Moving from market-based revenues to a broader funding approach

1.1 The need for multi-criteria national land-use planning

In addition to the potential inclusion of REDD as a recognized climate change mitigation activity, the literature has proposed a shift in governance from a project to a national scale. Leakage, permanence and sovereignty concerns are the main reasons for the proposed change in governance scale (Ebeling and Yasue 2008, Strassburg et al. 2009).

Moving from project scale to national scale implementation makes good sense for SSA nations. In addition to helping address concerns regarding the ‘leakage’ of GHG emissions from the project area to other parts of the territory and the sovereignty of countries regarding land use decisions within their territory, this shift is advantageous in terms of economies of scale, and developing local capacity and knowledge for climate change mitigation as well as adaptation activities. Shifting REDD policies to the national scale also allows for the entrenchment of polices for adaptation and mitigation within government legislation (Visseran-Hamakers and Glasbergen 2007, Ebeling and Yasue 2008, Tschakert et al. 2008).

In addition to the current proposed national REDD strategies and options (Angelsen et al. 2009, Streck et al. 2009), it may be prudent to conduct a comprehensive multi-criteria land-use planning exercise to improve the probability of permanence. By 'multi-criteria land-use planning' we mean a broad planning process where ecosystem service and alternative land-use trade-offs are assessed at a national scale prior to REDD implementation. Such a planning process also provides an opportunity to plan the national-scale management of ecosystem services as well as plan ‘ecosystem-based adaptation’ (Vignola et al. 2009).

The main reason and need for a land-use planning exercise to be concluded upfront, is that the implementation of REDD essentially requires a host nation to commit a substantial portion of the country to a certain land-use, if not in perpetuity, then for a substantial period of time (Fearnside 2002). This may seem obvious, but the implications are enormous in terms of foregone opportunity costs and options for alternative land uses. This is especially true for countries with high forest cover, such as Angola, Mozambique, the DRC and Liberia. Alternative land-use options such as agriculture or mining need to be explored prior to the establishment of REDD goals and implementation. The planning process also provides an opportunity to downscale and appropriately adapt mechanisms developed at a global scale to local context realities and existing rural development strategies (Boyd et al. 2007) . In addition, mechanisms, procedures and approaches for revising land-use plans in the context of established REDD agreements could be defined *a priori*.

Practically, a complete national land-use plan will take a considerable amount of time. The intention of undertaking a multi-criteria planning exercise is not to develop a 'final' land-use plan that a developing nation governments would need to commit to in perpetuity but to provide a useful and necessary tool for assessing opportunity costs and alternative land-use options at a particular moment, prior to REDD implementation. The process may also provide an opportunity for multiple legitimate contested values and interests to be explicitly recognised and considered in land-use planning decisions.

A typical suite of criteria and objectives that would be used in the assessment of a planning process are:

- Potential REDD benefits in terms of avoided GHG and net radiative forcing
- The opportunity cost, measured as the forgone production or development opportunities of alternative land uses, such as logging, cash crops or mining
- Local livelihood implications in terms of employment, resource tenure and additional benefit flows such as access to food, fuelwood and water
- Ecosystem service trade-offs such as with food security, including ‘positive tradeoffs’ such as improved water quality
- Biodiversity and conservation implications
- Climate change adaptation priorities
- Governance and tenure of the areas targeted for REDD

- Dealing with shifting land-use plans

For the planning process to be effective and to add value, it first needs to be pragmatic, balancing the pressing need for implementation with essential requirements. The intention is not to introduce another time-consuming hurdle to REDD implementation. Secondly, the aim is to draw on the established multi-criteria land-use planning models such as the Analytic Hierarchy Process (Ananda and Herath 2008, 2009) and the methodologies developed through the Millennium Assessment (Scholes and Biggs 2004, van Jaarsveld et al. 2005). Thirdly, the process would include all relevant Government departments as well as other stakeholders to avoid the 'silo affect' where Ministries and institutions work in isolation leading to a particular area been demarcated for several land uses (Angelsen 1999, Ali et al., 2005, Crittenden and Wilder, 2008, Cotula and Mayers 2009). Lastly, a phased strategic approach may be taken to expedite the process of prioritizing areas under high risk of deforestation and degradation. A realistic lead-time of at least three to five years prior to the initiation of national scale REDD implementation and monitoring seems appropriate. For this reason, leading initiatives such as the United Nations REDD program and the World Bank Forest Carbon Partnership Fund have proposed a phased approach to implementation (Angelsen et al. 2009, Streck et al. 2009). A similar, practical and phased approach for a land-use planning exercise could be adopted as well.

In addition, the process provides an opportunity to plan land-use based climate change adaptation strategies and the often cited sustainable development and poverty alleviation co-benefits (Boyd et al. 2007, Bizikova et al. 2007, Myers 2007, Streck and O'Sullivan 2008, Cotula and Mayers 2009). Landscape adaptation, mitigating climate change through REDD, carbon sequestration and managing ecosystem services through restoring ecosystem structure and function are integrated processes and reinforce one another (Swart and Raes 2007, Bizikova et al. 2007, Chazdon 2008, Miles and Kapos 2008).

1.2 Expanding the scope of REDD

A significant constraint to the widespread rollout of land-use based climate change mitigation ventures in SSA under the CDM (UNFCCC 1998) were limitations placed on recognized mitigation activities, vegetation types and carbon pools in which activities were allowed to take place (Zahabu et al. 2007). Under the CDM, mitigation activities are limited to afforestation and reforestation. It excluded REDD as well as carbon sequestration belowground (in the soil and roots where much of the carbon in dryland ecosystems occurs), avoided soil carbon degradation, fire management, and reduced tillage (www.cdm.unfccc.int, Watson et al. 2000). CDM land-use project activities are limited to 'forest' vegetation types defined as a "vegetation type with tree crown cover of more than 10-30 per cent with trees with the potential to reach a minimum height of 2-5 meters at maturity in situ" (UNFCCC, 1998, 2001). While this broad definition of a forest includes almost all of the woodland, savanna and sub-tropical thicket biomes of SSA, it excludes activities in grasslands, shrublands, wetlands and peat systems.

There is, in our view, good reason to reconsider regulatory constraints as land-use climate change mitigation policy are renegotiated prior to 2012. Expanding recognized mitigation activities and vegetation types may not only lead to increased greenhouse gas emission reductions, but also promote additional climate change adaptations, human livelihood and ecosystem service benefits. In terms of reducing greenhouse gas emissions, 73 percent of emissions from sub-Saharan Africa are generated from deforestation, degradation, rangeland fire and agriculture activities (WRI, 2008). Moreover, non-forest rangeland ecosystems contain a third of the globe's terrestrial carbon and cover an area of approximately 2.1 billion hectares in Africa (Watson et al. 1997, Watson et al. 2000, Noble 2003). Regulatory constraints on permissible mitigation activities and vegetation types have therefore significantly curtailed the potential of CDM and opportunities for African countries and are in danger of doing so again in a post-2012 agreement if not revisited.

The rangelands (i.e. shrublands, grasslands and open savannas) of SSA are home to many more people than are the moist closed forests. Sixty-two percent of the population of SSA live in rural areas with 80% of the poor dependent on subsistence agriculture and their immediate surrounds for food, water, shelter and energy (Faures and Santini 2008). REDD and maintenance of soil carbon through soil preservation, zero-tillage and soil management has been widely identified as an appropriate response to expected decreases in rainfall and increased temperature variability in the region (Smith et al. 2007, Thompson et al. 2009). The restoration and maintenance of rangeland systems not only makes sense in terms of reducing atmospheric GHGs, but forms a practical climate change adaptation strategy in its own right (Bizikova et al. 2007).

The urban human economies of the region rely on river catchments, often ‘forested’ in the broad sense, for water services, both in terms of flow and sediment regulation (Scholes and Biggs 2004, Farley et al. 2005). The Zambezi River basin, for example, is home to 32 million people who are reliant on the river for water services. Maintaining the river’s catchment - the grasslands and woodlands of western Zambia and eastern Angola - would protect urban economies within the entire watershed. A similar situation exists for the temperate grasslands of the Drakensburg escarpment in Lesotho and South Africa as well as the forests of the East African Arc Mountains of Kenya and Tanzania. In a region were rainfall is expected to decrease and become more sporadic (IPCC 2007), maintaining river catchment structure and function is vital to the resilience of downstream human economies to climate change. Yet, both catchments in southern and East Africa, are under clear threat of deforestation and grassland degradation – approximately 300,000 hectares of the Eastern Arc Mountains were deforested between the year 2000 and 2005 leading to a loss of almost 50 percent of indigenous forests and the release of 90 million tons of carbon dioxide into the atmosphere (Milledge and Elibariki 2005, Mblinyi et al. 2006, ECCM 2007). If the current baseline scenario continues, the forests of the mountain belt will be totally lost within 20 years (ECCM 2007).

The counter-argument against broadening permissible mitigation activity and vegetation types, and the reason for their initial exclusion from the CDM, is that monitoring changes in carbon stocks in non-forest vegetation types as well as below-ground soil carbon stocks is believed to be expensive and difficult. Some of this perception is poorly grounded in evidence – carbon pools form a continuum in terms of how expensive it is to estimate their magnitude to within a given absolute range. Particular pools should not be arbitrarily excluded from consideration – this could really be a project-by-project operational decision (Wise et al. 2009). In addition, there are concerns regarding the permanence of such activities and potential leakage. Especially at a CDM project scale, these concerns are partially valid. However, as post-2012 UNFCCC policy and regulation is negotiated, there is good reason to reconsider these previous restrictive and inhibitory regulatory limitations.

Implementing a broader suite of climate change mitigation activities will require significant capacity: an army of monitors, extension officers and co-coordinating personnel. This should not be seen as a roadblock to implementation but an opportunity for skill development and employment opportunities in remote rural areas.

1.3 Moving from market-based revenues to a broader fund approach

As always, one of the key negotiation issues is who is going to pay? There are currently three broad options under discussion: a compliance market approach, a fund mechanism, and a 'nested' option (Myers 2007, Johns et al. 2008, Strassburg et al. 2009). The compliance emission reduction market option is based on demand for emission reduction units created by the Kyoto Protocol. If an entity in a developed country exceeds its allocated emission allowance, one option is to purchase certified emission reduction units ('carbon credits') from a CDM project in a developing country thereby providing the additional capital for the project to be financially viable.

The 'fund option' in comparison is where the implementation of REDD in a developing country is financed through non-market funding generated in a developed country. The funds would either be sourced from public coffers or through a new mechanism such as the 'auction of assigned amount units' as recently proposed by the Norwegian Government (CCAP, 2009). The 'nested' approach is essentially a combination of the first two where compliance-market projects would still be allowed (and encouraged) within a national framework paid for through public funding.

Which option would be most appropriate for the realization of REDD in Sub-Saharan Africa in terms of securing long-term payment for activities? To date, the compliance market based approach has failed in terms of implementing land-use projects. Only eight CDM land-use projects have been registered globally, and only one of them is in Africa (World Bank, 2009). One of the main reasons for this is due to the relationship between risk and returns associated with land-use versus industrial emission reduction activities (Boyd et al. 2007, Myers 2007, Climate Focus 2008). Table 1 contains a basic comparison of the risk and returns for a CDM land-use versus industrial sector project. Due to low returns that are heavily discounted forward and high-perceived delivery and counterparty risk (Myers 2007, Cotula and Mayers 2009,) investors have shied away from land-use projects in SSA.

For an investment manager who has a mandate to reduce the emission reduction liability of a company at the lowest possible cost and risk, the 'rational' investor choice will continue to favor alternative industrial sector activities or straight-forward allowance trading. If the positive non-financial externalities (for example, water services, soil conservation, biodiversity) were to be monetized and realized through a payment for ecosystem service mechanism, land-use projects may yield better potential returns. However, perceived high risk will still inhibit the level of investment required.

Going forward, if land-use based mitigation (and adaptation) are to be realized at significant scale, there needs to be a fundamental change in the manner in which activities are financed. If REDD is to be implemented as currently proposed - through a phased national approach, with initial phases aimed at establishing required institutional and human capacity - a separate, dependable, non-compliance market source of funding is required (Johns et al. 2008). While the phased approach is appropriate and what is required in a SSA context, the initial set-up phase does not generate emission reductions and a tradable asset under current compliance regimes. Angelsen et al. (2009) go so far as to state that the 'market- vs. fund-based REDD debate perpetuates a false dichotomy' that ignores the reality of urgent short-term needs and conditions, which need to be put in place, but that are not compatible with the compliance market.

However, compliance market funding should not be completely negated. Individual compliance funded projects are obviously valuable within a nested REDD approach. Better early returns from REDD (as opposed to waiting for trees to grow), as well as lower permanence and leakage risks due to projects being nested within a national scale framework, may significantly improve investor appetite for AFOLU projects. Moreover, in future a new form of compliance regime may provide the opportunity to generate revenues to finance REDD over the long-term, for example, through a new GHG emission related taxation of a particular sector such as shipping or aviation (Angelsen et al. 2009).

Taking a fund approach should not be misinterpreted as a 'soft' money option. It ought not detract from the robustness of emission reductions and results. Emission reductions still need to be real, permanent, measurable, reportable and verifiable with the principles of efficiency and inclusiveness guiding strategic spending (Winkler 2008).

2. Considering challenges more specific to land-use development in sub-Saharan Africa

If one takes a step back and views the implementation of REDD and landscape scale adaptation and mitigation more broadly, it essentially encompasses the appropriate development of the forestry, agriculture and natural resource sectors in developing nations. Yet 'aid' and 'development' agencies with considerable resources have been trying to develop these sectors for the past 50 years without significant success (Cleaver and Donovan 1995, Cleaver 1996, Abegaz 2005). While the consideration of REDD under the UNFCCC provides renewed energy and the promise of new financial flows, many of the fundamental challenges that have hindered prior efforts still remain.

Here we explore three issues that have challenged prior forestry and agriculture development initiatives and that ought to be urgently addressed for the successful implementation of REDD, namely:

- Working within current land-tenure arrangements
- Creating appropriate incentive schemes
- Adequate governance throughout the supply chain of forest commodities

2.1 Working within current land-tenure arrangements

A lack of formal land tenure over large areas of the rural landscape in the majority of SSA nations has been highlighted as a roadblock not only to the large scale implementation of climate change mitigation and adaptation activities, but to the general economic and agricultural development of the region (Crittenden and Wilder 2008, FAO 2008). In the context of realizing project-scale CDM climate change mitigation activities, a lack of private and documented land title under constitutional law has been widely identified as a fundament risk to investment (FAO, 2008, Cotula and Mayers 2009, Angelsen et al. 2009). To address this issue, most proposed REDD frameworks include 'land tenure reform' as part of the progress towards successful national scale implementation (Cotula and Mayers 2009, Angelsen et al. 2009). The general consensus in the climate change policy literature and discourse is that land-tenure reform is a prerequisite for successful REDD implementation.

However, is this necessarily true? Does land-tenure have to be reformed or can REDD be successfully implemented within existing customary and constitutional law at relatively low risk? In addition, is land-tenure reform appropriate for the general development of SSA nations and do sub-Saharan African governments actually want to change land-tenure law? There is often an underlying assumption in REDD policy development that SSA citizens and governments would want to reform land tenure legislation, which may not necessarily be true. Lastly, can REDD implementation afford to wait for land-tenure reform to occur? Answering each of these questions comprehensively is far beyond the scope of this paper, but initial observations made through the process of developing land-use based mitigation on the ground in SSA indicate that 'conventional wisdom' may not hold true.

First, the western market economy consensus that the reform of land-tenure and the establishment of private land rights is necessary for economic development and resource management has been widely challenged (Atwood 1990, Ostrom 1990, Platteau 1995, Ostrom et al. 1999, Cotula and Mayers 2009, Place 2009) - Elinor Ostrom recently won the 2009 Noble Prize for Economics for her work challenging the conventional wisdom that privatized or regulated land is in general better managed than communal land. A comprehensive analysis of theoretical and empirical evidence from sub-Saharan Africa by Platteau (1995), shows that establishing private land tenure does not automatically lead to economic development, increased agricultural production or a decrease in environmental degradation. Depending on prevailing market forces, it may even lead to land being left idle or overexploited to maximize capital gains in a relatively short period. In addition, the negative social impacts of land reform can potentially be severe (Platteau 1995). The notion of having to reform land tenure prior to REDD implementation and its appropriateness in the context of SSA, may therefore need to revisited.

Secondly, even if land-tenure reform was preferable and there was the political will to undertake the process, the time-period required may simply be too long. For instance, despite strong political will and resources, the post-1994 South African Land Reform process has been in progress for 15 years and is still on-going due to the enormity of the bureaucratic task, the complexity of the issues at hand and the contested nature of land itself (James 2006). Negotiating the structure of land-tenure legislation and systematically identifying the rightful landowner of each parcel of land in a fair manner will take an extraordinary amount of time in any country. Whether the implementation of REDD can wait for this protracted process to be completed is questionable.

An alternative may be to work within the existing land tenure structures and to rather focus on securing resource tenure within the legislative frameworks that currently exist.

One option, especially in areas with sparse or no resident population, would be a concession approach as illustrated by tourism concessionaires in SSA to date. Based on an appropriate 30-, 50- or 100-year lease agreement with central government and proposed national REDD co-coordinating institutions, the concessionaire would undertake to manage an area of land in accordance with certain REDD principles and rules and any other conditions that may be appropriate with respect to the rights and needs of people on the land, or national development objectives. The GHG emission and stakeholder benefits could be audited by a national authority or an established international standard such as the Climate, Community and Biodiversity Alliance Standard (CCBA 2005). The concessionaire need not always be a well-capitalized external party but could be a local party as well. For example, the Government of Tanzania has successfully initiated a 'village forest reserve' program that facilitates the management of unreserved forest areas by local residents (Zahabu et al. 2007).

A second option, especially in 'communal' areas, is to take a small-scale grower approach. Good examples are demonstrated in the honey, tobacco, sugar-cane and plantation forestry industries in SSA. There are for example, 198,000 individual tobacco growers in SSA with 122,000 growers in Mozambique alone (www.universalcorp.com). Each tobacco grower cultivates 2-4 hectares of land with the company providing required inputs and expertise, and purchasing the crop at the end of the growing season. The tobacco company, which has been operating since the 1960's does not own land but successfully 'farms' and exports 206,000 tons of tobacco each year. In a similar small-scale grower / manager approach, the honey industry of North-Western Zambia consists of 20,000 individual beekeepers who together export 1,500 tons of honey a year (Mickels-Kokwe 2006). Again, the industry has been operating successfully for several decades without individual land-tenure under constitutional law.

If implemented within the context and the support of a national REDD program, a small-scale land manager approach may be adaptable and suitable for REDD and ecosystem service management implementation. The role of the national REDD program would be to facilitate and support a small-grower program operating at an informal, local scale. The success of implementation however, relies on:

- The recruitment of local residents who have access to resource tenure under customary law to manage forest resources. The value and flexibility of indigenous arrangements needs to be utilized, not necessarily replaced (Platteau 1995, Cotula and Mayers 2009).
- Taking into cognizance that resident growers will most probably not have access to banking and credit facilities and therefore payments would need to be on a short cycle to maintain adequate cash flow. Payment for the maintenance of a forest or rangeland area, predefined, could be made on an annual or biannual basis. The massive penetration of cellular telephones into SSA has also provided innovative ways of paying large numbers of people small amounts of money in a safe and effective way (Porteous 2006).
- Adequate payment that outweighs opportunity costs. In an open market-based system, the long-term permanence of any land-use option relies on a competitive rate being paid.
- The approach being nested within a supporting national scale policy and monitoring framework. Essentially the sustainable development, policies and measures (SD-PAM) approach as proposed by Winkler et al . (2007).

Both the concession and small-scale land manager options above, address many of the risks conventionally associated with a lack of formal land-tenure. If implemented within a national scale monitoring and accounting framework, leakage risk could be significantly reduced. In terms of delivery risk, due to the sizes of the concession investment, a typical concessionaire will most probably be an adequate counterparty with underlying assets in case of default. Again, national scale monitoring would allow for adequate surveillance of the concession to address potential deforestation early. Under the small-scale land manager model, delivery risk could be adequately diversified across a portfolio of several (hundred) thousand implementers undertaking different mitigation activities across several vegetation types, and climatic and fire regimes.

2.2 Creating appropriate incentive schemes

In addition to the discussion on potential sources of REDD funding above, a significant source of debate within climate change policy literature and discourse is how to appropriately compensate for REDD (Ebeling and Yasue 2008, Crittenden and Wilder 2008, Johns et al. 2008, Angelsen et al. 2009, Combes-Motel et al. 2009, Streck et al. 2009).

With regard to who to pay and how, as Knoke et al. (2008) note, local residents who utilized forest resources prior to REDD implementation may not be prepared to stand idle. Instead, 'a field of activities' are needed to keep people gainfully employed in productive rather than destructive activities (Knoke et al. 2008). As Cotula and Mayers (2009) note: "REDD simply will not work unless it is locally credible; it will be undermined and overthrown". It is therefore crucial to insure that true, real benefits flow to as many local residents as possible. In the development of such a 'field of activities' that are compatible with REDD implementation, valuable lessons can be learnt from successful models that have been running for a period of time in SSA.

Small-grower production systems provide an example of a private sector business model that may provide insights to REDD implementation. In addition, especially in the context of the Miombo woodlands of southern and Eastern Africa, there is a vast literature and a pool of experience in the development of community-based natural resource management (CBNRM) that needs to be tapped; for example, Frost (1996), Desanker et al. (1997), the special edition of Ecological Economics introduced by Campbell et al. (2000) and the volumes edited by Campbell (1996) and Kowero et al. (2003). CBNRM has developed through several decades of trial and error which REDD policy makers need to capitalize on. Initial lessons from the CBNRM field and the successful production of cash crops are:

- To create local ownership and equity through dealing with individuals on the ground,
- For payments to be made in cash to individuals,
- To add value to forest commodities locally before export, and
- To employ as many individuals as possible at a reasonable wage.

An expanded public works program may be appropriate for ecosystem service management and REDD implementation beyond the scope and capacity of the individual. As an example, the South African Extended Public Works Program was initiated in 1995 and includes a suite of environmental programs aimed at restoring and maintaining ecosystem services while developing skills and providing employment opportunities in rural areas (Turpie et al. 2008, www.epwp.gov.za). The suite of programs includes Working for Water, -Woodlands and –Wetlands and Working on Fire. Together they employ 26,000 people full-time and considerably more on a temporary basis (www.dwaf.gov.za/wfw/). Working for Water focuses on restoring and maintaining the water regulation services of watersheds through the removal of alien invasive plants. Similarly, Working for Wetlands is aimed at restoring wetland and watershed structure and function. Working for Woodlands aims to restore and maintain indigenous forest and rangeland systems, and Working on Fire implements integrated fire management at a national scale.

Such a suite of programs, which could potentially be implemented throughout SSA, would provide an overarching level of support for individual small-scale land managers as well as concessionaires while creating a considerable number of long-term employment opportunities. It provides a mechanism to implement REDD while including 'landscape adaptation and mitigation' as advocated in current climate change policy discussions.

2.3 Adequate governance throughout the supply chain of forest commodities

The majority of SSA nations often have adequate forest governance legislation in place to govern timber concessions. Typically a timber concessionaire is legally obliged to harvest selected species at sustainable cropping rates and adhere to the principles of the Forestry Stewardship Council (FSC, www.fsc.org). Unfortunately, in practice this is not always the case. The important issue is not necessarily a lack of legislation, but a lack of adequate capacity to enforce such legislation (Cotula and Mayers 2009). This situation is particularly true in countries that have emerged from periods of conflict where the current tax-base is insufficient to fund the policing of logging and charcoal operations.

On the positive side however, timber and charcoal are literally big commodities - they can't be smuggled in a suitcase. To reach market in economically viable quantities, timber and charcoal needs to be transported on main arterial roads, large rivers and through deep-water ports. There are therefore crucial bottlenecks in the supply chain where enforcement agencies could cost-efficiently and effectively control the flow of illegally harvested forest commodities.

A potential 'quick-win' early in the process towards comprehensive REDD implementation, is to finance such governance and policing capacity throughout the supply-chain of forest commodities. It would need to be implemented by national government due to sovereignty concerns and audited to ensure funds are being spent efficiently, but it could be implemented relatively quickly while comprehensive, national-scale REDD programs are developed.

Yet if reforestation and forest degradation is to be tackled over the long-term, adequate governance and enforcement is required throughout the supply chain of forest commodities. Efforts to address deforestation may not only occur on the supply-side of the equation, but on the demand-side as well. Insatiable demand for cheap, unprocessed, uncertified forest products is one of the main drivers of deforestation and degradation in SSA. If entities on the demand side of the equation only purchased forest commodities (timber, charcoal) from forests that have been managed at sustainable cropping rates and through the principles of the FSC, deforestation and degradation would be dramatically reduced. Like narcotics or ivory, only addressing the supply-side may not work and lead to international leakage. Supplies will merely shift, taking advantage of poorly resourced governments and peoples.

Conclusion

The reader may note that the fundamental challenges to the implementation of REDD in SSA revolve more around development studies and emerging nation economics than the natural science disciplines traditionally associated with forest management. While international policy, efficient monitoring and knowledge of forest ecology are crucial and in need of substantial further development, the main unresolved issues relate to permanence and liability (Myer 2007). These issues presented a roadblock to CDM land-use implementation in SSA and could do so for REDD unless prioritized and resolved properly.

Moreover, the solutions to permanence and liability may lie in placing more attention on understanding social and economic complexities operating at a local-scale rather than at a national or regional scale. Bottom-up approaches of understanding the local context before developing REDD activities, of developing alternative livelihoods and compensation may be more appropriate.

When developing bottom-up activities however, caution should be taken to not waste resources in reinventing the wheel. As noted above, CBNRM advocates have been investigating appropriate incentive schemes for several decades. Development anthropology and economics have been focusing on land-tenure and communal governance issues since the 1960's (Ostrom 1990, Platteau, 1995). While the final solutions may not exist, considerable time and error can be saved through reviewing and assimilating past efforts.

Of utmost importance for realization of REDD in SSA, is the rate at which implementation will happen (Koenig 2008). As numerous nations emerge from lengthy periods of conflict and inaccessibility, the forests and woodlands of region are now accessible to the negative effects of international market demand. It is a case of either taking the opportunity to implement REDD in the short-term or losing it.

Table 1: An elementary comparison of the risk and returns for Clean Development Mechanism land-use versus industrial sector activities

Variable	Land-use sector	Energy and Industrial sector
Returns		
Financial returns		
Income	Low returns per unit area and low comparative returns on total investment	High, especially for HFC emission reduction activities
Costs	High transactions costs due to monitoring on landscape scales and engagement with a large number of stakeholders	Relatively low. Generally low maintenance and monitoring costs after project initiation
Timing of payment	Once every 5 years, therefore heavily discounted forward	Annual
Non-financial returns		
Climate change adaptation	High, an avenue through which to realize advocated “landscape adaptation”	Nil*
Ecosystem services restoration and maintenance	High	Nil*
Biodiversity	Potentially high	Nil*
Rural livelihood benefits in terms of employment, energy, water and agriculture (food)	Potential good to high	Low
Risk		
Technology risk	Moderate - sequestration rates (returns) are dependent on rainfall, fire and pests	Low, usually the technology is based on known and proven engineering principles
Delivery risk	Moderate – risk from fire, illegally logging and clearing	Low
Counterparty risk	Potentially high – especially as a small project scale where the counterparty lives in poverty and has no assets as collateral	Low, the counterparty is usually a registered entity with an underlying business and assets

* Nil in situ. It can however be argued that reducing GHG emissions reduces anthropogenic climate change and therefore benefits the long-term conservation of biodiversity and the maintenance of ecosystem services.

Chapter 3

Preparing community forestry for REDD+: Engaging local communities in the mapping and MRV[4] requirements of REDD+

This chapter has been published in: *UNEP RISO Perspectives. Series 2010*
Can carbon markets promote REDD+ activities in Developing countries: Section 1.

Authors: Tony Knowles, Michael McCall, Margaret Skutsch and Leon Theron

As parties started to develop project-scale REDD activities 3-4 years ago, one of the needs that emerged is how to cost-effectively monitor carbon stocks across vast landscapes in a robust manner and with the required level of accuracy. Although advances in remote sensing may reduce the need for ground-based surveys to a certain degree, field surveys are still required to ground-truth estimated carbon stocks using satellite imagery. This is especially true in the heterogeneous woodland and rangeland systems of East and southern Africa. In addition to measurements of biomass carbon stocks, numerous additional socio-economic and biodiversity metrics need to be measured which require substantial time and cannot be estimated using satellite imagery. To reduce the cost of employing more expensive consultants, the notion of community-based monitoring emerged as a means of completing required field surveys in a cost-effective manner while providing a local employment and skill-development opportunity.

The authors were requested to write the chapter based on their hands-on experience in developing community-based monitoring programs. Following early exposure to projects located in South Africa and Zambia, Leon Theron and I developed a monitoring program for the Tayna-Kisimba REDD project located in the eastern Democratic Republic of Congo that is being facilitated by Conservation International.

[4] MRV is an acronym for monitoring, reporting and verification

In a similar manner, Margaret Skutsch and Mike McCall have been involved in the development of community-based monitoring programs in Tanzania.

The chapter reviews the appropriateness of community-based forest carbon monitoring programs in terms of cost-effectiveness, creating local value and employment opportunities, and meeting local and international monitoring, reporting and verification (MRV) requirements.

Preparing community forestry for REDD+: Engaging local communities in the mapping and MRV requirements of REDD+

Tony Knowles, Michael McCall, Margaret Skutsch, Leon Theron

1. Introduction

In her plenary speech at the UNFCCC COP 15 in Copenhagen, Elinor Ostrom highlighted the importance of creating clear local livelihood incentives to ensure the sustainable management of forests and woodland resources. Citing meta-analysis studies, for example Chhatre and Agrawal (2009), Ostrom noted that local monitoring, management and control of forest and woodland resources create a sense of local ownership and value in forests that is crucial to their long-term acceptance and sustainability term.

In this chapter we explore the notion of community-based forest carbon monitoring (CBFM) as means of creating local employment opportunities and value in forests. We first focus on the ability of CBFM to meet project- and national-scale monitoring, reporting and verification (MRV) requirements. Then a comparison is made of the advantages and disadvantages of forest monitoring undertaken by local residents compared to external consultants. We conclude with a discussion of the livelihood benefits and skill-development opportunities created through CBFM, as well as its potential to reduce the transaction costs of landuse-based climate change mitigation and adaptation activities.

1.1. Objective

The objective of this paper is to review the experiences of local community people's involvement in MRV-type activities in REDD+ projects, or in community carbon forestry in general. There are not as yet many examples of communities being actively involved in REDD+ MRV, so we refer also to experiences and findings coming from other community carbon forestry initiatives, or 'traditional' community forestry projects.

The aim is to give a practical assessment of the capacities of communities to become involved in MRV, and what recommendations to develop these capacities. Equally important, we look at the potential benefits for communities, and what real interest communities themselves might have for involvement in MRV for REDD+.

1.2. Context: What are the likely future scenarios of REDD+

Although there is still much uncertainty about the form REDD+ will take, in this paper we assume it will be at a sector level across whole countries, or (for large or physically disjointed countries) sectorally across major administrative regions, rather than as individual projects. The technical reason for this is to avoid leakage from treated areas to untreated areas, but there are political reasons too. CDM project experiences have been rather negative in many developing countries, and negotiations at the UNFCCC level reflect that many countries would prefer to retain control over the REDD process rather than release it to project developers. Thus, most of the 35 REDD Readiness proposals submitted to the World Bank's Forest Carbon Partnership Facility are committed to a national REDD approach; although given the difficulties of starting-up a national approach, a likely scenario is that programmes would begin with pilot projects within a country.

This paper also assumes that REDD+ will incorporate measures for enhancing removals of atmospheric carbon dioxide as well as reducing emissions. The detailed policy discussion on REDD+ at COP15 (UNFCCC, 2009) strongly suggest that incentives could be offered for emission reductions from lowered national deforestation and degradation rates, and also for increases in forest carbon stock (forest enhancement), and in addition, some kind of compensation for conservation of forest that is intact, although how these carbon savings will be rewarded is not clear (UNFCCC, 2009; RECOFTC, 2010).

Another uncertainty revolves around where the money for carbon credits will come from, and whether the carbon credits can be used for off-set or not, in other words, whether there will be a carbon market or a carbon fund. However, from the point of view of developing countries this distinction may not be so important, as the conditionalities are likely to be equally restrictive in both spheres. More important will be the size of the demand for credits, since this will fix the price at which carbon can be sold.

2. Information Needs in REDD+

2.1. MRV (monitoring, reporting and verification) requirements under probable REDD+ scenarios

When REDD+ will be run as a sectoral, national level programme rather than at project level, this raises immediate difficulties for MRV, since what has to be measured includes not only the areas subject to special treatment under REDD+, but all forest areas within the national territory. Moreover, individual REDD-like projects operating in the past, whether financed in the voluntary carbon market, or simply to combat deforestation as in PES (payment for environmental services) projects, have used different monitoring and reward systems. A national REDD+ programme would have to set standard procedures both for data gathering and the payments, which will require a major effort in public administration.

Measurement of change in national forest area (to detect changing rates of deforestation) can be carried out reasonably easily and cheaply through remote sensing. However, quantifying changes in density of biomass (i.e. of the carbon stock) in the forests is much more difficult, and cannot currently be done by remote sensing, although that may be helpful in identifying areas of forest disturbances. Measuring changes in forest density is essential if countries are going to claim not just reduced deforestation, but also for reduced degradation, forest enhancement and sustainable forest management. In many countries the majority of losses and gains in forest carbon will fall into these categories, rather than deforestation. Therefore the ability to gather reliable data on forest density change may be the key to their participation in REDD+. It is essential for these countries to find cheap, reliable methods for establishing rates of degradation and forest growth, which includes setting a baseline or reference level for these processes.

Different measurement methods produce data at different level of accuracy (Tiers 1 to 3 in IPCC terms), and it may be assumed that when a low level of accuracy is implied, a greater portion of the estimated carbon savings will be discounted from the crediting, on a principle of financial conservatism. A method which produces data of greater accuracy (i.e. small standard error) may in principle generate more confidence in the results, and hence leverage rewards in terms of eligibility for a higher proportion of the estimated carbon savings. (Wise et al. 2009) For a national REDD programme this presents a trade-off; the additional costs of increased accuracy of estimates of forest density changes, versus the financial benefits of the additional carbon credits that could be generated. The parameters for these calculations have not yet been defined (nor is the likely price of carbon credits known, and the margins for conservatism have not been set), but it is very evident that governments will look for methods that generate maximum accuracy at minimum cost.

2.2. Information and meta-data requirements at international level

The REDD+ concept adds carbon sequestration (due to good forest management) to the avoided deforestation and degradation envisaged by the REDD. Protocols for REDD have yet to be developed, but to make claims internationally a Reference Emissions Level (REL) will have to be drawn up by each country and approved by an international body. As with baselines for CDM projects, it is likely that several methodologies will be made available to meet different circumstances. Countries wishing to claim for reduced degradation and forest enhancement will have to provide credible RELs for this; no easy task since few have much data on changing forest density over time. Countries will have to declare what monitoring methodologies they are using; this is essential for transparency of data origins. Probably they also must declare their internal verification methodology, and independent, external verification will also be implemented. Table 1 gives the likely capacity requirements.

Table 1 . Capacities likely to be required for national REDD programmes

Measurement of Carbon	Site level field data on carbon stock changes to indicate: • Reduced deforestation and degradation compared to local baseline/REL • Increases in carbon stock – sequestration/ forest enhancement
Scales of functioning	• Measurement skills at community scale • Capability to upscale to region and country • Accessible central database for uploading data from decentralised projects
Verification and certification	• Transparent and independent verification system • Information availability from independent sources • Leakage estimated and included
Management Capability	Leadership; participation, resource rules enforcement • At local level • At national level
Acceptability	Acceptability to states, regions and local communities of the management and participation rules
Externalities	Damaging effects to environment and society and e.g. equity are dealt with at an acceptable level

2.3. Information and meta-data likely required at national and community level

For communities to credit and register the carbon sequestered in their forest, two levels of accurate and geo-referenced information is required for REDD+ .Firstly, a meso or 'landscape' level that involves information at 'community' scale to establish the initial forest management scenario (year 0). The second level is a more intensive collection of detailed plot level information (biomass sample plots, within management strata), and some at tree level. Accurate data on the size and location of every measurable tree and bush in sample plots are required for monitoring and to facilitate re-measurement in subsequent years (see Table 2).

Table 2. Information for Community Forest Management and Carbon Sequestration

Level 1: Spatial and other Information for establishing the initial management scenario (year 0)

- Boundaries of the community, and of its forest areas intended for a carbon payments project.
- Community's land claims - if necessary
- Community Forestry Management Approaches, land use plans
- Location of activities contributing to forest degradation such as: illegal logging, grazing, marginal agriculture, illegal settlements
- Location of areas potentially affected by hazards (e.g. fires, erosion, ecosystem degradation, flooding, strong winds)
- Conflicts (spatial information about land use, boundaries)

Level 2: Information for forest biomass inventories (year 0 and later)

- Delimitation of forest ecotype strata (zones)
- Location and geo-referencing of the sampling plots for measuring different carbon pools
- Geo-reference trees and other features for future location of the sample plots.
- Field measuring and storing of tree data: DBH (diameter at breast height), tree heights, species, status, etc. in databases

For communities, particularly in developing countries, finding most of the forest management information in Table 2 is not easy. Non-existence, unavailability or inadequacy of forest information, lack of technical knowhow, and deficient support from government institutions to produce or handle information are drawbacks normally faced. Therefore, for measuring and monitoring their forest resources, communities are liable to be dependent on external professionals, whose services would claim a big share of any feasible income from carbon sequestration payments.

2.4. Monitoring additional variables complementary to improved land management.

The majority of REDD MRV and associated policy focuses on monitoring carbon stocks and fluxes. While assessing carbon stocks is crucial, it is only one of several components that need to be measured in developing REDD activities. There are additional metrics that need to be monitored under current project-scale standards, such as the Voluntary Carbon Standard and Climate, Community and Biodiversity Standard. These are quite likely to be included in future national-scale REDD+ frameworks:

- Socio-economic information – documentation of stakeholders, social governance structures, household income, energy use, food and cash crops production, and benefit flows,
- Quantification and explanation of land use and land use change - understanding the nature of deforestation drivers so as to model deforestation scenarios and develop appropriate responses. Monitoring changes in land-use types and accessibility (especially roads) is thus an additional requirement.
- Biodiversity – the majority of REDD verification standards require quantification of project effects on biodiversity, especially threatened species; thus a need for adequate assessment of biodiversity, including in the forest surroundings.

3. The capacity and potential of community-based forest monitoring to meet information needs

3.1. Comparative advantages and benefits of community-based monitoring

Required skill-sets and expertise

Most activities related to carbon monitoring are regarded as technically highly-demanding and therefore in the realm of professionals. Experiences in Africa and Asia however show that with adequate training, key activities such as forest inventories, assessing and measuring forest resources, tree measuring and quantification of the current carbon stock and changes can be performed by the local residents, as demonstrated by the K:TGAL programme (Skutsch, ed. 2010).

Under K:TGAL, a 'participatory geographical information system (PGIS)' was implemented in which local communities become conversant with the use of IT for carbon forest data capturing and geo-referencing. When trained, local people were able to map forest reserves rapidly and with precision, locate permanent sampling plots with accuracy, and record measurement data for trees and other vegetation in the plots. Local residents with a basic level of education could be trained in approximately two weeks to conduct surveys. If they have used cell phones before, it can go faster, because of their similar handling characteristics to GPSs and handheld computers. A basic premise is that even people unfamiliar with computers (including illiterates and innumerate) can learn to follow the inventory protocols used by professionals, if suitable translation methodologies are developed and made available. The resulting data in several tested areas were within the desired levels of precision and reliability of that produced by professionals (Box 1; Zahabu 2008; Skutsch (ed.) 2010).

Interest in CBFM has encouraged the emergence of new techniques and technologies to improve the collection of data. Hand-held electronic data-entry hardware (PDAs) and software products have been developed as field equipment particularly for CBFM which reduce sampling error and loss of data. The hardware unit, typically including a GPS receiver, records location and vegetation metrics in predefined fields to improve data quality and completeness. The data are then uploaded into a database system that automatically verifies them and flags potentially incorrect values for the attention of the field team. The Tropical Ecology Assessment and Monitoring Network (www.teamnetwork.org) is an example. Data are either uploaded from a PDA unit or entered into a predefined spreadsheet that ensures recorded values and metrics are correct. Data are systematically backed-up and additional analysis can be performed immediately.

Local residents often have good basic knowledge of local tree species, the distribution of species, forest products, and an understanding of the local ecology. Moreover, residents have good understanding of local logistics of access, permissions, role players and the local and traditional authorities. In comparison, external consultants have to go through lengthy introduction and permission processes, as it is usually frowned upon to simply start working communal or private lands. Permissions can absorb much valuable monitoring time, and even then, local residents are often suspicious of outsiders. External consultants therefore anyway require a local facilitator to manage logistics, deal with permissions and communicate with local residents. Knowledge of local languages and traditional structures is especially advantageous in monitoring the non-carbon metrics. Stakeholder engagement and the monitoring of socio-economic metrics take considerable care and time requiring many engagements with individuals and groups. In such cases, external consultants may be prohibitively expensive as well as impractical.

Cost efficiency

The financial viability of REDD ventures relies on cost-efficient monitoring of carbon stocks across landscapes. While advances in remote sensing technologies allow carbon stocks to be measured with a reasonably high level of accuracy through satellite-borne sensors, experiences in assessing carbon stocks in the tropical forests of the eastern DRC and miombo woodlands of Zambia, indicate that the cost of the advanced imagery and processing is often more than the monetary value of the change in carbon stocks.

Relatively inexpensive remote sensing data such as MODIS or LANDSAT imagery have therefore been used until now to assess forest cover for REDD projects. Although significant progress has been made in the processing and analysis of MODIS and LANDSAT data, intensive ground-truthing is still required to estimate carbon stocks to an adequate level of certainty, as well as to verify alternative land-uses such as food and cash-crop farming, pasture, roads and exotic plantations (Trodd and Dougill 1998). In addition, ground-truthing is required over vast, inaccessible areas demanding lengthy time investments. Empowering local people to undertake such monitoring, as opposed to external consultants, saves significant costs, and maybe also time, when the additional flights, local mobility and accommodation for external consultants is factored in.

Concerning the precision of carbon estimates, there is always the possibility of sampling errors creeping in, whether residents or consultants undertake them. While monitoring effort should be proportional to the improvement in the certainty of the estimate (Wise et al. 2009), based on the central limit theorem, it is generally prudent to invest in increasing the number of replicates per stratum rather than the accuracy of each sample in order to increase the precision per unit cost. The number of replicates, not necessarily the accuracy of each replicate, is key.

Since local wages for field staff are typically a fraction of external consultant costs, by incorporating local people it is economically feasible to undertake many more replicates (plot measurements) in each stratum, thereby reducing the variance in the carbon estimates of each stratum. This is often crucial to project viability, especially in woodland and savanna systems (Wise et al. 2009). (Box 1)

Required capacity for national-scale REDD

In practical terms of capacity, there may not be enough external consultants to undertake the number of carbon, socio-economic and biodiversity surveys required for national REDD+ initiatives (Burgess et al. 2010). Although monitoring procedures, techniques and technologies are becoming more efficient, it will require a vast number of monitors to cover all the metrics across all countries entering REDD+. Moreover, it is proposed that REDD projects have a lifespan of at least 20 years, the Noell Kempff Mercado Project in Bolivia for example is planned for 99 years. Forest monitoring and verification need to cover the project lifespan. It is efficient rational therefore for local residents to undertake the monitoring, and utilise external consultants for the necessary verification.

Hurdles to community-based forest carbon monitoring

In countries with entrenched bureaucratic governance, officials may balk at handing over a function to local residents that is traditionally seen as a government responsibility and prerogative. Moreover, currently some REDD projects are being developed by external NGOs unfamiliar with local government structures. This can result in parties being left out of the project development and officials feeling alienated and buying-into the CBFM concept.

Where there is funding to contract-in external monitoring consultants, it is common practice in certain countries to take on local officials to act as national facilitators. These posts are often paid at lucrative international rates. When local community monitoring is proposed in place of external consultants, some officials may be apprehensive as it could jeopardise future facilitation contracts.

Management of monitoring teams is a key issue. In remote areas people often do not have access to the internet and there is limited accessibility. This means it is often infeasible to remotely manage monitoring teams from say, the capital city or a foreign-based NGO. It is highly recommendable that teams have a permanent local manager, especially a CBO or local NGO, who can be readily contacted for reporting and who will deal with technical matters.

Table 3. Comparison of monitoring components - undertaken by external consultants or by local residents' teams

Monitoring component	External Consultants	Local Residents
Cost	High professional fees, travel and accommodation costs	High initial setup and training costs followed by substantially lower salary, travel, accommodation costs over time
Local knowledge	Usually poor. Local guides and translators usually needed	Good. Residents typically know the area well – for access, logistics, local authorities, laws, and species names
Data quality	Good	Good, but dependent on appropriate training and data verification
Consistency	Potentially low if same consultants cannot continue with monitoring over lifespan of project	Potentially high if same team members or at least the same coordinators can be maintained
Intensity	Usually low. Too costly to spend long periods in field.	Good. Even if sampling is done part-time, substantial travel and setup time is saved
Value addition	Low. Usually limited to technical input and PDD compilation	High. Project success depends on local resource users. Monitoring by locals creates ownership..
Spin-offs	Maybe for consultants' business, not for community.	Participation adds to the skills levels and capacity of local residents. Possible spin-offs to other community PES activities
Management	Expected to be Good	Potential area of concern in many communities.
Logistics	Consultants' flights, vehicles and accommodation costs are high. In remote areas, costs escalate when vehicles are needed.	Locally-organised is cheaper, and more appropriate, e.g. by foot or animal can be effective because field surveys are spread over time.
Initial inputs e.g. time	Low. Assumption is that professional teams need relatively little preparation time	High. Could take considerable amount of time to identify, train and equip teams
Collection of other important data e.g. socio-economic information	Generally Poor. Very challenging to understand local socio-economy and culture, time-consuming to collect the data.	Good. In-built knowledge of local economy and culture; easy to collect initial information and monitor changes

Resistance to local monitoring may come from within the community as they can view it as restricting their use of natural resources. To avoid this situation, the REDD+ process needs to be open and transparent right from the onset through a public participation process. However, this itself will not ensure long term buy-in from local residents. Community-based participatory projects by governments and NGOs are common today in rural areas and there is an increasing reluctance by local people to become involved. Participation is time-consuming, and more importantly these projects have often brought no real, lasting benefits. For REDD+ to succeed, there have to be long-term financial, material and empowerment benefits for local residents. Employing community residents as monitors is one way of reaching this.

4. Creating value in forests for local communities

Creating rural employment opportunities and income flows

A key component of the long-term acceptance and sustainability of REDD initiatives is the creation of a sense of local ownership and value in intact forest and woodland systems (Chhatre and Agrawal 2009). Moreover, income flows and value added by REDD+ activities need to be greater than the foregone opportunity costs of deforesting or degrading the area for timber, charcoal or agriculture.

Clear, substantive incentives, such as employment, direct cash incentives, sustainable livelihood opportunities, and community development projects are essential to ensure the appropriate management of forest resources over the long-term. Experience in REDD-type activities in sub-Saharan Africa has shown that creating real incentives in the form of alternative livelihoods and equitable sharing of benefits is particularly challenging in rural areas remote from communications and markets, and thus from employment opportunities.

The sound forest management practices needed for REDD+ sustainability demand much labour employment from the community, - for fire prevention, livestock grazing controls, water and soil erosion management, defence against illegal felling and encroachments, and so on. The PES model is that the true costs of these are accounted for and compensated within the carbon payment levels. Moreover the forest carbon monitoring presents an additional opportunity to create employment and income flows. While the monitoring only takes a portion of the year, the trained team can also be involved in the surveillance, fire control and other management tasks.

Local skills development and creating the human and institutional capacity for national scale REDD+ implementation

In impoverished rural areas, formal educational levels are low, thus training people as REDD monitors adds a set of life skills that potentially spill over to other spheres. The involvement can lead to social and institutional strengthening in the community. There is not only an expanded understanding of local natural resource management and values, but the skills such as data capture and mapping empower people, and bring at least some into decision-making processes. When the skills are developed and retained, or passed on, and new technical knowledge is acquired, then, importantly, the ability to deal with strong government agencies or NGOs is greatly strengthened. There is great potential for utilising the participatory survey and mapping skills for other community purposes, such as making land claims, resolving land conflicts, collaborative land use planning, and applying for PES finance such as hydrological or biodiversity services.

Ability to respond from an empowered, informed position to REDD+ developments

Currently, local communities have little say in REDD developments, due to limited policy knowledge and institutional support. Through participation in monitoring and mapping activities, residents become more exposed to the operations of international climate change policy, REDD mechanisms, and payments for climate change mitigation. This increased awareness allows people to respond to national REDD agencies and the global carbon project developers from a more informed and confident position. Moreover, it may allow communities to engage with REDD+ initiatives with ‘free, prior and informed consent’.

Lower transaction costs are essential

Cost efficiency and reducing the transaction costs of crediting carbon is always crucial to the financial feasibility, but its importance is amplified where the volume of emission reduction units produced is relatively small, whether this is because of the small size of the management units or because the carbon stock growth rate is relatively low (Cacho et al. 2004). Reducing the monitoring costs is crucial to the financial viability especially in small community projects in dry forest and savanna areas, for example the miombo woodlands of sub-Saharan Africa,

Community-based monitoring greatly reduces the transaction costs of monitoring and management as the operational costs are a fraction of those of external professionals. (Box 1). The key is to develop the protocols, mechanisms and associated training, so that community residents can perform monitoring and reporting with sufficient accuracy and reliability to be acceptable in a formal carbon finance mechanism.

5. Conclusions

Community management is currently being promoted in many national REDD programmes, but few consider the full implications for the range of data collection. While changes in forest area (relating to deforestation) can be measured relatively accurately using remote sensing, the changes in forest density - related to reduced degradation and forest enhancement – must make use of decentralised field measurements. We have argued above, for many reasons, communities already involved in management of their forests should be empowered and mobilised to carry this out. Some authorities may view this idea with scepticism, but our and other's experiences show that given a clear protocol and appropriate training, communities are able to gather data as accurately as professionals, and at a fraction of the cost.

The protocols used, such as those in the K:TGAL programme, need to be based on internationally accepted methodology to ensure confidence in the data, and independent verification will be essential. Detailed guides are available (Bhishma et al 2010, Theron 2009, Verplanke and Zahabu, 2010), and see also van Laake et al. (2009) and Peters and McCall (2010),

´What is in it for communities?` depends crucially on the financial margins that communities would receive as a result of their participation in REDD activities, and how these are distributed. At present it is very difficult to estimate either the market price of REDD carbon or the transaction costs that will be incurred in running national programmes. These issues will become more visible as national programmes get started. However, the low cost and high effectiveness of community monitoring offers the hope that communities will one day be able to measure and sell credits for the increases in carbon in their forests as another ´non-timber forest product´.

Box 1: Costs and reliability of community versus professional carbon monitoring

In the K:TGAL programme, costs of community measurements were made at several sites, including the costs of training by an intermediary organisation, and a daily wage rate for the community members undertaking the forest inventory. These are compared with costs of professionals in two sites. The variations in cost between communities reflect variations in size of forests (considerable economies of scale), their accessibility (to the intermediary organisation), and their carbon productivity. The costs of professional measurement appear to be at least double the costs for local measurement.

Location	Cost per hectare, community ($)	Cost per tonne carbon, community ($)	Cost per hectare, professional ($)	Cost per tonne, professional ($)
Nepal site 1	2.4	0.2		
Nepal site 2	4.7	1.5		
Nepal site 3	5.1	2.5		
Tanzania	3.1	2.3	10.0	7.4
Uttarakhand (India)	5.4	0.8	11.0	1.6
Papua New Guinea	3.8	0.4		

Comparing the results of professional and community inventories in the Tanzanian and Indian sites indicates that the two sets of estimates are similarly reliable; there were no statistically significant differences in their estimates of mean biomass.
Both the professionals and the communities measured diameter at breast height and height of trees, identified the species, and applied the relevant allometric equations to reach the estimate of above ground woody biomass. (from Skutsch (ed.) 2010)

Postscript

Although Chapter 3 has been published in a peer-reviewed publication, the examiners of this dissertation provided further insights that I believe should be included here. Some of the insights pertain to issues beyond the strict topic of 'community-based forest carbon monitoring', but they do remain interesting and pertinent in the greater context of considering the feasibility of REDD activities in sub-Saharan Africa.

Encouraging effective self-enforcement through the adoption of a broader set of community activities

One of the principle challenges in the development of REDD activities is the creation of local-value in intact forests. As Ostrom (2007) reports, creating local-value and buy-in generally leads to effective self enforcement and the sustainable use of the resources over the long-term. It is therefore imperative to take a local community-based approach to management activities were possible. Such activities should extend beyond monitoring and reporting to include policing, enforcement, and forest and fire management.

Considering the structure of incentives and benefit flows

The chapter reported that there is an increasing reluctance by local people to become involved in projects as they are viewed as time-consuming and often have few real long-term benefits. In addition, the promise of future awards from carbon sales often has little merit with local residents who may live on a day-to-day subsistence basis.

A means of addressing this need for meaningful benefits is to provide communities with an equity stake in the underlying venture and an annual dividend. This structure allows more immediate and regular income from the underlying business (for example, sustainable charcoal production or eco-tourism), while carbon revenues are realized once every five years. Such a financial structure is more likely to lead to real buy-in and the protection of the asset base.

Required institutional capacity to support decentralized MRV

The decentralization of natural resource activities is not a new concept. Especially within the field of community-based natural resource management (CBNRM), there is considerable experience in implementing decentralized approaches that should be capitalized on. In particular, experience has shown that there needs to be equal emphasis and investment in supporting capacity in addition to the field activities in the field. Such a need to create centralized supporting capacity is often overlooked leading to the failure or collapse of activities.

Chapter 4

The effect of transaction costs and economies of scale on the financial feasibility of land use climate change mitigation ventures in Africa

One of the first questions one is asked when presenting to an audience of potential REDD implementers, be it an academic forum, a delegation of Government officials or a group of wine farmers, is: How much is it going to cost and what returns can realistically be expected?

Despite the apparent simplicity of the question, few publications exist on the transaction costs of realizing the carbon assets of a REDD + activity and expected returns. Furthermore, little readily available information exists on the nature of transaction costs. For example: how do transaction costs scale relative to project area and to returns; where are there opportunities to reduce transaction costs; and how large does a project need to be, in terms of spatial scale or the number of generated emissions reduction per year, to be financially viable? Such knowledge is not only of academic interest and use to REDD project implementers, but to policy makers and strategists attempting to compare the risk and returns of land use based climate change mitigation against industrial or energy sector options.

This paper seeks to address this need by reviewing the costs required to 'take carbon to market'. This includes the cost of monitoring changes in carbon stocks and other biodiversity and socio-economic metrics, the compilation of required project documentation, local- and Government stakeholder engagement, as well as verification, legal and brokerage fees.

The need and idea for the paper was formed through discussions with Dr Bob Scholes. Following the completion of the analysis and first draft, Dr Albert van Jaarsveld, Dr Allan Ellis and Dr Bob Scholes provided valuable input that significantly improved the draft text and presentation of results. Dr Russell Wise is thanked for explaining the basics of cash flow discounting and financial feasibility analysis to me.

The effect of transaction costs and economies of scale on the financial feasibility of land use climate change mitigation ventures in Africa

Tony Knowles, Robert. J. Scholes, Albert. S. van Jaarsveld and Allan Ellis

Abstract

As the world struggles to address deforestation and land degradation in response to global climate change, one of the important emerging issues is how much such interventions are likely to cost. Despite the obvious climate, socio-economic and biodiversity benefits associated with land-use based climate change mitigation activities, comparatively few interventions have been implemented to date. Concerns regarding non-permanence, leakage as well as high transaction costs and a lack of certainty regarding financial returns have inhibited the required investment.

We assess the additional transaction costs incurred through creating carbon assets associated with either afforestation and reforestation (AR) ventures or activities aimed at reducing emissions from deforestation and forest degradation (REDD). A representative set of African woodland and rangeland systems are used to assess the effect of transaction costs, vegetation type and economies of scale on expected carbon revenues. Emphasis is placed on considering all transaction costs, including those incurred through monitoring forest carbon stocks, scenario development, project documentation, verification, the trade of emission reduction units, and engagement with national government and local stakeholders. As the value of aligning REDD implementation with national priorities and the inclusion of indigenous peoples rights emerges in the development of post-2012 climate change policy, particular importance is placed on including the cost of adequate national government and local stakeholder engagement.

Results indicate that the two key issues controlling expected carbon revenues generated by both AR and REDD activities are ecosystem type and economies of scale. Estimated transaction costs range from US$ 600,000 to 720,000 depending on the scale of the venture. Activities need to be in the order of 1,500 to 12,500 hectares

for the revenues to at least equal transaction costs, with the lower size being in systems that are more productive. The expected returns for AR ventures are closely related to the net primary productivity of the ecosystem, which in the systems studied is correlated with rainfall.

From a policy perspective, the results provide support for a shift to larger scales: from project- to national-scale; and the creation of in-country capacity at a national scale through non-market funded platforms. If REDD and AR are to be widely implemented as the preferred rational economic land-use choice at a significant scale, new financial mechanisms need to be developed that would reduce high upfront transaction costs and the time-interval before income streams are realized.

1. Introduction

The findings of the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report create a strong case for urgently slowing deforestation and land degradation across the globe (IPCC, 2007). Land-use change accounts for a fifth of anthropogenic Greenhouse Gas (GHG) emissions. The restoration and maintenance of natural ecosystems is a step towards climate change adaptation due to the close link between ecosystem services, human livelihoods and economies (Scholes and Biggs, 2004; Clewell and Aronson, 2006; Boko et al., 2007; Fischlin et al., 2007). If implemented appropriately, ecosystem protection and remediation for climate change mitigation purposes could generate revenues for the conservation of important biodiversity areas (Bekessy and Wintle, 2008; Ebeling and Yasue, 2008).

Based on this logic, much effort is currently being invested in activities aimed at avoiding deforestation and land degradation activities; technically referred to as Reduced Emissions from Deforestation and Forest Degradation or REDD. Substantial progress has been made towards the development of REDD implementation options (Chomitz, 2006; Angelsen et al., 2009; Streck et al., 2009) as well as the creation of REDD readiness capacity through multilateral platforms such as UN-REDD and the World Bank Forest Carbon Partnership Facility (www.un-redd.org).

Afforestation and Reforestation (AR) have a slightly longer history, since they were mechanisms defined under Article 3.3 of the Kyoto Protocol. But while in principle developing countries could implement Clean Development Mechanism (CDM) projects relating to AR, very few have done so (World Bank, 2009).

Understanding why this is so, and helping to remove the barriers to REDD requires an understanding of the economics of such activities. Concerns regarding non-permanence of stored carbon, the risk that carbon sinks created in one place will simply result in sources elsewhere ('leakage'), as well as a lack of readily available knowledge of the financial feasibility of land-use based mitigation activities, have been cited as significant inhibitors to investments to date (Marland et al., 2001; Cacho et al., 2004; Richards and Stokes, 2004; van Kooten et al., 2004; Michealowa and Jotzo, 2005; Jindal et al., 2008; Carlson and Curran, 2009; Torres et al., 2009).

As part of developing an efficient, effective and equitable institutional framework for REDD implementation, progress has been made towards identifying appropriate funding sources depending on the phase of capacity creation or implementation (Streck et al., 2009). The economic cost of REDD, AR or any land-use based climate change mitigation venture is typically composed of opportunity, implementation as well as transaction costs. In this study, we focus on the transaction cost component of the economic feasibility equation, and particularly the nature of transaction costs incurred through creating the carbon asset associated with the implementation of REDD and AR activities. The transaction costs we consider are those incurred through the process of documenting, auditing and trading the volume of emission reduction units (ERUs) generated by underlying climate change mitigation activity. These transaction costs include the expense of GHG monitoring and modeling; scenario development; project documentation; verification and certification; the trade of emission reduction units; as well as engagement and negotiation with national government and local stakeholders. The transaction costs considered do not include project development or management costs such as the cost of managing a conservation area or an agroforestry enterprise.

The majority of REDD cost analyses to date have mainly focused on the opportunity costs (for example, Grieg-Gran, 2006; Stern, 2006; Kindermann et al., 2008). In general, the results have favored REDD activities, especially in poorer developing countries where avoided deforestation is assumed to have low opportunity costs and generate more income than the current land uses (Osbourne and Kiker, 2005; Stern, 2006; Grieg-Gran, 2006; Bellason and Gitz, 2008).

There is a need to better understand the nature of implementation and transaction costs and the effect they have on the overall financial viability of REDD and AR as a land-use choice (Jindal et al., 2008; Potvin et al., 2008; Carlson and

Curran, 2009). Numerous studies call for better estimation of transaction costs and cite the lack of knowledge of transaction costs as an impediment to the broader acceptance of REDD and AR activities (Jindal et al., 2008; Carlson and Curran, 2009). While certain studies assume transaction costs to be low or inconsequential, others cite them as high and view them as a roadblock to implementation, especially for small-scale projects (Skutsch, 2005; Cacho and Lipper, 2006; Jindal et al., 2008; Potvin et al., 2008).

Of the studies that have assessed the transaction costs of land-use based climate change mitigation, few have focused on REDD activities and fewer on projects located in ecosystem types other than moist forests. The past bias in AR research towards small-scale agro-forestry projects, for example Cacho and Lipper (2006), is due to REDD only really emerging as a potentially legitimate climate change mitigation activity after the publication of the 2007 IPCC reports and the adoption of the 'Bali Action Plan' at the United Nations Framework Convention on Climate Change (UNFCCC) 13th Conference of Parties meeting in December 2007.

In this study, we focus on the effect of transaction costs on the financial returns generated through the creation of the carbon asset associated with AR or REDD activities. For the purposes of this study, such financial returns are referred to as 'carbon income', which is equal to the revenues generated through the sale of Emission Reduction Units (ERUs, colloquially referred to as 'carbon credits'), minus the additional transaction costs incurred in creating the carbon asset generated by the underlying AR or REDD activity. The transaction costs considered are those incurred through: net GHG monitoring and modeling; scenario development; project documentation; verification and certification; the trade of emission reduction units; as well as engagement and negotiation with national government and local stakeholders. Emphasis is placed on including *all* the additional transaction costs incurred in the process of creating the carbon asset over and above the usual operational costs of project.

Further site-specific costs such as purchasing land, project implementation (e.g. planting and looking after trees, combating fires) and opportunity costs (the income foregone by adopting the AR or REDD land use options) were not included in the analysis.

The key questions explored were:

- What is the expected range of transaction costs incurred through AR and REDD activities in African woodland and rangeland systems?
- How do transaction costs change relative to the spatial scale of the mitigation activity?
- How can future policy and implementation mechanisms be structured, such that transaction costs and associated investment risks are minimized, thereby making AR and REDD a more financially attractive land-use choice?

Case study: AR and REDD activities in Sub-Saharan African woodland and rangeland systems

Although the carbon stocks per hectare may be lower than those observed in tropical forests, the woodland and rangeland systems of the planet contain at least one third of the global terrestrial carbon stocks, thanks to their vast spatial extent (Noble, 2003). Such systems are under considerable transformation pressure, particularly in emerging nations in sub-Saharan Africa (FAO, 2006). For example, in the period 1990-2005, the area of 'other wooded land' (i.e. the land with trees on, but not considered to be a forest – including woodlands and savannas, FAO 2006) in the Republic of Tanzania decreased from 22.3 to 4.7 million hectares at a rate of 11.1% per year.

Nearly three-quarters of anthropogenic emissions in sub-Saharan Africa are generated through land-use change. Therefore AR and REDD activities are the main mechanisms through which African countries can contribute to global climate change mitigation efforts (IPCC, 2007; CAIT, 2009). At the same time, woodland and rangeland restoration and maintenance can form a climate change adaptation option in its own right. The population is 70% rural. These small-scale farmers have been identified as one of the most vulnerable groups to climate change due to a close dependence on their immediate surroundings for resources and few other available livelihood options (Thornton et al., 2006; Shackleton et al., 2007; Laye and Gaye, 2008). In response to projected climate change-related increases in aridity and extreme events such as droughts and floods for the region, maintaining woodland and rangeland ecosystems and associated water, soil, grazing and fuel wood services, can form a practical climate change adaptation option (Boko et al., 2007).

2. Data and Methods

2.1. Vegetation functional type as a proxy for ecosystem type

As there is significant variation in climate, soils and plant structural and physiological characteristics across the woodlands and savannas of southern Africa, it is unrealistic to assume that carbon accumulation and storage will be the same everywhere. A 'functional group' approach was taken to broadly classify the vegetation of southern Africa based on the ecosystem function of carbon sequestration. A vegetation functional type (VFT) can be defined as a non-phylogenetic group of organisms that affect ecosystem functioning or respond to a disturbance in a similar manner (Gitay and Noble, 1997; Scholes et al., 1997; Walker et al., 1999; Hooper et al., 2002).

The main determinants of VFTs in southern Africa with regard to net primary productivity are plant available moisture and nutrient availability (Ellery et al., 1991). Using these two determinants as axes of variation, the vegetation functional types of southern Africa were separated into four broad classes (Fig. 1). Sub-divisions within a class are due to additional biophysical factors, such as fire, that can significantly affect observed carbon stocks. Vegetation types with very low sequestration rates, such as deserts and arid shrublands were omitted.

For each vegetation functional type considered, a well-studied field site was used to provide the parameters for simulation models and to validate the output. Appendix 1 contains particulars for each study site.

2.2. The context of the implementing entity

For the purposes of this analysis, it is assumed that the implementing agent, be it a public, private or not-for-profit sector entity, is already based in the host nation and has basic office premises, vehicles and administrative staff in place. The implementing agent could range from a national government, to a large-scale concession holder, a large-scale commercial farmer, or a development NGO focusing on community based agro-forestry and forest management. In all cases, it is assumed that the implementer is assessing a new venture, for example an agro-forestry enterprise or a wildlife tourism concession, which may lead to a net reduction in atmospheric GHGs and be eligible for emission reduction units if additionality criteria are met.

While the range of woodland and rangeland vegetation types examined are all based on actual sub-Saharan examples, the broad results are likely to be applicable to REDD and AR activities on other continents as well. Although project revenues may change significantly according to the carbon sequestration rates and standing biomass stocks of the vegetation type under study, the transaction costs incurred through creating the carbon asset are expected to be similar across different regions. Although these cost structures cannot be absolutely universal some international benchmarking was incorporated to assess generality and broad applicability of the proposed costs. The main cost item that would be expected to change relative to vegetation type are monitoring costs, which as explored below, forms a small component of the total transaction cost bill.

2.3. The financial model

A coupled biophysical and financial model was developed to assess expected carbon income generated by AR and REDD activities in each vegetation functional type. The biophysical component was simulated using the Century Ecosystem Model and the financial component was modeled using a spreadsheet (Microsoft Excel). To assess potential economies of scale, the financial model was run for projects from 5,000 to 100,000 hectares in 5,000 hectare increments, assuming that the vegetation type remained constant across all those scales.

2.3.1. Modeling carbon sequestration rates

The Century Ecosystem Program was used to model an AR scenario for each of the sites listed in Appendix 1. Century is freely available online from Colorado State University (www.nrel.colostate.edu/projects/century) and has been used successfully in numerous past studies on carbon dynamics (for example, Antle et al., 2003; Song and Woodcock, 2003; Mooney et al., 2004; Capalbo et al., 2004; Luo et al., 2008).

The results of a typical Century simulation are illustrated in Fig. 2. The model is initially allowed to run for 2000 simulated years to allow the main carbon pools to reach an equilibrium state (section a, Fig. 2). Thereafter, a disturbance event (b, Fig. 2) is simulated that reduces carbon stocks to the level found when the system is in a degraded state (point c, Fig. 2). The disturbance event can be a human intervention (for example fuel-wood harvesting) or a natural disturbance (for example fire) that leads to a loss of carbon from the system. Following the disturbance, a management regime is introduced from point c that allows the system to regenerate and accumulate carbon (d, Fig. 2). The simulated amount of carbon sequestered during the rehabilitation phase d is used in the financial model.

For each vegetation type, we endeavored to simulate a typical real-world degradation event followed by a feasible rehabilitation intervention. The restoration pathway for each vegetation type was based on observations described in published studies (Table 1). For the woodland and savanna sites situated in the Kruger National Park (Mopane, Combretum and Acacia savanna), the simulations were based on the studies of Shackleton et al. (1994), Shackleton (1997) and Scholes (1987) on rural areas adjacent to the Park. The sub-tropical thicket simulation was based on Lechmere-Oertel et al. (2005) and Mills et al. (2005) observations of thicket rehabilitation following unsustainable goat farming in the 1960's and 1970's. The Miombo woodland simulation is based on the reestablishment of woodlands following clear cutting (slash-and-burn) practices commonly found in central Zambia (Chidumayo, 1997).

It should be noted that only the biomass carbon pool was considered in this analysis making the reported net change in atmospheric carbon dioxide due to AR or REDD activities a conservative estimate. The changes in soil carbon stocks are typically greater that those in biomass stocks in these vegetation types, but for now are not eligible under CDM AR rules (Ringius 2002). For the purposes of this analysis, the below ground root biomass carbon pool was estimated using a generic allometric equation as per the IPCC Good Practice Guidance for land-use based activities (Cairns et al., 1997; IPCC, 2003). If implementation were to be considered for a particular location, better species-specific estimation of the below ground root biomass carbon pool as well as consideration of changes in the soil carbon pool could significantly increase the volume of emission reduction units awarded to the implementing entity (Wise et al., 2009).

2.3.2. Financial model

The financial model was based on the structure of a 'standard' Clean Development Mechanism (CDM) or Verified Carbon Standard (VCS) land-use change mitigation activity as described by Watson et al. (2000) and Penman et al.

(2003). For the purposes of this study, a project period of 20 years was chosen with a monitoring, certification and ERU issuance interval once every five years (Fig. 3).

There is a project development period of one to two years prior to the start of the project (Fig 3). It is during this period that approximately 60% of the total transaction costs are incurred through forest carbon monitoring, land use modeling, impact assessments, documentation and stakeholder engagement activities (see Section 2.3.3 and Table 2, 3). Thereafter, the creation of the carbon asset generated by the project occurs in five-year cycles where, at the beginning of each cycle, the project implementer establishes a forward-contract with an ERU purchaser that stipulates that the purchaser would provide upfront capital for a return in ERUs in five years' time. At the end of the five-year cycle, the project implementer provides a monitoring report to an auditor (or "Designated Operational Entity"), who verifies that the implementation of the project has lead to the claimed reduction in GHG emissions. Thereafter, either the CDM or VCS governing authority issues the project implementer with registered ERUs. The cycle is repeated six times until the end of the 30-year project period.

The relatively straightforward financial model described above was adopted for the purposes of the analysis, as it is broadly the general form of projects in the sector. In practice, the agreement between the ERU purchaser and project implementer is more complex. For example, the purchaser may provide the capital in a number of installments dependent on performance milestones, or may require an equity share in the underlying project, or include a form of risk-sharing agreement. As these additional aspects are often highly case-specific, a generalized model was adopted.

The financial analysis is based on the 'compensated reduction' model as proposed by Santilli et al. (2005) where the implementing entity is rewarded for reducing emissions from deforestation and degradation below a predetermined historical reference level. Whereas numerous compensation models for REDD have been proposed, the compensated reduction model has been adopted here due its simplicity and ease of understanding. In practice, a REDD activity developer would need to review all compensation for REDD models in terms of their appropriateness to the particular context under consideration. For the purposes of this analysis, the reference level is simply assumed to be the historical regional deforestation and degradation rate as estimated in the FAO Global Forest Resources Assessment (FAO 2006). As a mean deforestation and woodland degradation rate of 1.46% was calculated for the countries of region with adequate data for the period 1990-2005 (Angola, Zambia, Zimbabwe, Tanzania, Mozambique and Malawi), a conservative 1.0% deforestation rate was assumed for the analysis.

A carbon price range of US$ 4.00, 7.00 and 10.00 tCO2e was assumed for the analysis. Although the market price of carbon can vary substantially, most carbon market intermediaries contacted agree that a range from US$ 4.00 to 10.00 tCO2e is a reasonable estimate for carbon credits generated by land-use change mitigation ventures.

To manage and reduce exposure to non-permanence and general project risk, the Voluntary Carbon Standard (VCS) developed a buffer mechanism as a means of first assessing project risk and second, holding a portion of generated VERs in a 'buffer account' as an insurance mechanism. Through the VCS buffer mechanism, the quantity of VERs initially awarded to the implementing agent is generally discounted by 10-60% dependent on the perceived risk of the particular venture (www.v-c-s.org). To incorporate perceived non-permanence and leakage risk in this analysis, the quantity of awarded VERs for both AR and REDD activities were discounted by 25%.

Future income and costs were discounted at a risk-free interest rate for expected inflation. The average discount rate used for the analysis of 8% per annum was derived from the yield on a South African government 10-year bond (www.resbank.co.za). As the discount rate used may vary depending on numerous factors, discount rates of 6%, 8% and 10% per annum were modeled.

2.3.3. An overview of transaction costs

Transaction costs consisted of all costs incurred through creating the carbon asset associated with the underlying land use activity. Such costs include GHG monitoring and modeling, engagement and negotiation with national government and local residents, project documentation, verification and certification, registration, the trade of VERs, and transport and coordination costs.

The list of estimated transaction costs in Tables 2 and 3 are based on the authors' past experience in developing AR, REDD, bioethanol and energy sector climate change mitigation ventures in southern Africa as well as a review of past published CDM transaction costs (Appendix 2, Michaelowa, 2003; Cacho et al., 2004; Krey, 2004, 2005; Michaelowa and Jotzo, 2005; Cacho et al., 2005; Tyler, 2006; SSN, 2006; Antinori and Sathaye, 2007).

Table 2 contains the estimated transaction costs for a 100 000 hectare AR or REDD project located in woodland or rangeland system in sub-Saharan Africa. Table 3 contains variable cost items whose values are expected to vary relative to the spatial scale of the venture.

The estimated documentation, registration, and marketing and brokerage costs are based on empirical experience and are well within the range of past studies (for example, Michaelowa and Jotzo, 2005; Cacho, 2006; Tyler, 2006). The costs of monitoring biomass carbon stocks is based on field experience and assume that local residents (as opposed to international consultants) will undertake the majority of monitoring. Such a community-based approach to monitoring has been shown as a cost-efficient means of assessing carbon stocks while providing local employment opportunities (Skutsch, 2005). The required number of sampling plots quoted in Table 2 and used to estimate monitoring costs, are based on the number of plots calculated by Wise et al. (2009) when illustrating an optimal sampling effort methodology using similar southern African savanna sites. As optimal sampling effort is site specific and dependent on the size of carbon pools, carbon sequestration rates, the market price of ERUs, the marginal cost of each additional sample, and the number of sampling strata and variability within strata; the quoted number of samples should only be seen an approximate estimate of optimal sampling effort across all vegetation types.

Although it is generally not included in past analyses of transaction costs, consideration of the resources required for adequate, comprehensive community and national government engagement and negotiation is particularly important in the context of AR and REDD activities. Compared to industrial or energy sector climate change mitigation activities where a large volume of GHG emission reductions are generated at a point location with a small number of stakeholders and low negotiation costs relative to carbon revenues; AR and REDD activities typically cover vast spatial scales and require agreement from a large number and diverse range of stakeholders ranging from local residents to civil society organizations and national government. A successful implementation essentially requires all parties to commit to a certain type of land-use for at least the next 20 years (if not in perpetuity). Therefore, at a national scale it needs to be ensured that the proposed activity has government approval and is aligned with national land-use and resource strategies. At a local scale, the rights of indigenous peoples must be adequately considered. This requires negotiation with a large set of stakeholders, including education and awareness training on REDD and climate change issues. Payment and incentive mechanisms need to be negotiated with a large number of individuals. Moreover, the social welfare components of the activity need to be certified through an appropriate social impact assessment process such as the Community, Climate and Biodiversity Standard (CCBS). Giving such engagement, negotiation and certification due diligence can come at considerable cost (Table 2, 3).

The cost of workshop and negotiation meetings as well as transport of participants is based on prior experience in the sub-region. Transport costs were calculated using the South African Automobile Association's estimated running costs for a three-liter diesel vehicle including fuel costs (US$ 0.29 km^{-1}). Typically, GHG emissions from project operations, such as those generated through the transport of monitoring staff, would be included in estimating the net reduction in atmospheric GHGs due to the AR or REDD activity. Due to the relatively small size of such emissions, they have not been included in this analysis - the GHG emissions from the transport of monitoring staff for example would result in approximately 2.8 tCO2e been emitted (Defra 2010, 20,000 km traveled in a diesel vehicle at 1.4 $kgCO2e.km^{-1}$).

2.3.4. Sensitivity analysis of the price of carbon and discount rate used

A sensitivity analysis was performed on the price of carbon and discount rate used. The price of carbon and discount rate were identified as key input variables that could change substantially between projects and regions. As described above, a carbon price of US$ 4.00, 7.00 and 10.00 tCO2e was modeled and annual discount rates of 6%, 8% and 10%.

3. Results and discussion

One of the main principles of climate change mitigation activities is that emission reductions need to be real, measurable, permanent and in addition to that which would have occurred in the absence of the mitigation activity. This provides the basis for the flexible mechanisms of the Kyoto Protocol (Joint Implementation and CDM) and legitimacy to the concept of emissions trading. In addition, particularly for land-use change activities, it needs to be ensured that implementation does not have detrimental environmental and biodiversity effects and recognizes the rights and concerns of local residents and the authorities at various levels. While these principles are sound and necessary and will most probably be adopted in post-2012 climate change policy, ensuring that they are adhered to increases the total cost of the activity.

3.1. The absolute size of transaction costs

Our analysis conservatively estimates the transaction costs incurred through project documentation, registration, trading, monitoring, and stakeholder engagement and negotiation to be between US$ 500,000 – US$ 700,000 depending on the spatial scale of the project (Table 2 and 3). Of this amount, approximately 50-60% consists of documentation, registration and trading costs, with 15-20% is spent on stakeholder engagement and 10-20% on forest carbon monitoring costs. Approximately, 60% of the expenditure occurs in the development phase of the project, with the remaining amount spent on monitoring, verification and carbon issuance events once every five years (Fig. 3).

Although US$ 500,000 to 700,000 per activity may not appear an insurmountable amount in the context of global efforts to address climate change, it can provide a significant roadblock to project implementation especially when the implementing party is a development or environmental NGO and where most of the financial costs are upfront capital at risk. It is improbable that US$ 500,000 to 700,000 can be absorbed within annual budgets and venture capital appetite may be limited when post-2012 policy and mitigation mechanisms are still to be finalized.

Compared to other analyses of transaction costs, the amounts reported in this study tend to be similar or higher than those published elsewhere (Antinori and Sathaye, 2007; Cacho and Lipper, 2006; Jindal et al., 2008; Potvin et al., 2008; Torres et al., 2009; Wise et al., 2009). Antinori and Sathaye (2007) in their meta-analyses of transaction costs incurred through industrial and energy sector CDM projects estimate costs at US$ 0.03- 4.05 $tCO2e^{-1}$ depending on the scale of the venture. In comparison, this study found costs to range from US$ 0.01 to 68.92 $tCO2e^{-1}$ (Table 4 and 5). The higher costs, particularly at smaller spatial scales, can be attributed to the additional expense of extended stakeholder engagement and negotiation associated with land-use sector projects as well as the verification of social and environmental parameters.

In general, direct comparison of transaction costs between studies is difficult due to inconsistency in reported components and what each component constitutes (Cacho and Lipper, 2006; Antinori and Sathaye, 2007; Potvin et al., 2008; Wise et al., 2009). For example, this study and that by Cacho and Lipper (2006) are of the few that include stakeholder engagement and negotiation costs. Most tend to only report on monitoring costs and perhaps very basic documentation and registration fees.

3.2. The influence of spatial scale on financial feasibility

Strong economies of scale are found for the modeled transaction costs (Table 4 and 5). The observed economies of scale are typical of a production system that requires large upfront capital expenditure (virtually independent of the size of the operation), followed by smaller marginal costs that decrease for each additional hectare added. Income per hectare remains constant irrespective of the spatial scale of the venture since it is dependent on the fundamental carbon sequestration rates or the standing biomass stocks of the system. The result is that carbon income is highly dependent on spatial scale (Fig. 4 and 5). Especially for smaller scale projects (i.e. those below 5,000 – 10,000 hectares) transaction costs can significantly affect carbon income. For ventures above 20,000 hectares, transaction costs and their influence on the financial feasibility of the venture are relatively negligible: less than US$ 1.00 $tCO2e^{-1}$ decreasing to a few cents per tCO2e for activities over 500,000 hectares (Fig. 4 and 5, Table 4 and 5).

The break-even spatial scale (i.e. where income from traded emission reductions is equal to the transaction cost) ranges from 1,500 – 7,500 hectares for AR ventures and 4,000 – 12,500 hectares for REDD activities, depending on the vegetation type under consideration (Fig 4, 5, and 6). These results corroborate the findings Michaelowa et al. (2003), Cacho et al. (2005) and SSN (2006) that estimate a break-even volume of emission reduction been between 20,000 and 50,000 tCO2e per annum.

While this study focuses on transaction costs, similar economies of scale are likely to be found for implementation costs as well. Costs incurred through fire management, non-timber forest product (NTFP) commercialization (for example honey) or sustainable harvesting of timber are characterized by large upfront capital costs which are fairly independent of the spatial scale of the venture and thereafter decreasing marginal costs per additional hectare.

3.3. The influence of vegetation type on AR and REDD project feasibility

The carbon sequestration rate (net ecosystem production) and the upper limit to carbon storage of an ecosystem (amount of biomass in an intact state) are mainly determined by plant available moisture and soil fertility (Ellery et al. 1991, Sankaran et al. 2005). In general, the higher the annual rainfall, the higher the sequestration rate and the lower the break-even spatial scale for carbon sequestration (AR) ventures (Appendix 1 and Fig. 4, 6, 7). As an example, carbon income for a 50,000 hectare project ranges from a NPV of US$ 40.00 ha^{-1} for a project located in Mopane woodland on the drier end of the vegetation types modeled (506mm annual rainfall) to US$ 230.00 ha^{-1} for a venture located in Miombo woodland (956mm annual rainfall. Both modeled using an ERU price of US$7.00 tCO2e and a discount rate of 8%). It should be noted that a project in a vegetation type with a higher sequestration rate will not only generate more income per hectare, but be able to service upfront debt sooner thereby paying less debt over the lifetime of the venture.

On first reflection, one may assume that for a particular vegetation functional type, the higher the annual rainfall, the higher the sequestration rate, and therefore the higher the standing biomass stocks of the system in an intact 'equilibrium' state. In reality, this assumption does not always hold true (Table 1, Appendix 1, Sankaran et al. 2005). In addition to plant available moisture and soil fertility, parameters such as fire and herbivory can lead to a decrease in the standing biomass carbon stocks of the system in an equilibrium state (van Wilgen and Scholes, 1997; Bond and Keeley, 2005; Bond et al., 2005; Sankaran et al., 2005). In many woodlands and rangelands the equilibrium biomass stocks of the system and therefore the financial returns of REDD ventures are below what may be theoretically possible as a result of disturbance events such as fire and herbivory that may be practically impossible or undesirable to eliminate (Fig. 5, 6, 7).

The effect of fire and herbivory is not constant across all vegetation types modeled. A particular anomaly among the suite of vegetation functional types assessed is subtropical thicket (Fig. 5, 6, 7). Over 80% of the biomass in subtropical thicket is constituted by a single succulent tree species (*Portulacaria afra*, Mills and

Cowling, 2004). The system is therefore not prone to fire which results in higher biomass stocks in an equilibrium state compared to woodland and rangeland systems with similar annual rainfall and higher expected financial returns for REDD ventures located within the vegetation type. However, this species is extremely vulnerable to herbivory.

3.4. Expected carbon income from AR and REDD activities

Our analysis indicates that under current financial mechanisms and VER prices, AR and REDD activities in southern African rangeland systems are likely to be marginal, particularly for those systems with an annual rainfall of below 600-800mm per year (Fig. 4, 5, 7). Even at spatial scales greater than 50,000 hectares, carbon income ranges from a NPV to 20.00 to 80.00 US$.ha^{-1} for REDD activities and 40.00 to 230.00 US$.ha^{-1} for AR ventures depending on vegetation type (Fig. 7). Note that our analysis does not include implementation or opportunity costs, which can also be considerable (van Kooten et al., 2004; Potvin et al., 2008).

While the results are not encouraging for small-scale implementation (< 10,000 hectares), revenues generated through AR and REDD may provide crucial additional revenues to large-scale operations where the underlying land-use choice leads to the maintenance of forest and rangeland structure - for example, ecotourism, NTFP and sustainable timber harvesting. Especially in the Miombo Woodlands, AR and REDD activities over spatial scales of greater than 20,000 hectares could lead to substantial additional income for the implementing entity.

3.5. Sensitivity analysis of the price of carbon and discount rate used

In addition to the effect of spatial scale and vegetation type on expected returns, the sensitivity of carbon income to changes in carbon price and discount rate was assessed, as they are two particular variables in the financial model that may fluctuate considerably depending on the location and type of AR or REDD activity.

An increase in discount rate of one percentage point results in a decrease in carbon income of approximately 10% over the lifetime of a 50,000 hectare project (Table 6, 7). This relatively high sensitivity to the discount rate used is due to the occurrence of substantial upfront costs and then revenues that only occur once every five years from Year 5 and are therefore substantially discounted. Such sensitivity to discount rate is exaggerated in smaller scale ventures (< 20,000 ha) or projects located in vegetation types with relatively low carbon stocks, which still retain most of the substantial upfront transaction costs, but with lower revenues being realized.

The sensitivity of returns to carbon price is also dependent on the spatial scale of the AR or REDD activity. For projects of 20,000 hectares or larger, returns are closely related to carbon price – a 30% increase in carbon price results in a 32-35% increase in returns (Table 6, 7). Yet, as the size of the project decreases below 20,000 hectares, the sensitivity to carbon price is increasingly exaggerated due to the relative size of upfront transaction costs relative to revenues.

4. Implications for policy and implementation

If AR and REDD activities are to be broadly seen as the preferred rational economic land-use choice in woodland and rangeland systems, significant financial barriers to implementation need to be addressed. This is especially true for smaller scale AR and REDD ventures. The results of this study provide support for current proposals advocating national scale implementation (Angelsen et al., 2009) or at least a similar approach to 'programmatic CDM' that bundles a large number of similar projects together thereby partially sharing transaction costs. The creation of a national entity that facilitates project implementation while aligning projects with national land-use strategy and policy could significantly improve the attractiveness of AR and REDD as a land-use choice.

For the cost component of the net income equation, approximately 60% of upfront activities and associated costs such as scenario development, monitoring support (remote sensing and expertise), project documentation, and buyer engagement and brokerage could be undertaken by an appropriate national entity. While the national entity may need to charge a fee for such services, it is estimated that the costs to the project will be substantially reduced due to improved efficiencies. Moreover, the availability of such services would remove significant knowledge and capacity barriers that currently exist. The roll out of "REDD readiness" as currently been implemented through UN-REDD and the World Bank may address many of the capacity issues highlighted in this analysis and other studies (Cacho and Lipper, 2006; Ebeling and Yasue, 2008; Jindal et al., 2008). Such in-country capacity creation and the costs that are absorbed in the process may significant improve the attractiveness of AR and REDD as a rational land-use choice.

At a local scale, transaction and implementation costs could be substantially reduced if local residents undertake monitoring and project management (Skutsch, 2005). Not only does community–based implementation reduce costs, but it provides employment and skill development in rural areas where few such opportunities exist and ensures that part of the generated benefit stream reach local residents. In short, it has the potential to function as a typical ecosystem service public works program as demonstrated for water service delivery in South Africa (see Turpie et al., 2008).

For the revenue component of the net income equation, aside from just increasing the market price of VERs, national scale implementation may reduce the risk profile of AR and REDD activities and therefore reduce the level at which revenue streams are currently discounted for risk. First, following the principles of portfolio theory and efficient diversification of risk, a national entity could appropriately bundle numerous individual projects leading to lower net risk across the portfolio. Secondly, the national entity could ensure that the project is aligned with long-term national climate change and land-use strategies, which in turn would decrease non-permanence, leakage risk and resultant discounting of potential future revenues.

The development of ‘nationally appropriate mitigation actions’ (NAMAs) as well as ‘sustainable development policies and measures’ (SD-PAMs, Winkler et al., 2007), which are currently been proposed for inclusion in post-2012 climate change policy under the UNFCCC, provide a process where it is ensured that future mitigation actions have local government support and are nationally appropriate and sustainable. As the implementation of AR and REDD essentially requires commitment to a certain type of land-use for at least 20-30 years, such a thorough planning process is integral to addressing non-permanence concerns which may again reduce the risk profile of AR and REDD activities and the associated discounting of future revenues.

An underlying theme that has emerged throughout our analysis is the importance of the rate at which a mitigation venture is able to service upfront debt and the impact of this on the financial viability and attractiveness of REDD and AR activities. The effect of compounding interest on high upfront development and transaction costs expenditure together with revenues only being realized on Year 5, 10, 15 and 20, exaggerates the effect of vegetation type as well as economies of scale on the financial feasibility of mitigation ventures. Although only transaction costs are considered in this study, similar high upfront capital costs are be expected for implementation such as the cost of establishing NTFP commercialization or fire management operations. As funding options for REDD and AR are considered in post-2012 climate change policy (for example, Johns et al., 2008; Angelsen et al., 2009; Streck et al., 2009), emphasis should be placed on reducing the upfront debt incurred by the implementing entity as well as developing a mechanism that allows upfront payment, or at least more frequent payment, while not discounting the price paid due to risk inappropriately. Current proposals that combine public-funding to establish upfront national-scale capacity followed by a transition to compliance-market derived revenues may provide such a solution (for example, the 'phased approach' described by Angelsen et al., 2009). Such alternative financing approaches need to be encouraged and developed further if land-use based climate change mitigation and associated social and biodiversity benefits are to be realized at a significant scale.

Acknowledgements

The National Research Foundation, the Council for Scientific and Industrial Research and Stellenbosch University are thanked for contributing to this study. Dr Bill Parton is thanked for providing TK with Century Model tuition.

Table 1 - The equilibrium, degraded state and growth rate parameters obtained from published studies for each vegetation function type. State 'a', 'c', and phase 'd' refer to the corresponding labels in Fig. 2. The mid-range value for 'd' was used to calibrate the model.

Vegetation functional type	Study site	Equilibrium state 'a' - tons C / hectare	Degraded state 'c' - tons C / hectare	Restoration phase 'd' - % increase / year	Reference
Mopane	Letaba, S.Africa	11.5	3.3	2 - 3	Scholes, 1987, Shackleton, 1997
Combretaceae, savanna	Skukuza S. Africa	9.5	1.9	2.0 - 2.7	Scholes 1987, Shackleton *et al.*, 1994, Shackleton, 1997

Acacia savanna	Skukuza, S.Africa	16	2.5	2 – 6.4	
Miombo woodlands	Mongu, Zambia	27.5	1	1 – 2	Chidumayo, 1997
Subtropical thicket	Baviaanskloof, S.Africa	41	8	1 - 2	Mills *et al.*, 2005 Lechmere-Oertel *et al.,* 2005

Table 2 – The estimated transaction costs of a 100,000 hectare REDD or AR activity in sub-Saharan Africa

Cost description	Type	Year incurred	Estimated cost (US$)
Documentation and registration			
Project Design Document	Fixed	1	50000
VCS Project validation*	Fixed	1, 5, 10, 15, 20	20000
CCBS	Fixed	1, 5, 10, 15, 20	20000
Legal fees	Fixed	1	25000
Scenario development	Fixed	1	30000
Social and environmental impact assessment	Variable**	1	50000
Marketing and brokerage			
Buyer engagement and marketing	Fixed	1	20000
Brokerage	Variable	1, 5, 10, 15	20000
Stakeholder engagement and negotiation			
National Government engagement			
- Workshop meetings (US$ 10.000 x 2)	Fixed	1	20000
- Negotiation meetings (US$ 2000 x 5)	Fixed	1	10000
- Negotiation meetings (US$ 2000 x 2)	Fixed	5, 10, 15, 20	4000
Local stakeholder engagement			
- Workshop meetings (US$ 4000 x 6)	Variable	1	24000
- Negotiation meetings (US$ 500 x 5)	Variable	1	2500
Monitoring			
Monitoring costs	Variable	1, 5, 10, 15, 20	15870
Local office			
Office rental, electricity, telephone (2 years)	Fixed	1	24000
Project coordinator (salary, 2 years)	Fixed	1	50000
Transport			
15 return trips to capital city***	Fixed	1	4350
5 return trips to capital city	Fixed	5, 10, 15, 20	1450
TOTAL: (8% discount rate)			538,935

* Validation cost through the VCS assuming that a VCS methodology already exists

** Variable costs are quoted as estimated for a 100,000 hectare project in this table. Table 3 contains an estimation of how the variable costs may change relative to the spatial scale of project

***It is assumed that at least 15 return trips between the project site and the capital city will be required for monitoring, documentation, stakeholder engagement, negotiation, verification, and social and environmental impact assessments (mean distance is assumed to be 500km one-way at US$ 0.29 km^{-1}.

Table 3 – An estimation of how variable costs change relative to the spatial scale of REDD and AR ventures in sub-Saharan Africa

Project Size (ha)	1,000	10,000	100,000	1,000,000
Monitoring costs				
Indicative number of samples (Wise et al. 2009)	*64*	*297*	*1377*	*5995*
Human resources				
Monitoring supervision / trainer fees*	2500	5000	7500	15000
Local / community monitors	80	240	1040	4800
Transport				
On-site mileage @ US$ 0.29/km	29	290	580	1450
Flights (International)	1000	1000	1000	2000
Equipment and remote sensing				
Remote sensing**	5000	5000	5000	10000
Field Equipment***	750	750	750	1500
Sub-total: Monitoring costs	9,359	12,280	15,870	34,750
Local stakeholder engagement and negotiations				
Workshop meetings (US$ 4000 each)	16000	16000	24000	32000
Negotiation meetings (US$ 500 each)	2500	2500	2500	5000
Sub-total: Local stakeholder engagement	18,500	18,500	26,500	37,000
Social and environmental impact assessment				
Social and environmental impact assessment	20,000	30,000	50,000	100,000
TOTAL:	47,859	60,780	92,370	171,750

* This person would be responsible for designing the monitoring methodology, training of local monitors, initial field sampling, analysis and reporting

** Supervised classification using multi-temporal Landsat 7 data

*** Field equipment (GPS unit, clinometer, measuring tapes, stationary, backpack and clothing)

Table 4 – The transaction costs per ton of sequestered carbon dioxide for AR activities in sub-Saharan Africa relative to the spatial scale of the venture

Hectares	Transaction costs per vegetation type (US$.tCO2e^{-1})				
	Mopane	Combretum	Acacia	Miombo	Thicket
1,000	22.76	13.50	8.25	4.59	10.49
10,000	2.49	1.48	0.90	0.50	1.15
100,000	0.27	0.16	0.10	0.06	0.13
1,000,000	0.03	0.02	0.01	0.01	0.01

Table 5 – The transaction costs of REDD activities in sub-Saharan Africa relative to spatial scale measured in tons of carbon dioxide per US$.

Hectares	Transaction costs per vegetation type (US$.tCO2e^{-1})				
	Mopane	Combretum	Acacia	Miombo	Thicket
1,000	56.93	68.92	40.92	23.81	15.97
10,000	6.23	7.55	4.48	2.61	1.75
100,000	0.68	0.83	0.49	0.29	0.19
1,000,000	0.07	0.09	0.05	0.03	0.02

Table 6 – Afforestation and reforestation activity: An assessment of the sensitivity of financial returns (NPV in US$) to changes in carbon price and annual discount rates. A carbon price of US$ 4.00, 7.00 and 10.00 per tCO2e and discount rates of 6%, 8%, 10% were modeled. Project area: 50,000 hectares.

Carbon price	Discount rate	Vegetation type				
		Mopane	Combretum	Acacia	Miombo	Thicket
4,00	6%	1 247 634	2 419 634	4 177 634	7 810 835	3 181 434
	8%	1 000 458	1 964 433	3 410 394	6 398 715	2 591 016
	10%	812 641	1 615 873	2 820 721	5 310 741	2 137 974
7,00	6%	2 478 234	4 529 234	7 605 735	13 963 837	5 862 385
	8%	2 012 631	3 699 587	6 230 019	11 459 580	4 796 107
	10%	1 656 034	3 061 691	5 170 175	9 527 710	3 975 367
10,00	6%	3 708 834	6 638 835	11 033 836	20 116 838	8 543 335
	8%	3 024 804	5 434 740	9 049 644	16 520 446	7 001 199
	10%	2 499 428	4 507 509	7 519 629	13 744 679	5 812 761

Table 7 – REDD activity: An assessment of the sensitivity of financial returns (NPV in US$) to changes in carbon price and annual discount rates. A carbon price of US$ 4.00, 7.00 and 10.00 per tCO2e and discount rates of 6%, 8%, 10% were modeled. Project area: 50,000 hectares.

Carbon price	Discount rate	Vegetation type				
		Mopane	Combretum	Acacia	Miombo	Thicket
4,00	6%	677 221	544 316	975 730	1 733 076	2 616 471
	8%	531 292	421 977	776 816	1 399 737	2 126 332
	10%	421 708	330 621	626 291	1 145 339	1 750 776
7,00	6%	1 480 012	1 247 428	2 002 402	3 327 758	4 873 700
	8%	1 191 590	1 000 290	1 621 258	2 711 368	3 982 911
	10%	971 902	812 500	1 329 922	2 238 257	3 297 771
10,00	6%	2 282 803	1 950 541	3 029 074	4 922 441	7 130 929
	8%	1 851 889	1 578 602	2 465 699	4 023 000	5 839 489
	10%	1 522 096	1 294 379	2 033 554	3 331 175	4 844 766

	Infertile	Fertile
Wet	Miombo woodlands	Moist grasslands
Dry	Mopane woodlands Broad-leaf savanna	Subtropical thicket Fine-leaf savanna Dry grasslands

Fig. 1 - A diagram showing the separation of vegetation types in southern Africa based on plant available moisture and nutrient availability. Further subdivisions were driven by additional prevailing biophysical variables (for example, fire frequency).

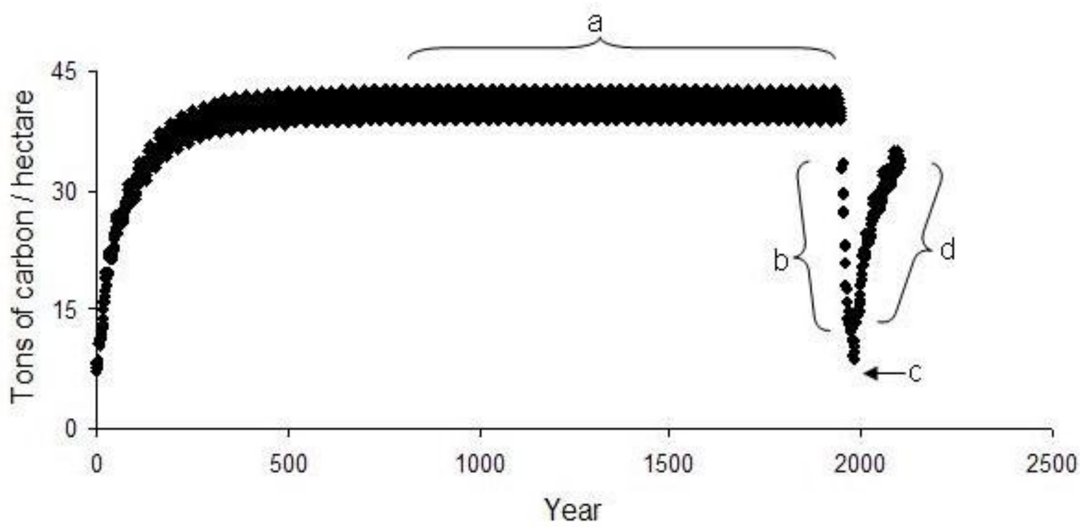

Fig. 2 - An example of the results of a typical Century Ecosystem Program simulation run for sub-tropical thicket. See Table 1, for values for a, c and d for each vegetation functional type. ‘b’ is the period in which the carbon stocks are reduced to a ‘degraded state’(indicated by “c”) through substantial increases in herbivory or the harvesting of wood. “d” is the reforestation recovery period during which the potential rate of carbon sequestration is estimated for the particular vegetation type modeled.

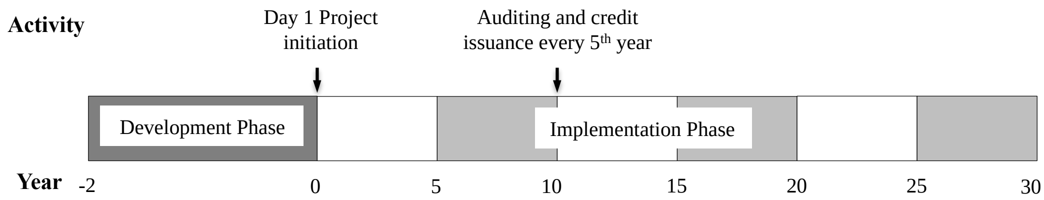

Fig. 3 - A typical time frame for land use based climate change mitigation ventures including a project period of 30 years with a monitoring, verification and ERU issuance event once every five years. It is important to note that the majority of transaction costs are incurred prior to the start of the project in the development phase.

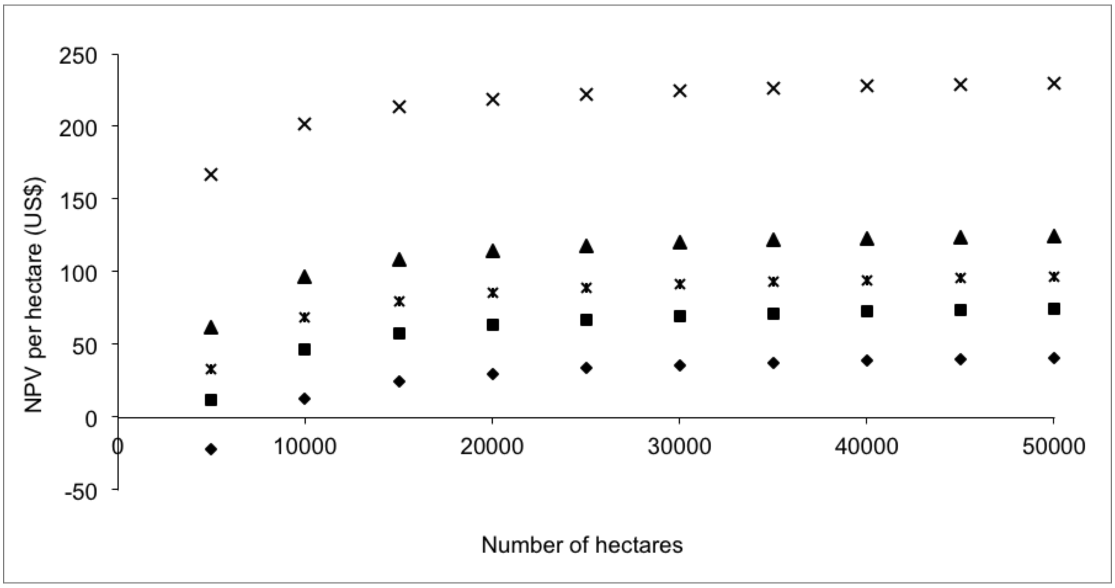

Fig. 4 – The change in financial returns from the creation of carbon asset generated by *afforestation or reforestation* activities relative to spatial scale: In Mopane woodland ♦, Combreteace savanna ■, Acacia savanna ▲, Miombo woodland ×, Sub-tropical thicket ж. (ERU price = US$7.00 tCO2e, Discount rate: 8%)

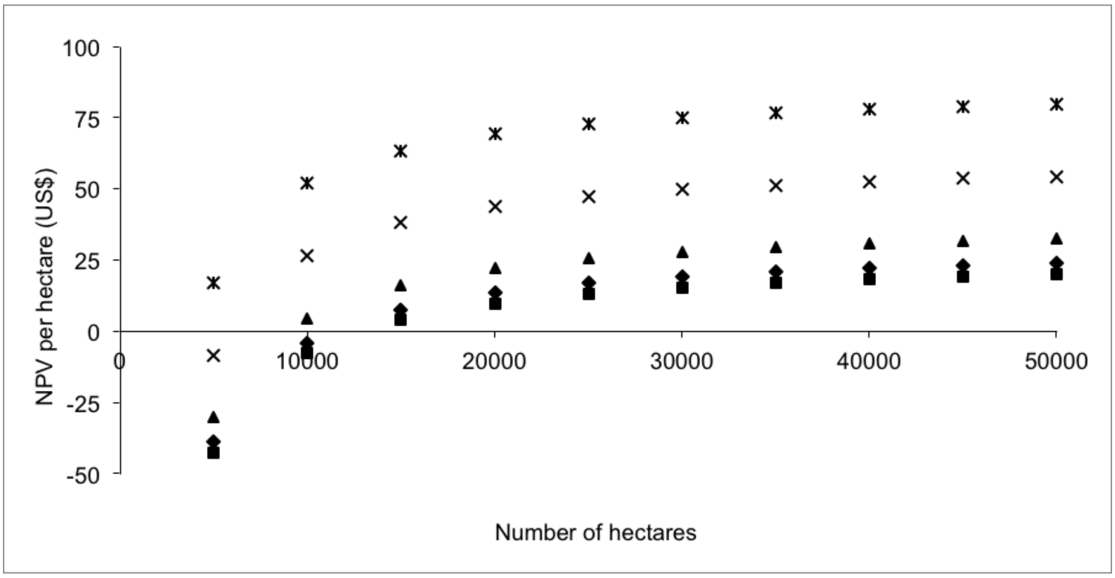

Fig. 5 - The change in financial returns from the creation of carbon asset generated by *REDD activities* relative to spatial scale: In Mopane woodland ♦, Combreteace savanna ■, Acacia savanna ▲, Miombo woodland ×, Sub-tropical thicket ж. (ERU price = US$7.00 tCO2e, Discount rate: 8%)

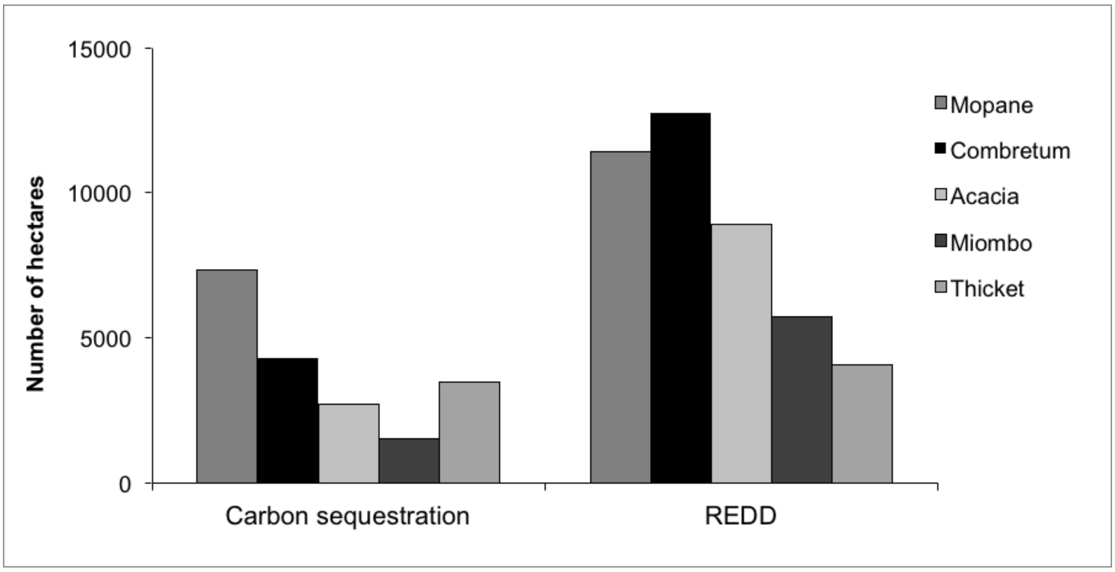

Fig. 6 - The spatial scale at which the revenues from carbon credit sales equals transaction costs for each vegetation functional type (the break-even point). A comparison of an additional sequestration versus a REDD venture.

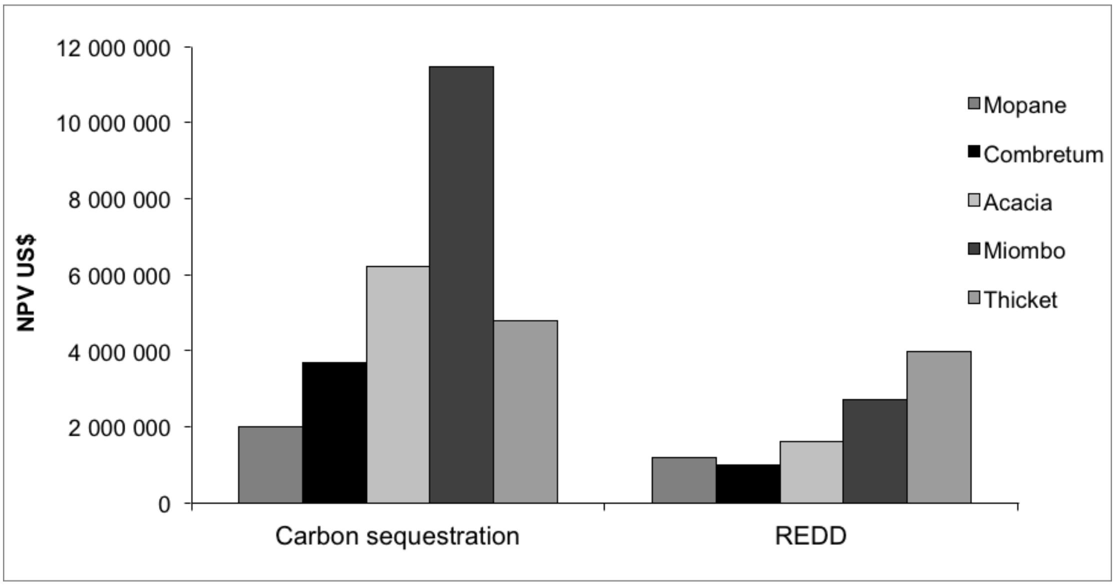

Fig. 7 – The financial returns from the creation of the carbon asset generated by a 50,000 hectare land-use sector climate change mitigation activity. A comparison of additional sequestration and REDD activities in a range of vegetation types.

Appendix 1 - A brief attribute description of the study sites employed in this modeling study.

Vegetation type	Study site	Rainfall*	Temperature*		Soil		Dominant plant species	Biomass
		mm/year	Mean °C	Range °C	Type	Sand/silt/clay %		tons / hectare
Mopane	Letaba, S. Africa	506	23	32.3 - 7.5	Coarse loamy	80.5/10/9.5 Paterson and Steenkamp 2003	*Colophospermum mopane*	21.6 Scholes 1987
Combretaceae savanna	Skukuza, S. Africa	572	22	2.4 - 35.9	Sandy	69/5/26 Woghiren, 2002	*Combretum apiculatum*	18.9 Shackleton 1997
Fine-leaved savanna	Skukuza, S. Africa	572	22	2.4 - 35.9	Clayey	65/3/32 Woghiren, 2002	*Acacia nigrescens*	28 Scholes and Walker 1993
Miombo woodlands	Mongu, Zambia	956	23	6.5 - 39.2	Sandy	95/2/3 Pinheiro *et al* 2001	*Brachstegia spiciformis*	55 Desanker *et al*., 1997
Subtropical thicket	Baviaanskloof, S. Africa	413	19	5.1 - 34.5	Loamy	83/9/8 Lechmere-Oertel 2003	*Portulacaria afra*	80 Lechmere-Oertel 2003

* Climatic data was obtained from the South African Weather Service (www.weathersa.co.za)

Appendix 2 - A table showing the results of a literature survey of the transaction costs incurred through the creation of the carbon asset associated with land use based carbon sequestration ventures. The amounts are reported in US$.

Transaction cost	**Publication**						
	Tyler (2006)	Little (2006)	Cacho and Lipper (2006)	UNDP (2003)	EcoSecurities (2002)	Krey (2004, 2005)	Michaelowa and Jotzo (2005)
Project Development and PDD	41061	39000 – 104000	22000 - 120000	30000 - 60000	15600 - 19500	28000 - 159500	65000
Project validation	15600	6500 – 19500	7000-120000	10000 - 15000	13000 - 26000	6000 - 80000	19500 - 39000
Project registration	5533	975 – 1950		10000		5000 - 34000	13000
Project verification and monitoring*	8450	5200 – 14000	5000-270000	3000 - 15000	13000 - 19500	16000 - 30000	23400
CDM Adaptation levy (2% of CERs)	1456	n/a	n/a	n/a	n/a	10193 - 212349	n/a
ERPA and legal fees	27671	n/a	n/a	20000 - 25000	19500 - 32500	n/a	n/a
Broker commission	5096	n/a	n/a		n/a	16000 - 30000	n/a
Total:	104,866	n/a	n/a	73,000 - 125,000	n/a	160,000 - 715,000	*Min fixed:* 195,000

Based on an exchange rate 1 Euro = 1.3 US$

* (per validation)

Chapter 5

The risk of fire to forest-based carbon storage activities in Africa

In the process of considering REDD and reforestation activities, one is often asked the question: What happens if it burns down? As a student of African savanna ecology, one is quick to point out that although the majority of woodland and rangeland systems in southern Africa are prone to fire, a single fire event is unlikely to have a significant affect on the net carbon stocks of the system. The follow up question is then: But what about plantation fires and the recent fires in Indonesia? And so the discussion continues.

Although the risk of fire to REDD has been highlighted following particularly large fire events, to date there has not been a comprehensive review of the nature of fire risk, how it may be quantified, and how the risk of the fire may vary depending on the region and vegetation type in which the REDD activity is located.

This review paper aims to provide a foundation on which to consider the risk of fire to REDD activities in sub-Saharan Africa. The ideas presented in the paper emerge from a review of the literature and discussions with my supervisor, Dr Bob Scholes, as well as conversations with Dr Winston Trollope and Chris Austin of Working on Fire International.

The risk of fire to forest-based carbon storage activities in Africa

Tony Knowles, Robert. J. Scholes, Albert. S. van Jaarsveld and Allan Ellis

Abstract

Recent large wildfires around the globe have highlighted the potential impact of fire on climate change mitigation activities based on forests. As parties develop large-scale programs aimed at reducing emissions generated through the deforestation and degradation of forest and rangeland systems (REDD), the impact of fire on the likely success and feasibility of the initiatives needs to be better understood. This is important not only to investment in smaller scale project-based activities but also for the consideration and funding for national-scale programs.

We review the risk of fire to REDD activities located in several widespread African vegetation types. Our approach combines the fields of systems ecology and investment finance, with 'variability in expected outcomes' as a common currency between the two disciplines. The fire ecology of both fire-prone and non-prone systems is explored with a view to understanding the risk fire presents to the feasibility of a REDD activity over a 20-30 year time period. REDD activities are assessed in a broad sense, including the avoided degradation of open savanna and grassland vegetation types in addition to the deforestation of forests and woodlands. The influence of projected changes in climate as well as possible changes in human activities on the risk of fire is also considered.

The review is based on published literature and data. It concludes that fire does not pose a significant and unmanageable risk to the expected outcome of REDD activities located in grassland, savanna and woodland ecosystems. In particular, when the net effect of fire on the total carbon pool of a system is considered (including root and organic soil carbon), carbon stocks are shown to be resilient to fire over decadal time-scales. The impact of climate change and anthropogenic disturbances on the probability of fire in moist tropical forest systems where it has been previously absent may be a greater concern. Such fires can lead to a large fraction of the system's carbon stock being released into the atmosphere, and not be recoverable within the time- and financial- constraints of a 20-30 year REDD activity.

Introduction

The importance of both land-use based climate change mitigation and adaptation was brought to the fore with the publication of the Intergovernmental Panel on Climate Change's (IPCC) 2007 reports (Boko et al. 2007, Denman et al. 2007). As approximately a fifth of anthropogenic greenhouse gas (GHG) emissions are generated through land-use change, reducing the deforestation and degradation of forest and rangeland systems can make an important contribution to limiting anthropogenic climate change (Denman et al. 2007). However, comparatively few land use based climate change mitigation projects have been established to date, particularly in Africa. One of the main reasons for the scarcity of projects is the perceived risk-return relationship associated with investments in REDD and reforestation activities (Angelsen et al. 2009, Streck et al. 2009).

If the risk and uncertainty associated with a land use based activity is compared to that of an industrial sector mitigation venture, it is understandable why the majority of project investments to date have been in the latter category. Emission reduction activities in an industrial sector project typically occur at a single point, and relate to a well-known technology. The implementation and maintenance of the technology is generally not highly dependent on national capacity or policy, and the owner of the factory is likely to be a sizable registered entity that forms an adequate legal counter party. It is therefore relatively straightforward to estimate the risk of non-delivery of emission reduction units (ERUs) and the degree of surety of the investment.

In comparison, the risk of non-delivery of ERUs in a land-use based venture is less well understood. The success of the land-use based activity is more reliant on supporting national capacity and policy, especially when considering the issues of permanence, leakage, land-use planning and sovereignty (Chomitz 2002, Angelsen et al. 2009, Streck et al. 2009). The emission reduction activity requires carbon stocks to either be maintained or restored across an extensive, variable and sometimes hard to access landscape, which in turn increases the complexity of the project, the monitoring costs and introduces additional forms of risk such as fire, drought and pests. There is a clear need to explore and quantify our understanding of the risks that fire, drought and other biophysical factors bring to investments in land-use change activities. Chapter 6 of this thesis for example, focuses on estimating the risk that projected changes in temperature, rainfall and atmospheric carbon dioxide present to the outcome of REDD activities. Better estimation of risks improves the attractiveness of a project to investors and reduces the percentage risk allocation when calculating the price of ERUs. Whether the ERUs generated through REDD and reforestation activities are traded through a market-based or national-scale compensation mechanism in the future, a more reliable estimation of risk is likely to improve the attractiveness of land-use change as a climate change mitigation option and its associated financing.

The risk of fire to indigenous forests and REDD and reforestation activities has been widely raised as an important concern (Pierce et al. 2004, Running 2006, Westerling et al. 2006, van der Werf et al. 2008, Conrad and Soloman 2009). The occurrence of unusually large and severe wildfires in Australia, Russia and Indonesia in the past few years has drawn attention to the risk to carbon stocks located in forests (Page et al. 2002, van der Werf et al. 2008, Bowman et al. 2009). Yet the risk of fire to carbon stocks is not equally high in all ecosystems. Many systems are consistently too wet to burn (van Wilgen and Scholes 1997, Thonicke et al. 2001) - such as the moist forests of the Congo Basin. Other systems, such as the dry deciduous 'Miombo' woodlands of southern and eastern Africa burn at one to four year intervals (Frost 1996, Williams et al. 2008), and have done so for millennia. In the latter case the fire is generally in the form of a surface burn (rather than a canopy fire) that consumes the dead grass, with little effect on total carbon stocks of the ecosystem system, which are predominantly held in the soil and woody biomass.

To understand the risk of fire to REDD and reforestation activities better, we systematically explore the notion of investment risk and the nature of the impact of fire on carbon stocks (Figure 1). The classic economic definitions of 'risk' and 'uncertainty' as defined by Knight (1921) have been adopted for the purposes of the analysis, where risk is present when future events occur with *measurable probability* and uncertainty is present when the likelihood of future events is *indefinite or incalculable*.

The discussion initially focuses on the factors that determine if a system is prone to fire and the likely impact fire may have on carbon stocks over the lifespan of a land-based carbon storage project. Thereafter, the effect of climate change and human-disturbance on fire regimes is considered as well as the potential to manage the risk of fire to REDD and reforestation projects over a 20-30 year time period. Twenty to thirty years is emerging as the norm with regard duration of Voluntary

Carbon Market REDD projects. Although the accounting period for national-scale REDD has yet to be finalized, 20-30 years is assumed to be a reasonable estimate of the duration of future national-scale financing agreements in support of REDD activities.

We focus on the fire risk in several very extensive African vegetation types that are likely to be the location of REDD or reforestation activities. Sixty percent of greenhouse gas (GHG) emissions in sub-Saharan Africa are generated through land-use change (CAIT 2009). If South Africa is excluded, the contribution of land-use change to regional GHG emissions rises to 80%. Addressing deforestation and forest and rangeland degradation is therefore the primary mechanism through which African nations can reduce GHG emissions. The African woodland and savanna ecosystems in which REDD and reforestation projects are likely to concentrated are prone to fire, which makes sub-Saharan Africa a particularly relevant case study. The discussion is relevant to similar forest and rangeland systems elsewhere in the world.

1. Defining the risk of fire to the outcome of REDD and reforestation activities

Introducing the notion of fire risk

Where the outcome of a project or investment is not fixed, an element of risk exists. Conversely, the presence of risk implies that a range of outcomes is possible. Each potential outcome has a probability of occurrence relative to every other outcome. Risk quantification is the estimation of the probability of each outcome occurring (Bodie et al. 1996, Reilly and Brown 2003). The standard measures of risk are variance, the standard deviation of expected returns from the mean, or the range of returns (Reilly and Brown, 2003).

The causes of variability in expected outcomes are known as a ‘risk factors’. They are conventionally grouped into classes, for example, legal, political or operations risks (Chance, 2003). Investment in ecosystem service ventures presents an emerging class of risk that we term ‘biophysical risk’. Biophysical risk can be defined as ‘the deviation from expected returns in an ecosystem service investment due to variability in biophysical conditions such as rainfall, temperature, fire or pests’.

To date, the potential effect of biophysical factors on the long-term outcome of ecosystem service ventures has been a concern but has generally remained unquantified. It is therefore seen as a source of uncertainty that reduces the attractiveness of land-use based climate change mitigation activities to financiers. Authors have for example, cited 'uncertainty regarding the permanence of sink projects' (Subak 2003), 'ecological risk' (Baer 2003) or 'risk due to disturbances such as fire or pests' (Cairns and Lasserre 2004) as reasons for investor caution. Fearnside (2000) and King (2004) illustrated how such uncertainty may lead to significant risk adjustment on the volume of carbon offsets awarded to a project and the effect this has on the financial viability of ventures. Most authors highlight the need for an improved understanding of the nature of biophysical risk factors, in order that physical and financial risk management techniques may be applied (Fearnside 2000, Laurikka and Springer 2003, Cairns and Lasserre 2004 and King 2004).

Variability as a common currency for multi-disciplinary risk analysis

Assessing fire risk in REDD and reforestation projects requires an approach combining the fields of systems ecology and investment finance. When considering multi-disciplinary analysis it is useful to have a metric or currency that is common to both disciplines. In the context of systems ecology and investment finance, the common currency of 'variability in expected outcomes' may be used.

Understanding variability in expected outcomes lies at the core of both disciplines. Understanding variability in expected returns and the risk - return relationships between investments is central to investment finance. Similarly, understanding variability in ecosystem structure and function is an integral part of systems ecology. Within both disciplines it is recognised that it is often not the mean or expected outcome that is the crucial factor, but the variability in expected outcomes.

Both disciplines estimate variability in expected outcomes in a similar manner, using historical or statistical analysis. Both quantify 'risk' or 'variability' in terms of frequency and magnitude. However, the terminology used differs. Investment analysts refer to 'risk' and 'risk factors', whereas ecologists refer to 'variability' and 'ecological disturbances'.

Introducing the quantification of risk

The quantification of risk is an assessment of the probability of a certain outcome occurring. Probability distributions are used to describe the range of potential outcomes and their probability relative to each other. The standard metrics used are the mean (the 'expected outcome') and some measure of the variability of the distribution or the 'range of returns' such as the 95% confidence interval (Reilly and Brown, 2003). Generally, the greater the standard deviation, the greater the range of returns, the wider the confidence interval and the greater the risk that the outcome will deviate from the expected mean return. The value-at-risk can be quantified using the measure of variability.

Value-at-risk, which in this case can be thought of as carbon-stock-at-risk, is defined by Chance (2003) as 'the loss that would be exceeded with a given probability over a specific time period'. Estimation of value-at-risk includes an assessment of the magnitude of loss and the associated probability over a specified time period (Chance 2003). The risk in the case of a land use-based climate change mitigation venture is the probability that the carbon stocks at the end of the project's lifetime fall below a specified value. Many factors may contribute to variability in expected carbon outcomes, including climatic variability, operational and political risk.

We focus on the risk fire presents to carbon stocks over the 20-30 year lifetime of a REDD or reforestation project. We explore the biophysical factors that determine the occurrence, type and intensity of fire in major African vegetation types. Particular emphasis is paid to the fire ecology of ecosystems that experience frequent fires ('fire-prone', fire return periods of 1-20 years). In these systems current carbon stocks are typically below their climatically-feasible potential. Secondly, we review the fraction of the above- and below-ground carbon pools that are released into the atmosphere through a typical fire event and the period of time required for the carbon pool to recover to a pre-fire value. We conclude with a discussion of the risk of fire to REDD and reforestation projects in Africa as well as a consideration of the feasibility of reducing fire risk through management or policy actions.

2. What factors determine if an ecosystem is prone to fire?

The occurrence of fire requires the simultaneous presence of three factors: a combustible fuel in sufficient quantity to sustain a fire; weather conditions that allow ignition and a source of ignition (van Wilgen and Scholes 1997, Pechony and Shindell 2010). Warm and dry weather conditions are required to sustain fire – in African circumstances this means air temperatures above about 15° C and a relative humidity below 50%. The fuel needs not only to be sufficiently dry in order to ignite, but be sufficient in quantity within a given packing volume in order for the flame to propagate from one fuel element to the next (van Wilgen and Scholes 1997, Trollope et al. 2004). Herbaceous layer fires peter out if the mean fuel load falls below about 30 g DM/m^2 (or 300 kg/ha). The spatial heterogeneity (patchiness) of the fuel plays a role here as well – fires may die out if the fuel is in clumps separated by bare areas. If the fuel is initially too moist to burn, the source of ignition needs to be of sufficient power and duration to reduce the moisture content of the fuel prior to ignition. Wind - especially a hot dry wind such is characterised by descending air - increase the chance that a fire, once ignited, will spread. It does so through three mechanisms: drying the fuel, increasing available oxygen at the flame front, and carrying the ignition from one burning element to the next. A well-oxygenated fire will burn more intensely and completely (Trollope et al. 2004) than an oxygen-starved one.

Fires in Africa, in the present time and probably for the past thousands of years, are overwhelmingly ignited by people. In general, there is no shortage of ignition opportunities, thus African fire regimes are said to be ‘not ignition-limited’. As a result the relationship between fire probability and human population density in Africa is counter-intuitive – it *decreases* as the rural human population density increases above a critical threshold of about 10 people/km^2 (Archibald et al 2009). This is because more populous landscapes have more barriers to the spread of fires (roads, settlements, fields, rivers) and a greater fraction of the potential fuel is consumed by domestic livestock. An increase in the frequency of small fires is often observed in the immediate vicinity of densely settled (urban) areas.

There is evidence of hominid use of fire in Africa from over one million years ago (Brain and Sillen 1988), however it is believed that humans only gained control of fire (i.e. the ability to ignite fire at will) more recently, perhaps a quarter of a million years ago. But the frequency of fires in Africa sharply increased before this, around 5 million years ago (Beerling and Osborne 2006). So although contemporary fires are lit by humans, human agency is clearly not a necessary condition for the prevalence of fire in Africa. The most likely alternate ignition source is lightning. Long term records from savanna protected areas (e.g. the Kruger National Park, Hluhluwe Game Reserve) suggest that lightning ignites 10-20% of fires. It is speculated than in pre-human times, a smaller number of lightning ignitions was nevertheless sufficient to achieve more-or-less the same burned area fraction as in the post human period (i.e., a similar probability of fire at any given point) because the individual fires were larger, given the abundance of fuel and absence of barriers to their spread.[5]

Ignition, fuel and weather vary considerably across and within terrestrial ecosystems. This in turn leads to the variability in observed fire regimes and the degree to which fire influences ecosystem structure and associated carbon stocks (Geldenhuys 1994, van Langevelde et al. 2003, Bond et al. 2005). The fire regime of a system is defined by the frequency, intensity, seasonality and type of fire (Scholes et al. 1996). Figure 1 describes how fire risk differs between African ecosystems with particular focus on the potential impact of fire on carbon stocks over decadal time scales. We adopt the functional classification of vegetation types developed by Scholes et al. (1996), which considers, among other things, differences in fire regime. (see also Archibald et al 2010a Southern African fire regimes as revealed by remote sensing).

The type and abundance of biomass and associated fuel elements is dependent on the balance between the net primary productivity of the ecosystems and the various avenues of biomass loss – decomposition, herbivory and fire. Primary productivity in Africa is a function of plant available moisture and the availability of soil nutrients (Ellery et al. 1991 van Wilgen et al. 2003). Arid and semi-arid ecosystems only generate a sufficient fuel load to sustain fire after exceptionally wet years, which happens rarely (van Wilgen and Scholes 1997). Above approximately 450 mm of

---

[5] Athough fires are overwhelmingly lit by people in Africa, this does not mean that they are 'anthropogenic' in the sense implied by the UN Framework Convention on Climate Change or the Intergovernmental Panel on Climate Change, which uses this term to mean things that have changed in the modern era, ie since about 1750. There is no evidence that fire frequency has systematically changed in Africa over this period.

annual rainfall, enough of a fuel load accumulates in summer-rainfall rangeland and woodland ecosystems to burn on a frequent basis (<10 year return time), and the mean fire return time decreases with increasing rainfall, to around once every 3 years at about 700 mm and above about 900 mm can reach once a year (in those parts of the landscape that are fire prone – the whole landscape burned area fraction is seldom more than 50%). When annual rainfall increases above about 1800 mm the effective dry season gets progressively shorter to the point that continuous moist conditions inhibit the occurrence of fire even though there is sufficient fuel (van Wilgen and Scholes 1997). There is a strong feedback between tree canopy closure, herbaceous fuel load and fuel moisture content (in a closed forest or woodland, grass growth is suppressed and the humidity remains high, whereas in an open savanna or grassland there is both fuel and the opportunity to dry it out during a rainless period of a few months). The result is, at this end of the rainfall gradient, a sharply-defined vegetation mosaic consisting of forest patches in fire-protected areas and grasslands with a few fire-tolerant trees in fire-accessible areas (Accatino et al. 2010).

Fire therefore mainly occurs within the 400-1800 mm annual rainfall range (Figure 1, 1.1). In addition to annual precipitation as a broad determinant of fire, seasonality in rainfall, particularly the regular occurrence of a prolonged (> 3 months) dry season is a key characteristic of fire prone systems (Archibald et al. 2009, Pechony and Shindell 2010). Fire requires a dry period sufficient to cause the herbaceous layer to die and dry out. This rainfall range supports a wide range of vegetation structural types, with varying degrees of tree cover on a continuum from virtually treeless grasslands, to low grassy shrublands, open savannas (<40% tree cover), woodlands (savannas with 40 to 60% tree cover) and forests (>60% tree cover).

The pattern of tree cover within a landscape often shows a repeated and thus predictable pattern associated with topography and soil conditions (the latter are also partly related to topography, in terms of the catena concept: Milne 1935). Steep slopes and areas near rivers are tree covered, due to the reduced intensity of fires, except for areas that are seasonally waterlogged (often midslope seeplines or bottomland marshes called *vleis* or *dambos*), which are treeless and burn frequently. In the moderate rainfall range (450 to 900 mm/y), clayey, fertile soils - such as those derived from basalts - support a more variable fire regime than sandy, infertile soils. The reason is a greater grass production per unit of rainfall, but also a greater amount of rainfall needed to have initial production (Scholes 2003). They support a higher grazer biomass, but in exceptionally wet years, the grazers cannot consume all the grass, and in the subsequent dry season intense fires result. This leads to a lower tree biomass, because the occasional intense fires act as a bottleneck on tree recruitment (Bond et al. 2003b). The clayey soils have an inherently higher soil organic carbon store than the sandier soils.

The basic climate and soil template can be affected by fire to varying degrees (Scholes and Archer 1997, van Langevelde et al. 2003, Sankaran et al 2005, Accatino et al 2010). The woody biomass and associated carbon stocks of the grasslands, savannas and woodlands of east and southern Africa are therefore typically substantially below their climatically-determined potential, mainly due to fire (Scholes and Archer 1997, Bond et al. 2003b, 2005, Sankaran et al. 2005). Other factors potentially keeping the carbon stock low are (1) herbivory of the trees - particularly by large browsers such as elephant and giraffe – in areas where these species are abundant, and (2) harvest of the woody plants by people for charcoal, fuelwood or poles, or to clear lands rotationally for crop agriculture (i.e. slash-and-burn landscapes).

Despite the underlying complexity of the interactions between climate, soil, topography, herbivory and human use, a number of trends emerge, especially when viewed at landscape scale and over the decadal time scales appropriate for climate change mitigation activities.

Grass production is inversely related to tree cover through a complex set of direct and indirect interactions and factors (Scholes and Archer 1997, Higgins et al. 2000, van Langevelde et al. 2003, Midgley et al. 2010). The suppression of the grass layer, through increased grazing pressure or frequent low intensity surface fires, generally leads to an increase in the woody component (Higgins et al. 2000, van Langevelde et al. 2003). In contrast, pressure on the woody component through increased browsing, more intense fires, harvesting of trees by humans or damage to trees by elephants leads to the conversion of closed woodlands into more open savanna or grassland states (Higgins et al. 2000, Smit 2004, Holdo 2007).

The grass sward generally recovers following a fire event within a single season (Everson 1985, Trollope et al. 1996). The impact of ground-level fire of modest intensity on mature savanna trees is limited. Mature tree mortality due to fire can be as high as 10% in high intensity fires in open savannas (Trollope and Trollope 1996, Casey and Williams 2010), it is generally negligible in closed savanna and woodland systems, where the fire intensity is low since the grass fuel load is small (Trollope and Trollope 1996, Trollope 2007, Ribeiro et al. 2008).

Fire in the grass layer may inhibit the recruitment and survival of saplings forming, a 'fire trap' (Scholes and Archer 1997, Higgins et al. 2000, Trollope 2007, Hanan et al. 2008). Most savanna and woodland tree species have the capacity to resprout and grow rapidly following a fire, using belowground stored resources of water, nutrients and photosynthates (Frost 1996, Luoga et al. 2004). If recurrent fire is sufficiently frequent to scorch the regrowth before it grows taller than the flame zone, the system can be held in an apparently open grassland state for some time (Higgins et al. 2000, Bond et al. 2005). It is therefore appropriate to consider the survival and growth of adult trees (those that have escaped the 'fire trap') separately from saplings (Hanan et al. 2008). This is particularly important when considering the effect of fire on carbon stocks in savanna systems over longer time-scales.

Non-indigenous pine and eucalypt plantations in sub-Saharan Africa are particularly prone to fire as they are often located in seasonally dry fire-prone grassland and savanna systems (de Ronde et al. 2004a). Unlike the indigenous savanna trees, the plantations have a perennial, flammable canopy layer. If this ignites it results in fires of considerable size and intensity that can destroy afforestation projects and their associated carbon stocks: 'stand replacing' fires rather than minor disturbances.

We briefly considering other important African vegetation types in which REDD and reforestation activities may occur. The tropical moist forests of the Congo-Guinea zone are generally too moist throughout the year to burn (Thonicke et al. 2001, Figure 1 [1.1]). The occurrence of a coupled atmosphere-biosphere system modulates rainfall, temperature and specific humidity leading to a climate that is generally too humid to support fire (Zheng and Eltahir 1998, Kiang and Eltahir 1999). Changes in climate or the removal of the canopy layer may lead to the disruption of the coupled atmosphere-biosphere system that may result in the occurrence of hot-dry conditions that support fire. Fires in the Congo rainforest are largely associated with clearing for agriculture. Such climatic and anthropogenic disturbance events are further considered in sections 1.3 and 1.4.

The subtropical thicket of the southern Cape of South Africa is another vegetation type in which carbon storage activities are in the process of being implemented (http://www.africarbon.co.za). Over 80% of the biomass in subtropical thicket is provided by a single succulent tree species, *Portulacaria afra* (Mills and Cowling, 2004). The system is therefore not highly susceptible to fire. This is one explanation provided for the higher biomass stocks in these thickets compared to other ecosystems receiving a similar annual rainfall.

3. Surface *versus* crown fires

Fire requires a reasonably continuous fuel layer to spread laterally. Either the surface or the canopy fuel load therefore needs to be sufficient in quantity, flammability, and ‘packing’ (arrangement in vertical and horizontal space) to sustain fire. Moreover, the fuel load needs to be present during the prolonged dry season during which climatic conditions are suitable for fire.

In grassland and savanna systems where a continuous tree canopy is absent, fires are limited to those that occur near the ground surface, consuming the grass and herbaceous layer (and surface litter) during the dry season. In such systems, grass-layer fuel loads of at least 300 kg/ha are required to sustain fire (van Wilgen et al. 2003). As grass accumulation is closely related to antecedent rainfall, the occurrence of surface fires in southern African grassland and savanna systems is correlated with rainfall in the previous two years (Archibald et al. 2010a).

Crown fires (fires in the canopy of the tree or shrub layer) do occasionally occur in savanna systems, for instance in *mopane* woodland (Kennedy and Potgieter 2003). Flammable leaf matter, especially leaves that are sufficiently dry and have a relatively high content of terpenes, oils and fats, assist in the maintenance and spread of canopy fires (van Wilgen et al. 1990, Ormeno et al. 2009). Crown fires in Africa tend to be limited to the schlerophyllous thicket of the south-western Cape of South Africa known as '*fynbos*' (van Wilgen 1982, van Wilgen et al. 1990) and to exotic pine and eucalypt plantations. Such systems have a fairly closely-packed tree or shrub canopy with perennial and flammable leaf matter (van Wilgen et al. 1990, Ormeno et al. 2009), and in the case of the *fynbos*, a long dry season in summer.

Although fire in such systems, particularly sclerophyllous thicket, may be classified as a ‘surface bush fire’ rather than a true canopy fire such as those located in the upper canopy of tall forest, the effect of fire on the fraction of net carbon stocks released into the atmosphere is similar. The fire is ‘stand replacing’ compared to less intense surface fires: most of the above-ground biomass and associated carbon stocks is consumed (van Wilgen 1982), just as it is in boreal forests (Westerling et al. 2006).

In comparison, the dry, deciduous forests (*miombo* woodland) of south-central Africa may have a near continous canopy, but drop their leaves matter during the prolonged winter dry season (Frost 1996). The presence of a tree canopy in the wet growing season reduces the amount of fuel accumulated by grass and herbaceous understory layer. The surface fires which characterise this system are fuelled by a combination of dry grass and fallen tree leaves and twigs (Shea et al 1996). They are of high frequency but low intensity (Archibald et al. 2010b). They have limited impact on the survival of mature trees unless they are already damaged, for instance by elephants, people or insects (Guy 1989, Trollope 2007, Hanan et al. 2008). The frequency and intensity of fire may change considerably if *miombo* woodland is disturbed (typically by clearing for swidden agriculture or charcoal manufacture). Where the canopy is removed, resulting in a more open savanna state, the grass fuel load can quickly reach up to 10000 kg/ha, since it is too fibrous and low in nitrogen to be eaten by herbivores (Casey and Williams 2010). Potential disturbance drivers in the form of elephants or anthropogenic harvesting of wood are further explored in section 1.4.

In summary, near-surface fires consuming the grass layer predominate in the southern, eastern and Sahelian regions of Africa. Crown fires tend to be limited to the schlerophyllous shrubland of the southwestern Cape, plantations of pines and eucalypts, and occasionally in *mopane* woodlands.

4. Do projected changes in climate alter the fire risk?

Anthropogenic climate change may manifest itself in a number of ways: increases in mean temperature; increases or decreases in rainfall; changes in seasonality; and increases in the frequency and/or intensity of extreme events such as droughts or floods (Boko et al. 2007). Underlying the changes in climate is an increase in the concentration of atmospheric carbon dioxide, which has direct effects on plant growth that are different in trees (C3 photosynthetic system) and tropical grasses (C4) (Bond et al. 2003a, Kgope et al. 2010). Each of these changes has potential consequences for the fire regime.

The creation of a fire season, or the extension of an existing fire season, has widely been identified as a particular impact of projected climate change in systems where warming is accompanied by drying and there is at least sufficient rainfall to generate a base fuel load to sustain fire (Thonicke et al. 2001, Pierce et al. 2004, Westerling et al. 2006, Pechony and Shindell 2010). A global-scale analysis by Pechony and Shindell (2010) indicates that increases in temperature and drying may lead to an increase in fire after ~2050. Note that in general, warming is associated with increased rainfall, but this may not be true everywhere, or in all seasons. In southern Africa, there is spatial variation in the both the sign and magnitude of the anticipated rainfall change (drier in the south and west, wetter in the north and east, by about 20% in both cases) and variation in the magnitude of temperature change (lower near the coast and higher inland, overall southern Africa is projected to warm at about twice the global rate (Scholze et al. 2006, Engelbrecht et al. 2008, Pechony and Shindell 2010). In addition there is considerable disagreement between models regarding both the location and magnitude of the changes (IPCC 2007).

Regionally-specific ecosystem responses result in a range of potential outcomes even from the same forcing. For example, in the boreal forests of Yellow Stone National Park in North America, antecedent rainfall in the previous two years has a strong *negative* influence on annual burnt area (Balling et al. 1992); whereas van Wilgen et al. (2003) and Archibald et al. (2009, 2010a) found a *positive* relationship between rainfall in the previous two years and annual burnt area in savanna systems in southern Africa. In mesic North America the fuel load is generally sufficient to sustain fire, and of the fire activity is controlled by the length of the burning season (Balling et al. 1992, Westerling et al. 2006). In the savannas and woodlands of southern and eastern Africa, the availability of fuel rather than the length of the burning season is the key determinant (Archibald et al. 2009, 2010a). In the Kruger National Park (a representative savanna area of two million ha) an average of 44,149 km^2 burned during sequences of above-average rainfall years (mean annual rainfall of 638mm) versus 20,834 km^2during sequences of dry years (rainfall 450mm). This corresponds to an equivalent fire return period of 4.3 vs 9.1 years (van Wilgen et al. 2003).

Assessment of the impact of climate change on the risk of fire to a REDD or reforestation activity, therefore requires the use of climate models that correctly capture regional-scale climate patterns, and ecosystem models that consider system-specific dynamics. In savannas it is important to consider the differential effect of rising atmospheric carbon dioxide ([CO2]) on tree and grass growth (Bond and Midgley 2000, Bond et al. 2003a, Kgope et al. 2010). Changes in annual rainfall, grass fuel loads and associated fire frequency and intensity may change the size and nature of the 'fire trap' (Higgins et al. 2000, Hanan et al. 2008), thereby either inhibiting or improving the likelihood of the survival of tree saplings. At the same time an increase atmospheric carbon dioxide is predicted to increase the relative growth rate of C3 woody plants compared to C4 grasses thereby improving the probability of trees escaping the fire trap, leading to increased tree cover over time (Kgope et al. 2010).

Most southern African ecosystems are pre-adapted to fire. The carbon stocks are already below their climatic potential due to fire and herbivory (Sankaran et al. 2005). Thus marginal changes in the fire regime are likely to have marginal consequences to carbon stocks. This is not true where projected changes in climate may lead to the occurrence of fire where it was previously absent or rare. Fire risk to REDD and reforestation projects is greater where the changes in climate lead to fire occurring in moist forests (where woody biomass are considerable, approximating their climatic potential) and where fire events are likely to be a 'stand replacing' (Figure 1, 1.3). The 1997-1998 Indonesian and Amazonian forest fires are examples of where the introduction of a fire into a typically moist system, not exposed to fire in recent history, leads to considerable loss of carbon and the emission of additional GHGs (Page et al. 2002, van der Werf et al. 2004). Moreover, the recovery period of carbon stocks in moist forest to pre-fire levels is likely to be considerably longer than the 20 year lifespan of a REDD activity. The change in local climate resulting from a fire may lead to further fire events, keeping the moist forest in an alternative state with reduced carbon stocks.

It is therefore prudent to assess the risks of projected changes in climate and associated changes in observed fire regimes prior to the implementation of both project-scale as well as national-scale REDD and reforestation activities. Anthropogenic climate change is unlikely to increase fire risk universally. In sub-Saharan Africa fire activity is generally predicted to increase in semi-arid savanna and woodland systems such as the Sahel, southern and eastern Africa, but decrease in tropical areas such as the Congo-Guinea Moist Forest belt and the south-east African coast (Thonicke et al. 2001, Scholze et al. 2006, Pechony and Shindell, 2010).

5. The effect of human activities on fire risk

There is a long history of humans directly or indirectly influencing fire regimes (Bowman et al. 2009). Of concern in relation to the risk of fire to REDD and reforestation ventures are human activities that may increase the impact of fire on carbon stocks over the lifespan of a project- or national- scale activity. An example is the opening up of the tree canopy, either through commercial logging or clearing of land for agriculture or fuel collection, which leads to an increase the probability of fire in systems where it was previously generally absent (Cochrane and Laurance 2002, Page et al. 2002, Veldman et al. 2009).

In moist forests of the Indonesia, South America and the Congo-Guinea belt, the removal of the canopy layer uncouples the atmosphere-biosphere system, allowing the subcanopy strata to dry out sufficiently to burn (Zheng and Eltahir 1998, Kiang and Eltahir 1999, Cochrane and Laurance 2002, Page et al. 2002). In an analysis of the spatial extent of fires during the dry spell associated with the 1997 El Niño event in Indonesia, Page et al. (2002) noted that only 4.5% of the pristine forest was burnt compared to 29.2% of previously logged forests and 70.0% of fragmented land. In addition to the initial release of carbon to the atmosphere, open areas may be maintained in an alternative open stable state through fire over time (Veldman et al. 2009).

The removal of the tree layer and canopy may also lead to a change in the nature of an existing fire regime. Fires in intact dry deciduous *miombo* woodland of southern and eastern Africa are typically low intensity fires that consume a limited grass and litter layer (Frost 1996, Ribeiro et al. 2008). The removal of the canopy layer leads to an increase in the grass layer which in turn leads to a larger fuel load and more intense fires which may in turn decrease tree recruitment and survival (Frost 1996, Casey and Williams 2010). This may result in an alternative, low-carbon stable state.

Although not directly an anthropogenic disturbance, elephants can have a similar effect on ecosystem structure, fire regimes and the impact of fire on carbon stocks. Elephants push over mature trees, leading to an increase in grass cover, fuel loads and the intensity of surface grass fires (Laws 1970, van Wilgen et al. 2003, Holdo 2007, Ribeiro et al. 2008). The combined effect of elephants knocking over larger trees and intense grass fires that inhibit tree recruitment, may lead to the gradual opening up of woodland, savanna and even moist forest systems (Laws 1970, Guy 1989, van Wilgen et al. 2003, Holdo 2007).

Land-use changes may also lead to a *decrease* in the frequency and/or intensity of fire. Cultivation and the establishment of infrastructure fragments the landscape and reduces fuel load quantity and connectivity, which in turn leads to more but smaller fires. Archibald et al. (2010b), note the occurrence of large fire is sharply reduced above a population density threshold of about 15 people per km^2.

Livestock farming, particularly continuous grazing by cattle at stocking rates close to the maximum, suppresses both the competitiveness of grass and the accumulation of grassy fuels, leading to decrease in fire intensity, and in commercial ranching areas to a decrease in fire frequency as well – both contributing to an increase in woody cover (Higgins et al. 1999, Smit 2004). Subsistence pastoralists in nutrient poor grasslands and savannas set fire to the grass to remove moribund material and stimulate new, more nutritious growth. This may increase fire frequency, but the reduced fire intensity may still lead to an increase in the woody cover over time (Higgins et al. 1999, Midgley et al. 2010). Thonicke et al. (2001) note that such burning by pastoralists leads to a notably higher frequency of fire in the savanna and woodland systems of the Sahel, southern and eastern Africa, than would be expected. Although this phenomenon (‘bush encroachment’) may increase above-ground carbon stocks, it is not a preferred outcome because of the negative effects on biodiversity, livestock production and possibly water services.

In summary, in terms the main process through which anthropogenic disturbance may increase the risk of fire in REDD and reforestation ventures is through the opening up of the tree canopy in moist and dry forest systems, which in turn potentially leads to an increase in the frequency and intensity of surface fires and the potential for the system to become trapped in a an open state. This phenomenon may be of particular importance if the harvesting of timber were to be allowed within future REDD policies, and resulted in large, interconnected gaps. If the canopy is removed, fire may inhibit reforestation if it is not managed appropriately (Veldman et al. 2009).

6. Quantifying fire risk to REDD activities in key African ecosystems

When estimating the risk fire presents to carbon stocks and the uncertainty it leads to in the expected outcomes of a REDD activity, the impact of fire on the total carbon stocks of the system must be considered. Total carbon stocks include not only the carbon in above-ground biomass but also below-ground carbon in root and soil organic matter. Current protocols tend to focus on the aboveground wood stocks only, but the belowground changes are of equal or larger magnitude in savannas. Likewise, an important part of estimating the long-term impact of fire on the carbon stocks is the rate of recovery of carbon stocks relative to the duration of the REDD activity (nominally 20-30 years) and whether the system has the innate ability to recover to pre-fire conditions (and associated carbon stocks) or if one fire event can lead to a degradation of the system from a carbon storage point of view that persists beyond the project timeframe. Finally, if the degradation is in this sense persistent, is it practically possible to accelerate the return the system to the pre-fire state through management interventions and is it financially feasibility to do so?

The conceptual model of multiple alternative stable states, each with a degree of resilience, stability and persistence is useful for understanding the risk of fire to REDD activities. Holling (1973) and May (1977) showed that non-linear systems, which ecosystems typically are, could occupy more than one stable state. Alternative states are separated by thresholds, at which point the system exhibits a large change from one state to another that is not spontaneously reversed. The net effect of the ecosystem interactions is a positive feedback loop that keeps a system in a particular state until an external driver of change causes the it to cross a threshold into another state (Scheffer and Carpenter 2003, Walker and Meyers 2004). The 'resilience' of a system was originally defined as 'the ability of a system to absorb change in state variables, driving variables, and parameters, and still persist' (Holling 1973). The 'stability' of a system should not be confused with its resilience, and neither has a direct relationship to the constancy of the state variables. Stability 'is the ability of a system to return to an equilibrium state after a temporary disturbance' (Holling 1973). While a system may fluctuate substantially from year to year (and be viewed as unconstant), but may still be stable in the sense of still being attracted to a notinal equilibrium, and resilient in the sense that it is still fundamentally the same system. The 'persistence' of a particular state is most usefully defined relative to the requirements and perspective of the end user (Scholes 2009). If for example, the end-user is a REDD project manager, a decrease in carbon stocks that lasts longer than 20-30 years following a fire event would be viewed as a 'persistent decrease' in carbon stocks.

In this analysis we shall 'regime' as an inclusive term for the 'state' of an ecosystem (i.e. the size of the carbon store in its various pools, in this case) and the associated characteristic the disturbance pattern, in a similar manner to Scheffer and Carpenter (2003) and Walker and Meyers (2004). We do not wish to be sidetracked into cause-and-effect arguments regarding whether a 'true' alternative stable state has been entered, or merely a persistent subregion of a larger state. In particular we focus on (i) what fraction of the net carbon stocks of the system are affected by a typical fire event? (ii) How long does it take for carbon stocks to recover to pre-fire levels? (iii) Is there a potential for a fire to lead to a new set of positive feedbacks, restricting the recovery to the former condition (i.e. are the carbon stocks resilient to fire)? (iv) Does slow recovery or positive feedback mean a decrease in carbon stocks that persists beyond the duration of the project? and (v) Can the risk of fire to the project be realistically reduced, and if fire occurs can recovery be accelerated?

Infertile grasslands

In these grasslands (known colloquially as 'sourveld'), the nitrogen content of the grass sward declines after the spring or post-fire flush of green shoot to such a degree that for many months it is below the digestion threshold of ruminants (about 0.8% N).

The ungrazed dead grass accumulates, sometimes for several years, until it is burned. Following a fire, the above-ground grass biomass of 1 to 4 tonnes dry mass/ha (0.5 to 2.0 $tC.ha^{-1}$) recovers within a year or two, depending on rainfall, soil nutrient availability and grazing pressure, (Everson 1985, Manson et al. 2007, Table 1). Although most of the carbon located in above-ground biomass is released into the atmosphere through a fire event, it represents a small fraction of the net carbon stocks of the system – the belowground carbon in soil organic matter and roots is an order of magnitude or more larger.

In a case study to assess the potential to sequester additional carbon either through improved grassland management or to reduce GHG through avoided grassland degradation, Knowles et al. (2008) found the total soil carbon in an infertile grassland in southern Africa to range from 80 to 140 $tC.ha^{-1}$ to a depth of 30 cm (Knowles et al. 2008, Knowles and Pienaar 2010). Above-ground carbon stocks therefore only represent 1-4 % of the net carbon stocks of the system and it generally recovers within a year following fire (Everson 1985, Manson et al. 2007). Long-term field experiments in the high-altitude grasslands of the Drakensberg Mountains of South Africa indicate that frequent burning has little effect on below-ground carbon stocks and may even lead to a slight increase in soil carbon stocks in certain cases (Manson et al. 2007). In contrast, the Knowles et al (2008) study estimated that that intensive, continuous grazing regimes lead to a loss of 50 – 60 $tC.ha^{-1}$.

Although the above-ground carbon pool in infertile grasslands may therefore fluctuate considerably on an annual time scale, the total carbon stock is resilient to fire over a 20-30 year period. The risk of fire to additional carbon sequestration or avoided soil carbon degradation activities in infertile grassland systems is therefore estimated to be low (Table 1).

Fertile grassland

In contrast to infertile grasslands, the grass sward of fertile grasslands (‘sweetveld’ in Southern Africa) maintains its nitrogen content and nutritional value at a level above the minimum threshold for ruminant digestion through the season (van Wilgen and Scholes 1997). Grazing pressure therefore tends to continue through the season inhibiting the build-up of a significant grass fuel load. Fires therefore tend to be infrequent, or low-intensity, and typically occur following exceptionally high-rainfall years.

The above-ground carbon pool is an even smaller fraction of the net carbon stocks of the system than in infertile grasslands, because the soils are typically more clay rich, with a higher carbon store. The net carbon stock of fertile grassland systems is resilient to fire over a 20-30 year period. Fire is therefore not predicted to be a considerable risk to the outcome of an avoided grassland degradation activities (‘very low’, Table 1).

Moist savanna (infertile and broad-leafed)

Savannas are one of the most extensive and populated vegetation types in sub-Saharan Africa (Clarke et al. 1996, van Wilgen and Scholes 1997). Among the wide range of savanna varieties there are two distinct savanna types, namely the ‘moist, infertile broad-leafed’ and ‘dry, fertile fine-leafed’ savanna (Scholes and Walker 1993).

The infertile savannas, which include the Miombo woodlands experience fires every one to four years on average (Scholes et al. 1996, van Wilgen et al. 2003, Walker and Desanker 2004, Williams et al. 2008), for the same fundamental reason that the infertile grasslands burn frequently. The *miombo* woodlands in particular are a target area for national-scale REDD activities in countries such as Tanzania, Zambia and Mozambique (Kamelarczyk 2009, Burgess et al. 2010), and are the location of early pilot climate change mitigation activities in the form of community-based REDD, reforestation and energy-efficient cook stove initiatives (Williams et al. 2008, http://www.envirotrade.co.uk).

Fires in infertile savanna systems are typically in the form of a surface burn. The fires may inhibit the recruitment of saplings to a certain extent but the impact of fire on mature trees, where the majority of the above-ground carbon biomass pool is located, is limited (Guy 1989, Frost 1996, Trollope 2007). Some 95 -98 % of the above-ground biomass in Miombo woodlands is located in the woody component (Frost 1996). The carbon in the grass layer in savanna systems is 1 to 2 $tC.ha^{-1}$ (Trollope et al. 1996). Woody above-ground biomass ranges from 10 $tC.ha^{-1}$ in semi-arid savannas in South Africa (Scholes 1987, Shackleton 1997) to 45 $tC.ha^{-1}$ in the moist Miombo woodlands in northern Zambia and Mozambique (Chidumayo 1990, Frost 1996). Between 50 and 80% of the net carbon stocks in Miombo woodland systems are found in the top 1.5m of soil (Walker and Desanker 2004). To generate a conservative estimate for risk assessment purposes, a root:shoot ratio of 0.3 was applied to calculate below-ground woody root carbon (3-14 $tC.ha^{-1}$, Mokany et al. 2006, Table 1). Soil organic carbon, which includes grass root carbon, is conservatively estimated at 30 $tC.ha^{-1}.30cm^{-1}$ assuming a soil carbon content of 0.6 % (Scholes et al. 2003) and a bulk density of 1.7 $Mg.m^{-3}$ (Venter 1990 and McKean 1993 in Wise et al. 2009). Fire has little effect on below-ground carbon stocks or nutrient-availability (Coetzee et al. 2008).

If the lower bound of the woody and below-ground carbon pool estimates is used, a fire event is likely to result in up to 2% of the net carbon stocks of the system being released into the atmosphere (Table 1). The 2% that is located in above-ground grass biomass generally recovers in one to two years (Trollope et al. 1996). The total carbon stock of the system is resilient to fire over a 20-30 year period, resulting in a low risk of fire to REDD activities (Table 1).

More of a concern regarding variability in expected carbon stocks at the end of a 20-year period are substantial shifts in observed regimes, which may lead to a change in the net carbon stocks of the system that may be more considerable. The scenario described above, where surface fires remove the grass fuel load with limited impact on the woody component, assumes that while the woody to grass biomass ratio may fluctuate slightly, the system remains fundamentally a savanna – a quasi-stable tree-grass mixture. If human- or elephant-driven disturbances pushes the system into an alternative grass-dominated regime it may become stuck for periods longer than the typical project lifetime. More intense fires inhibit tree recruitment and maintain the system in the open grassland regime, which has considerably lower net carbon stocks compared to a closed canopy state (Casey and Williams 2010).

The regime shift is not necessarily permanent. It may be reversed through management interventions such as the implementation of more frequent but lower-intensity burns (de Ronde et al. 2004b), or by fire protection. Due to the long history of fire in savanna, the tree species have evolved means of rapidly resprouting (coppicing) following a disturbance (Hoffmann et al. 2003). The woody component and associated carbon stocks are therefore likely to recover if the 'fire trap' is reduced for a few years.

In summary, the impact of fire by itself on the variability in the net carbon stocks of moist, infertile savanna ecosystems with a 20 – 30 year REDD activity is likely to be limited (Table 1). If an external driver (typically harvesting of trees by humans) does push the system into an open state, it is generally possible to manage the system back towards the closed state. The risk of this circumstance may be further reduced by the REDD project implementer through appropriate burning and timber harvesting practices that maintain woody cover at required levels.

Arid, fertile, fine leafed savannas

Fine-leafed *Acacia* species characterize the arid fertile savannas of southern Africa. Compared to infertile savannas, the grass sward maintains its nutritional content leading to a greater fraction of grass biomass being consumed by herbivores and a smaller, more patchy fuel load being present at the end of the season (van Wilgen and Scholes 1997). Fires therefore tend to burn less frequently and less intensely, presenting less of a risk to the net carbon stocks of the system compared to moist infertile savannas.

An exception are the Mopane woodlands, dominated by the species *Colophospermum mopane* (Mlambo et al. 2005). Mopane has broad, flammable leaves (van Wilgen and Scholes 1997). Where the tree cover is sufficiently dense a form of canopy fire can occur. Such fires do not kill the trees, but consume the aboveground parts in direct proportion to the intensity of fire and inverse proportion to the size of the tree. Fires hardly affect tall trees (Mlambo and Mapaure 2006).

In the case of particularly intense fires, 'tall Mopane' may be converted into a 'shrub' state about 3 m tall (Mlambo and Mapaure 2006). While there is a significant decrease in above-ground biomass, the trees are not necessarily killed but coppice soon after the fire event allowing the above-ground woody component to naturally recover (Kennedy and Potgieter 2003, Mlambo et al. 2005). Repeated frequent fires may maintain Mopane in the shrub form for some time. Inter-stem competition between mopane shrubs is intense and may help to trap the vegetation in this low, multi-stemmed state (Scholes 1990, Sea and Hanan, 2011). A decrease in fire frequency would allow the woodland to recover to a tall state and stem-thinning may accelerate a return to a tall mopane state less susceptible to fire (Mlambo and Mapaure 2006).

The above-ground woody carbon pool in Mopane woodlands may therefore be relatively variable over a 5-15 year period due to fire. Whereas mopane woodlands may be resilient to fire over the long-term (>30 years), a fire in the latter half of a 20-year project may lead to a 5-15% decrease in the expected net carbon stocks of the system at the end of a REDD activity lifespan. For this reason, the risk of fire to REDD activities located in mopane woodland is estimated to be low to moderate (Table 1), whereas the risk in *Acacia* savannas is considered low.

Moist evergreen forest

A positive feedback loop in the form of a coupled atmosphere-biosphere system maintains intact moist forests in a state that is generally too humid to burn. Fire therefore presents little risk to the outcome of a REDD project located in intact moist forest (Table 1).

However, as explored in section 1.3 and 1.4, the coupling may be disrupted through human- or climate change disturbance. This allows fire to occur in a system where it was previously absent resulting, in a considerable release of carbon into the atmosphere (~70% of the net carbon stocks of the system, Table 1). Furthermore, if subsequent fire is permitted, the system may remain in an open regime and not naturally recover. Even if fire were to be controlled and the forest allowed to regenerate, the period of time required for carbon stocks to recover (>50-100 years) is longer than a typical REDD activity (20-30 years) and may thus result in a considerable decrease in expected carbon stocks at the end of the project's lifespan ('high risk', Table 1).

If changes in climate lead to the creation of prolonged dry periods the risk of fire would increase to a point where it may be difficult to keep the fires out. While it is possible to manage fire physically in the context of small-scale commercial plantations, the practical and financial feasibility of halting fire over large national spatial scales and decadal time periods is questionable. The system is therefore likely to transform patch-wise into an open state with an associated persistent reduction in net carbon stocks over time. It would therefore be appropriate to assess the effect of projected changes in climate on fire risk prior to the implementation of both project- and national-scale REDD activities in moist forests.

Sclerophyllous shrubland

The sclerophyllous shrublands ('fynbos') of the south-western Cape of South Africa are a particularly fire prone and fire dependent system (Heelemann et al. 2008). Many fynbos species rely on a fire event to stimulate germination (Heelemann et al. 2008). It is also one of the few systems in sub-Saharan Africa that have a dense, contiguous canopy in the prolonged, hot, dry, summer burning season. leading to intense, 'stand replacing' fires (van Wilgen et al. 1990). The fire return period in fynbos systems is between ten to thirty years (van Wilgen and Scholes 1997) and the above-ground carbon stocks of mature fynbos are 25 to 36 $tC.ha^{-1}$ (van Wilgen 1982).

Biomass stocks in fynbos are therefore relatively variable at a REDD project timescale. Although the system is likely to recover naturally and not be held in an alternative regime, there may be considerable variation in the realised size of the carbon pool after 20-30 years. The risk of fire to the outcome of REDD activities located in sclerophyllus thicket systems may therefore be considered moderate to high (Table 1).

Subtropical thicket

A number of reforestation activities are currently being developed and implemented within the semi-arid subtropical thicket of the Eastern Cape. Historical Angora goat farming practices led to the widespread degradation of the vegetation type (Mills et al. 2005). A large proportion of the biomass in sub-tropical thicket is comprised of a succulent tree species, *Portulacaria afra*. Fire is therefore rare within the subtropical thicket presenting little risk to the outcomes of REDD or reforestation activities in this ecosystem (Table 1).

Pine and eucalypt plantations

A number of exotic plantations that are dependent on carbon revenues for economic viability have recently been established in fire prone grassland and savanna systems in sub-Saharan Africa (http://www.climate-standards.org/projects/index.html). Such plantations are susceptible to canopy fires due to the presence of flammable leaf fuels during the prolonged dry fire season. Furthermore, the fires are likely to be intense 'stand replacing' events that remove the greater portion of above-ground biomass.

The below-ground root biomass in pine plantations may be vulnerable to fire as well (Table 1).

Although certain *Eucalyptus* species may coppice following fire, fire may lead to considerable variation in expected carbon stocks at the end of a 20-year project period if not adequately managed. Although such active fire management is practiced widely in countries like South Africa (http://www.workingonfire.org/), the decision of whether plantation afforestration is viable as a carbon sequestration project depends on the relative cost of fire management, the effectiveness thereof in reducing variability in carbon stocks over 20-30 years, and revenues generated through the sale of emission reduction units. In general, the risk of fire to REDD activities involving pine and eucalypt plantations is estimated to be moderate to high (Table 1).

7. Managing fire risk using financial techniques

In addition to the physical management of fire, the impact of fire on investment in REDD and reforestation activities may be managed through the use of financial techniques such as insurance, portfolio theory and risk-adjusted payments for generated ERUs. Risk-adjusted payments is where the price paid for ERUs is reduced relative to perceived risk of delivery at Year 5, 10, 15 and 20.

Portfolio theory, based is where a suite of projects are bundled together into a portfolio with the intention of reducing the net exposure of the investment to risk. The key underlying assumption is that of 'efficient diversification of risk', where a particular suite of projects is identified based on the manner in which risk variables co-vary between investments (Bodie et al. 1996, Reilly and Brown 2003, Rubinstein 2002). It is not sufficient to bundle a number of projects together if they are all susceptible to the same risk factors at the same time. The key is to bundle projects that do not co-vary in risk exposure. Insurance also uses the portfolio approach, but the risk is borne by the insurer rather than the project, in return for a fee.

The manager of a portfolio of REDD projects would therefore need to assess co-variation in the probability and intensity of fire across a suite of projects as well as the recovery period of carbon stocks relative to the length of the REDD activity. An understanding of the underlying ecology of fire and the resilience of carbon stocks to fire as described in this paper is required to appropriately estimate the risk of fire to the outcome of a particular project. An estimation of co-variation in the probability of fire could be calculated from historical fire-scar data (Justice et al. 2002, Clerici

2006). The impact of a fire event on the net carbon stocks could be based on past observations, data and literature (e.g. Table 1) and the recovery rate of carbon stocks could be estimated using a carbon accumulation model such as the Century Ecosystem Model (Parton et al. 1993).

Conclusion

Although the majority of grassland, savanna and woodland vegetation types in sub-Saharan Africa are prone to fire, over the 20-30 year period of a REDD project, the net carbon stocks of the systems may be relatively resilient to fire (Table 1). Anthropogenic harvesting (and possibly distance by large populations of elephant) may lead to a decrease in above-ground carbon stocks and more intense fires. It is generally possible to manage the system back to a prior woody state within a 20-30 year period.

We have also made the assumption that fire would affect the entire project area equally. If fire does occur, it may only affect a certain portion of the project area dependant on the spatial extent of the project, the heterogeneity of the landscape, habitat fragmentation and fire management. This is especially true in the case of 'community-based' REDD projects located in populated landscapes that may be fragmented due to cultivation, roads and prescribed burning for grazing requirements.

More of a concern is the risk of fire to REDD and reforestation projects in systems that have historically not been prone to fire, but where anthropogenic disturbances or changes in climate may lead to the occurrence of fire. This can lead to a regime shift to more open, sparse vegetation with a considerable reduction in the net carbon stocks of the system. Full recovery to the prior state is unlikely within the project period.

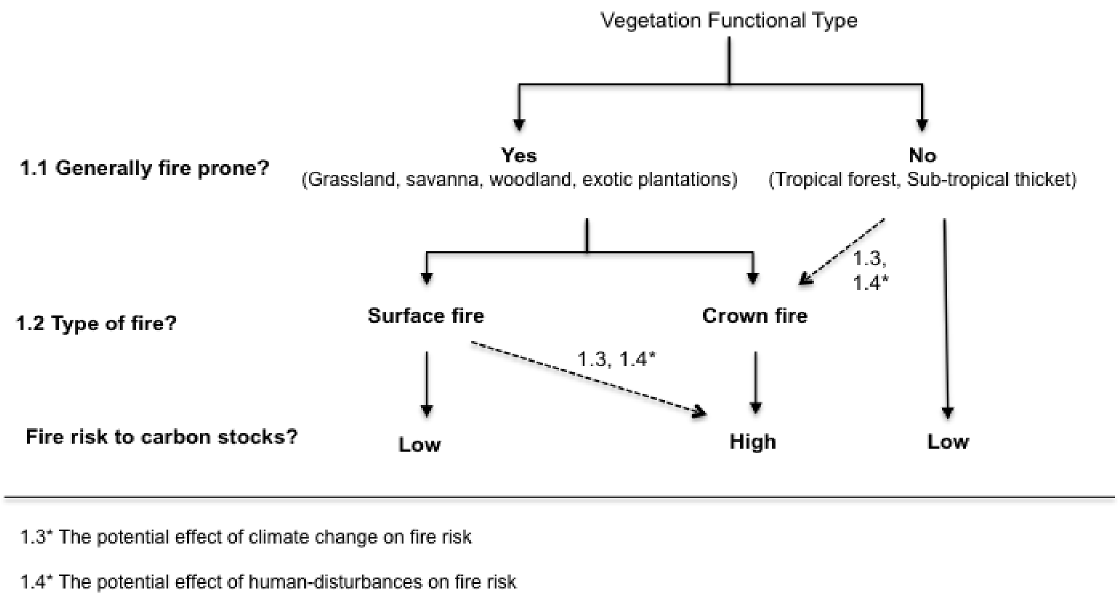

Figure 1: A step-wise classification of the risk of fire to REDD activities located in predominant African vegetation types. The numbering (e.g. 1.1) corresponds to the section of discussion in the text.

Table 1: A list of above- and below-ground carbon stocks in predominant African vegetation types with a view to estimating the risk of fire to the outcome of a REDD activity over a 20-30 year period.

Vegetation type	Fuel type and load	Fire frequency	Carbon stock ($tC.ha^{-1}$)		% Released through a typical fire		Recovery period	**Fire risk**	References
		Return period	Above-ground	Below-ground (30cm soil depth)	Above-ground	Below-ground		**Over 20 years**	
Grassland - infertile	Surface grass fire	1-2 years	Grass: 0.5-2.0	80-140**	100%	0%	1 year	**Very low**	Manson et al. 2007, Knowles et al. 2008
Grassland - fertile	Surface grass fire	1-4 years	Grass: 0.5-1.0	80-140** (Ask Bob)	100%	0%	1 year	**Very low**	Scholes et al. 1996
Moist infertile savanna and woodland	Surface grass fires	Mean: 4.5 years (range 1-34).	Grass: 1-5 Woody: 10-45	Woody: 3-14* Soil: 30**	Grass: 100% Woody: ~5%	0%	1- 3 years	**Low**	Scholes 1987, Frost 1996, ++
Mopane woodland (N. Kruger National Park)	Canopy bush fire	>8 years §	Woody: 11.5	Woody: 3.5* Soil: 30**	Woody: 30-50%	0%	< 10 years	**Low-moderate**	Scholes 1987, Mlambo and Mapaure 2006
Forest - moist evergreen (intact)	n/a	500 years	Woody: 160-260	Woody: 32-52* Soil: 45	0%	0%	>100 years	**Low**	Baccini et al. 2008, Batjes 2008
Forest - moist evergreen (disturbed)	Canopy bush fire	<10 years	Woody: 160-260	Woody: 32-52* Soil: 45	80-100%	0%	>100 years	**High**	Page et al. 2002, Batjies 2008, Veldman et al. 2009
Sclerophyllus thicket - fynbos	Canopy bush fire	10- 30 years	Woody: 25-36	Woody: 7.5-10.8* Soil: 70-80	100%	0%	20-40 years	**Moderate-high**	Van Wilgen 1982, van Wilgen and Scholes 1997,
Arid shrubland – sub-tropical thicket	n/a	n/a	Woody: 42	Woody: 26 Soil: 133**	0%	0%	n/a	**Very low**	Mills et al. 2005
Exotic Pinus and Eucalyptus plantations	Canopy fire	~100 years	Pinus: 20, 26@ Euc: 121, 23@	Pinus: 3, 4@ Euc: 18, 3@	100%	Pinus: est.-100% Euc: 0%	10-25 years	**Moderate-high**	Christie and Scholes 1995, Scholes et al. 1996

* Calculated using generic root:shoot ratios based on the review of Mokany et al. (2006). Root:shoot ration of 0.2 for moist forest trees, 0.3 for savanna trees

** The soil carbon estimate includes both soil organic content and grass fine roots; *** Calculated using a root:shoot ratio of 1.2 (Jackson et al. 1996)

++ Trollope et al. 1996, Shackleton 1997, Scholes et al. 2003, van Wilgen et al. 2003, Walker and Desanker 2004

§ Zambatis and Ratnam, KNP Scientific Services - unpublished, in Sankaran et al. 2005 supp data; @ Saw timber and pulp wood forests respectively

Chapter 6

Do projected changes in climate present a risk to land use based climate change mitigation activities?

Rainfall and temperature are two key determinants of plant growth through their influence on plant available moisture. A change in either the magnitude or seasonality thereof has the potential to result in a change plant growth, associated biomass accumulation and the amount of carbon sequestered in woody biomass over the lifetime of a REDD or reforestation activity. In addition, predicted increases in atmospheric carbon dioxide may also lead to a change in the growth rate of plants, particularly those with a C3 photosynthetic system.

The key question this paper therefore seeks to answer is: Do projected changes in temperature, precipitation and atmospheric CO_2 present a risk to the outcome of REDD activities?

The idea for the paper emerged through discussions with my supervisor Dr Bob Scholes. Dr Bill Parton of Colorado State University kindly provided tuition and support on the use of the Century Ecosystem Program. Following completion of the modeling and draft text, the final text and presentation of results has been significantly improved through advice and input from Dr Albert van Jaarsveld, Dr Allan Ellis and Dr Bob Scholes.

Do projected changes in climate present a risk to land use based climate change mitigation activities?

Tony Knowles, Robert. J. Scholes, Albert. S. van Jaarsveld and Allan Ellis

Abstract

There is good reason to assume that projected changes in climate will affect the outcome of land-use based climate change mitigation and adaptation activities. To estimate the potential affects climate change might have, we modeled the influence of projected changes in atmospheric carbon dioxide ([CO2]), temperature and rainfall on the outcome of avoided deforestation and forest degradation (REDD) as well as reforestation activities located in southern Africa.

The results indicate that the combined effect of elevated [CO2], and changes in temperature and rainfall could lead to a 3 - 48% increase in standing carbon stocks and a 7 – 32% increase in carbon sequestration rates. When modeled alone, expected changes in temperature and rainfall generally lead to positive increases in carbon stocks and sequestration rates. There is however significant variation in responses between different vegetation types, which may be contributed to site-specific degrees of ecosystem water and soil nitrogen availability.

The effect of elevated [CO2], when modeled alone is less clear, ranging from a -10 to +4% change in carbon stocks. Although this may appear to be counterintuitive, such variability in the direction and magnitude of the response is in line with the results from contemporary *in situ* studies on complex plant communities.

For the vegetation types and sites modeled, changes in [CO2], temperature and rainfall present little downside risk. While these results are location specific, the methods used in this study provide a means of estimating climatic risk in land-use based climate change mitigation activities.

Introduction

Global awareness of the potential contributions the land-use sector can make to global climate change mitigation and adaptation efforts has risen sharply since the publication of the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Reports in 2007 (IPCC 2007). From being a peripheral issue during the compilation of the Kyoto Protocol and associated Clean Development Mechanism (CDM, UNFCCC 1998), land-use sector mitigation and adaptation has taken a prominent position in post-2012 climate change policy negotiations. Not only did the IPCC Fourth Assessment Reports highlight that 20 percent of anthropogenic emissions are generated through land-use change, particularly deforestation, but the Report also drew attention to the crucial importance of land-use and ecosystem services in global efforts to adapt to climate change (Boko et al. 2007; Denman et al. 2007; Smith et al. 2007).

Considerable energy and funding is therefore currently being invested in the development of activities that reduce emissions from deforestation and degradation (REDD) as well as carbon sequestration (CS) through the restoration of forests and natural ecosystems (Angelsen et al. 2009). Since the adoption of the ‘Bali Roadmap’ at 13th Conference of Parties meeting in December 2007, significant progress has been made towards designing implementation options on a national scale in anticipation of the broader acceptance of REDD activities and land-use climate change mitigation in post-2012 policy (Angelsen et al. 2009; Streck et al. 2009).

There is good reason to assume that climate change may itself affect forest and rangeland systems and potentially the success of planned climate change mitigation and adaptation activities (Aaheim et al. 2010). Extensive literature shows that projected changes in climate are likely to affect the biota of the globe in terms of, for example, species extinctions, shifts in species ranges, changes in primary production and prevalent fire regimes (Ojima et al. 1996; Peng and Apps 1999; Thomas et al. 2004; Boko et al. 2007; Potvin et al. 2007; van der Werf et al. 2008; Rosenzweig et al. 2008; Doherty et al. 2010; Kgope et al. 2010).

In the context of land-use climate change mitigation and adaptation activities, the effect of climate change on existing terrestrial biomass carbon pools and the rate of change in such pools, is of particular interest. As the success of REDD and CS activities are based on the amount of carbon maintained (REDD) or increased (CS) in the forest carbon pool, potential variability in projected carbon stocks and therefore, the generated emission reductions units, can have implications in terms of investment risk, the calculating of additionality and the broader acceptance of land-use based activities within mitigation policy.

Such variability or perceived uncertainty in the outcome of REDD or CS activities, is one of the prime reasons why policy negotiators and funders have shied away from more widespread inclusion of land-use based climate change mitigation to date. Compared to industrial sector emission reduction activities, where known engineering technologies are employed and delivery risk is quantifiable; forest and rangeland management over decadal time-scales currently present uncertainties in terms of the permanence, long-term governance and the effect of changes in biophysical drivers such as rainfall, temperature and fire (Baer 2003; Subak 2003; Cairns and Lasserre 2004; Miles and Valerie 2008; Aaheim et al. 2010).

A key factor for the greater acceptance of REDD and CS as a practical means of mitigating climate change is the need to quantify sources of uncertainty and perceived risk. In a similar manner to industrial engineering projects, risk parameters for land-use based interventions need to be measurable and manageable in order for the sector to be broadly included in policy and funding options. While the end-goal is to reduce risk to acceptable levels, the first step is to estimate the frequency and impact of sources of variability, making them a known risk variable that policy makers and funders can consider in an appropriate manner. This study presents an approach towards quantifying the risk that climate change itself presents to REDD and CS ventures and forms part of the greater need to estimate the risk profile of land-use based climate change mitigation activities.

As an example of how climate change may affect the outcome of REDD and CS activities, this study assesses the effect of climate change on a suite of predominant vegetation functional types (VFTs) in sub-Saharan Africa over the next 20 years. From a range of climatic variables that are expected to change due to anthropogenic activities such as atmospheric carbon dioxide ([CO2]), temperature, precipitation, ozone, UV-B radiation and the frequency of extreme weather events such as exceptional storms (Ojima et al. 1996; Scholes et al. 1997a; Zepp et al. 2003; Felzer et al. 2004; Luo et al. 2008; Rosenzweig et al. 2008; Scheiter and Higgins 2009; Doherty et al. 2010; Kgope et al. 2010), three prominent expected changes have been chosen, namely changes in [CO2], temperature and rainfall.

This study focuses on the tropical woodlands, savannas and grasslands of southern Africa. In a region with relatively high levels of deforestation and degradation (Boko 2007), there is good opportunity to implement large-scale land-use based climate change mitigation and adaptation ventures. REDD and CS activities are assessed in their broadest sense, including the restoration and avoided degradation of grasslands and vegetation functional types (VFTs) that fall outside of the current CDM definition of a forest. In terms of REDD activities, the aim of this study is to assess if climate change will have an effect on above-ground carbon stocks, and for CS activities, the focus is on the effect climate change may have on the rate of carbon sequestration the restoration of a VFT, be it in grasslands, open savannas or closed woodlands.

The key questions this study seeks to answer are: (i) How do projected changes in [CO2], temperature and rainfall affect existing forest carbon stocks (REDD activities) or the rate of forest rehabilitation (carbon sequestration ventures)?; (ii) Does the direction or magnitude of the change differ between REDD and CS activities?; (iii) How do the effects of projected changes in [CO2], temperature and rainfall on carbon stocks compare when modeled separately?; and (iv) Do projected changes in climate present a significant risk to land-use sector climate change mitigation activities?

Materials and Methods

Simulated activities, models and scenarios

The effect of changes in [CO2], rainfall and temperature on the outcome of both REDD and CS activities were modeled. Each variable was modeled individually as well as collectively to assess the interaction between changes in [CO2], rainfall and temperature. To obtain an estimate of the range of the effect on REDD and CS activities in southern African, the effect was modeled on seven vegetation functional types using two Global Change Models (GCM) for two separate scenarios. An explanation of the choice of sites, model routines and GCMs follows.

A functional group approach

Due to the differing plant physiological characteristics and variation in site-specific factors such as temperature, precipitation and nutrient availability, it is unrealistic to assume that carbon stocks and sequestration will be influenced by climate change in the same manner across all vegetation types and geographic locations. We therefore took a functional group approach, delineating the vegetation of southern Africa into plant functional types based on the ecological function of carbon sequestration. Broader vegetation functional types (VFTs) were then identified which are defined by the spectrum of plant functional types they contain.

Ellery et al. (1991) showed that the main determinants of VFTs in southern Africa with regard to structural composition are plant available moisture and nutrient availability. The nature of the relationship between each of these determinants and net ecosystem productivity is often complex, non-linear and reliant on other prevailing determinants or conditions. Placing each of the determinants in a hierarchical order of importance and establishing a dendrogram to separate VFTs is thus not appropriate. A pragmatic, but well reasoned approach to delineating each of the VFTs was adopted based on the findings of Ellery et al. (1991) and Scholes et al. (1997b).

Using plant available moisture and nutrient availability as the axes of variation, the vegetation functional types of southern Africa were separated a priori into four broad classes (Figure. 1). Differentiation within classes, for example, sub-tropical thicket, fine-leafed savanna and dry grasslands within the Dry – Fertile class, is due to additional biophysical factors such as fire frequency or rainfall seasonality that significantly affects long term carbon accumulation potential. Not all vegetation types in southern Africa were considered due to their very low carbon stocks and sequestration rates. For example, deserts, arid karoo shrubland and fynbos were omitted.

For each vegetation functional type, well-described and much-studied field sites were used as a basis for modeling simulations (Appendix 1). For example, the Mongu and Skukuza sites have been intensively studied during the course of the SAFARI and CarboAfrica projects (Otter et al. 2002; Kutsch et al. 2008; Archibald et al. 2009; www.carboafrica.net), and the grassland sites near Bloemfontein and Cathedral Peak are the subject of long-term field studies by the Universities of the Orange Free State and KwaZulu-Natal (Snyman and Fouche 1991; Everson et al. 1998).

The modeling routine

Each of the sites in Appendix 1 was modeled using the Century Ecosystem Program. Century is freely available online from Colorado State University (www.nrel.colostate.edu/projects/century) and has been used successfully in numerous past studies on carbon dynamics (for example Parton et al. 1993; Hall et al. 1995; Ojima et al. 1996; Peng and Apps 1999; Antle et al. 2003; Song and Woodcock 2003; Mooney et al. 2004; Capalbo et al. 2004; Luo et al. 2008).

Figure. 2 is an example of a typical Century Ecosystem Program simulation run for a CS activity. The Century Model is initially run for ~ 2000 simulated years using an average climate allowing the program to reach an equilibrium state (section 'a', Figure. 2). A disturbance event ('b', Figure. 2) is then introduced that degrades the carbon in the system (point 'c', Figure. 2). The disturbance event could be a biophysical disturbance (for example fire), or a management intervention (for example goat farming) that leads to a loss of carbon. The pre-2000 simulation routine is then reintroduced from point 'c' onwards and the system is allowed to regenerate ('d', Figure. 2).

The degradation – restoration scenario for each vegetation type was based on observations relevant to that vegetation type described in published studies (Table 1, Figure.2). For each vegetation type, attempts were made to simulate a 'typical real-world' degradation event followed by a potential carbon sequestration, rehabilitation initiative. For the mopane, broad-leaf and fine-leafed savanna sites that are situated in the Kruger National Park, the degradation scenarios were based on the studies of Shackleton et al. (1994), Shackleton (1997) and Scholes (1987) on degraded land in rural communal areas adjacent to the Park. In such communal areas, degradation is due to overgrazing and unsustainable fuelwood collection that reduces the standing biomass to ~ 10-15% of its intact state. The Miombo woodland scenario was based on Chidumayo's (1997) work on the re-establishment of woodlands following clear cutting (slash-and-burn) practices commonly found in the Miombo woodlands of Zambia. The scenario for sub-tropical thicket was based on studies by Lechmere-Oertel et al. (2005) and Mills et al. (2005), that describe the rehabilitation of degraded thicket following unsustainable goat farming in the 1960's and '70's. The grassland scenarios were based on the observations of Snyman and Fouche (1991) in degraded grasslands.

Each of the degradation scenarios cited above were simulated in Century by introducing additional grazing, cropping or fire events during phase ‘b’ in Figure 2. These additional degrading events were then halted at point ‘c’, and the system was allowed to regenerate (phase ‘d’ in Figure. 2.).

For the REDD activity simulation, the model was allowed to reach an equilibrium state. Thereafter changes in [CO2], temperature and rainfall were ramped in over a period of 50 years depending on the particular GCM and scenario simulated.

Choosing scenarios and global climate change models

The UNFCCC requested the Intergovernmental Panel on Climate Change (IPCC) to develop potential climate change scenarios that could be used to analyze climate change and its possible impacts (Nakicenovic et al. 2000). The IPCC developed four climate change scenarios that covered “a wide range of the main demographic, economic, and technological driving forces of greenhouse gasses and are representative of the literature” (Nakicenovic et al. 2000). A number of institutions, for example, the Centre of Climate System Research (CCSR) and the Australian CSIRO, developed global climate change models (GCMs) for each scenario.

Our analysis spanned two scenarios (A2 and B1), using the projected climate from two models (CCSR and CSIRO). The B1 scenario describes a world where the focus is on a reduction in material intensity and the introduction of clean efficient technologies. The emphasis is on long-term sustainability and improved equity, and the global population peaks mid-century and declines thereafter. Forecast emissions are 983 GtC for the period 1990-2100 (IPCC 2000).

The A2 scenario portrays a world characterized by self-reliance, the preservation of local identities and a continually increasing global population (IPCC 2000). The forecast emissions are 1862 GtC for the period 1990-2100.

For a full description of the scenarios, see Nakicenovic et al. (2000). The CCSR and CSIRO GCMs were chosen based on data being readily available from the IPCC Data Distribution Centre for both the A2 and B1 scenarios (DCC, http://ipcc-ddc.cru.uea.ac.uk). These data were downloaded in gridded binary (.grb) format and extracted using GrADS (Grid Analysis and Display System – version 1.5.1.12), which is freely available through the IPCC-DDC website.

For this study, we used climate change data and software that are freely and readily available. Although updated GCMs have been developed since the CCSR and CSIRO GCMs used in this study, data are not readily available for the required number of scenarios and models. Moreover, the intention of this study is to obtain an estimation of the range of possible changes in carbon stocks and sequestration rates. For particular localities and future work, it is recommended that updated model runs be accessed.

Adjusting climate data for each site to match the GCM

A climate change anomaly was calculated for each scenario and model based on projected monthly minimum and maximum temperatures and precipitation for each study site. Due to high inter-annual variation in weather data, the baseline 2000 values were calculated as the average between 1990 - 2000 and 2050 as the average between 2045 - 2055. As the GCMs baseline projection for the baseline in the year 2000 differs from true observed data, the projected change in temperature and rainfall is applied to the historical weather data.

For minimum and maximum temperatures, the change is the absolute difference in temperature between the two projections. In the case of precipitation, the change is calculated as a multiplier so as to avoid negative rainfall data being calculated. The change in climatic variables between the baseline and 2050, whether in the form of an absolute value or a multiplier, is then applied to the observed baseline data using a sliding linear scale.

The observed baseline (2000) data are calculated by averaging the minimum and maximum monthly temperatures, and monthly precipitation of observed data for at least the past 30 years. The weather data for the South African sites was obtained from the South African Weather Service (www.saweather.co.za) and for the Miombo woodland site located at Mongu in western Zambia, data was acquired from the Global Historical Climate Network (http://www.ncdc.noaa.gov).

Changes to Century parameter files

The changes to the parameters in Century were made as per recommended for "Enriched CO2 Effects" in the Century 5 User Guide and Reference and previous published studies on the effect of changes in [CO2], temperature and rainfall on plant productivity (Ojima et al. 1996; Scholes et al. 1997a). An additional weather file for each site is created that includes the GCM projected climate changes as described above for 2000 – 2050. Transpiration and production rate, carbon/nitrogen ratio and root/shoot ratio variables are adjusted to simulate the presence of additional [CO2]. A 20 percent decrease in potential evapo-transpiration (PET), a 20 present increase in production rates, and 20 percent decrease in the nitrogen content plant materials were allowed for (Ojima et al. 1996). A change in [CO2] from 350ppm to 700ppm was 'ramped in' using a linear function between 2000 and 2100.

Results

The metric used to present the results is the percentage change in carbon stocks between ambient conditions and the treatment over the first *20 years* of the REDD or CS activity. A positive percentage indicates that the modeled change in carbon stocks is greater for the particular treatment compared to ambient conditions. It should be noted that the 'change' referred to is the net change in carbon stocks or Net Ecosystem Productivity (NEP), not the gross or net primary production.

The modeled effect of changes in rainfall, temperature and [CO2] on carbon stocks and the amount of carbon sequestered through ecosystem rehabilitation are presented in Figure 3 and 4 respectively.

The combined effect of a change in temperature, rainfall and [CO2] leads to a positive increment in carbon stocks for all VFTs and REDD and CS activities. This holds for all GCM models and scenarios simulated. The size of the positive change varies substantially depending on the VFT modeled (REDD: 3-48%, CS: 7-32%). However, the percentage change in standing carbon stocks versus carbon sequestered (REDD versus CS activities) is similar for a particular VFT (Figure 3, 4).

When modeled separately, the effect of changes in rainfall and temperature are generally greater than the effect of increased [CO2]. Exceptions however do occur, particularly for the sub-tropical thicket and dry grassland sites. Considering the effect of changes in temperature and rainfall alone, there is little clear trend between the CCSR and CSIRO GCMs and the A2 and B1 scenarios (Figure. 3, 4). This may be due to short period of interest (a REDD or CS activity over 20-30 years).

Comparing the *GCM* used, the effect of the CCSR and CSIRO GCMs are similar for the Miombo, Mopane, broad- and fine-leafed savanna VFTs located in western Zambia and the South African lowveld east of the South African escarpment. Larger differences between the two GCMs are found for grassland sites at Cathedral Peak and particularly for the dry grassland site west of the escarpment near Bloemfontein - not only is the difference in the size of the change larger compared to other sites, but notably, the direction of the change as well. When the effect of changes in temperature and precipitation on a CS venture at Bloemfontein Dry Grassland site are modeled alone, using the CSIRO GCM leads to an increase in carbon sequestered compared to ambient conditions, whereas the CCSR GCM leads to a decrease in expected carbon accumulation (Figure. 3).

With regard to the effect of the *scenario* modeled, the difference in carbon stock changes between the A2 and B1 scenario is generally negligible (< 5 percent, Figure. 3, 4). An exception is the CCSR A2 and B1 scenarios for the moist grassland site at Cathedral Peak where the inter-scenario difference can be up to 20 percent when the effect of changes in temperature and rainfall are modeled individually.

Compared to the effect of projected changes in rainfall and temperature, the effect of increasing [CO2] is less clear with both negative and positive directional changes in standing stocks and sequestered carbon compared to ambient conditions (Figure. 3, 4). The effect on standing carbon stocks in a REDD project scenario ranged from -5.22 percent for fine-leafed savanna to +1.88 percent for dry grasslands. The potential effect on carbon sequestration is greater, ranging from -10.40 percent for fine-leafed savanna to +6.15 percent, for broad-leafed savanna in the first 20 years of the activity.

Discussion

Our analysis presents a means of quantifying an existing source of uncertainty, thereby transforming the effect of changes in rainfall, temperature and [CO2] into a manageable risk variable. This forms an important part of assessing the overall risk to private or public sector investment in CS and REDD activities.

The combined effect of changes in [CO2], temperature and rainfall

Our results show that, in general, the combined effect of changes in temperature, rainfall and [CO2] leads to positive increment in standing carbon stocks and carbon sequestration for the sites modeled in southern Africa. Previous modeling as well as empirical studies that assessed the joint effect of [CO2], temperature and rainfall have found similar results in terms of the direction- and magnitude of the change (Mellilo et al. 1993; Hall et al. 1995; Ojima et al. 1996; Luo et al. 2008; Doherty et al. 2010; Kgope et al. 2010). Moreover, our results for the Mopane, fine- and broad-leafed savanna VFTs are of a similar direction and magnitude to those reported by Scheiter and Higgins (2009) who modeled the joint effect of changes in [CO2], temperature and rainfall according to the IPCC 2007 A1B SRES scenarios for similar sites at Pretoriuskop and Satara in the Kruger National Park.

While the magnitude of the positive change is similar for both REDD and CS ventures for a particular VFT, there is considerable variation between VFTs (REDD: 3-48%, CS: 7-32%). Such variation in VFT or site specific responses may be due to the degree of water limitation or the availability of soil nutrients such as nitrogen or phosphorus.

Several past studies, for example Hall et al. (1995), Ojima et al. (1996), Bond et al. (2003a), Gerten et al. (2008) and Luo et al. (2008), note that drier ecosystems were more responsive to changes in [CO2], temperature and rainfall compared to wetter sites. This study shows a similar pattern where drier grassland or savanna sites show a larger relative change in carbon stocks compared to wetter sites. An exception is the Subtropical thicket VFT that has a small relative change in carbon stocks even though it is one of the drier sites. However over 80 percent of the biomass in subtropical thicket is constituted by a single species, *Portulacaria afra* (Mills and Cowling 2004). *Portulacaria afra* is a succulent tree and therefore may not be constrained by soil water availability in the same manner as the woody species modeled for the Miombo, Mopane, broad- and fine-leafed savanna VFTs. Although the Century Model does not include a CAM photosynthesis plant module, the C3 tree module was calibrated where possible to simulate carbon accumulation rates observed for subtropical thicket in the field. This example highlights the advantage of estimating the level of 'dryness' in terms of "the degree of ecosystem water limitation" and not only in terms of water supply (Gerten et al. 2008). The "degree of ecosystem water limitation" is the ratio between soil water supply and atmospheric transpirational demand, and therefore appropriately includes a measure of required water demand as well.

In a similar manner to the inter-VFT variability in carbon sequestration due to water limitation, the availability of soil nitrogen (N) and other soil nutrients such as phosphorus may significantly affect the magnitude of the response in NEP to changes in [CO2], rainfall and temperature (Scholes and Hall 1996; Scholes et al. 1997a; Johnson 2006; Luo et al. 2006). A good example of the effect of available soil N is a comparison of the broad- and fine-leafed savannas in this study. Both sites are found in close proximity near Skukuza in the Kruger National Park with similar rainfall and temperature regimes. Yet, the fine-leafed savanna site generally shows a larger increase in carbon stocks and carbon accumulation compared to the broad-leafed savanna, especially to changes in rainfall and temperature.

The main difference between the two VFTs is the type of soil and the availability of soil nutrients. Scholes and Walker (1993) provide a detailed account of the nature of fine-leaf nutrient-rich savanna versus broad-leaf nutrient-poor savanna. Briefly, the "nutrient-rich" savanna which is typified by fine-leafed *Acacia* species tends to exist on more clayey soils with a higher N content (4635 kgN ha^{-1}), compared to "nutrient-poor" savanna that exists on sandy soils with a lower N content which is characterized by the presence of broad-leaf *Combretum* species (3310 kgN ha^{-1} Scholes et al. 2003). In a savanna system where one of the primary production constraints is soil N (Scholes and Hall, 1996), a nutrient-rich fine-leafed savanna is expected to show a larger response in primary production to a change in rainfall and temperature, compared to nutrient-poor broad-leafed savanna.

It is important to note that while soil N may play a role in determining the initial magnitude of the response for a particular site, over the longer term, N availability is expected to progressively limit continued increases in carbon stocks due to changes in [CO2], temperature and rainfall. In their comprehensive reviews on the nature of progressive N limitation, Johnson (2006) and Luo et al. (2006) note that while additional [CO2] may lead to N accumulation in soil pools and therefore may limit the complete downward regulation of increased primary production, progressive N limitation is still expected to constrain potential increased carbon stock growths over the long-term.

The effect of additional atmospheric carbon dioxide alone

The slight increase in carbon stocks in Miombo woodland, subtropical thicket and grassland sites under rising [CO2] are in line with previous empirical and simulation studies of similar systems (Ojima et al. 1996, Bond et al. 2003a, Scheiter and Higgins 2009; Doherty et al. 2010) and a general understanding of the anticipated effect of an increase in [CO2] on plant growth in terms of increased photosynthetic rates and improved water-use efficiency (Scholes et al. 1997b; Luo et al. 2006). This result may, however, be due to the shorter temporal scale of the modeling exercise, the absolute size of the change compared to other prevailing ecological processes such as fire and herbivory that are included within the Century model simulation.

Temporal scale - Considering the effect of changes in [CO2] over the next 20 years

In the context of this study, the aim is to estimate the effect on REDD and CS ventures in the next 20 years. Although the model linearly 'ramps' in a change in [CO2] from 350ppm to 700ppm over a 100 years in a similar manner to the studies listed above, the focus is on the first 20 years where [CO2] concentrations increase from 350ppm to 420ppm in the model simulation (a linear increase of 350ppm over 100 years would result in an increase of 70ppm in the first 20 years). Over such a relatively small time-period and increase in [CO2], the effect on carbon stocks may be obscured by inherent variation, particularly the effect of fire on savanna systems.

Previous meta-analyses of studies focusing on the effect of additional [CO2] on plant growth reported average increases in biomass or carbon stocks of 23 percent (Luo et al. 2006) and 30 percent (Kimball et al. 1993) following a doubling of [CO2]. More particular to the rangelands of southern Africa, Ojima et al. (1996) found that a shift in [CO2] concentration from 350ppm to 700ppm lead to a modeled 10 – 15 percent increase in primary production in both dry and humid savanna systems. The generally less than 5 percent change observed in the results of this study, is therefore well within the range of change reasonably expected within the first 20 years of a gradual 3.5ppm.yr^{-1} increase in [CO2].

The effect of the particular range of [CO2] simulated (350 to 420 ppm)

In their empirical experiment of the effect of [CO2] on African *Acacia* savanna trees, Kgope et al. (2010) observed that while stem length, shoot- and root- weight increase significantly between sub-ambient and ambient conditions (~180-370 ppm), the effect of additional increases in [CO2] on tree growth is less apparent within 370 to 1000 ppm range. Future predicted rises in [CO2] tended to lead to marginal changes in tree growth dependent on the species and season, and in certain cases may lead to a slight decrease in tree stem length, shoot- and root-weight as [CO2] increased from 370 to 550 ppm.

The effect of elevated [CO2] on REDD and CS activities, or more specifically, on carbon stocks and carbon accumulation rates over the next 20 years, may therefore be negligible. The Century Ecosystem Model is a complex model that includes fire and herbivory processes that may have greater effect on carbon stocks than increased [CO2]. Bond et al. (2003a) made a similar observation when assessing the effect of elevated [CO2] on comparable South African savanna sites. While tree densities of semi-arid and mesic sites in the study were expected to increase slightly with a doubling of [CO2], tree densities in arid sites decreased due to higher grass growth leading to more intense fires and higher tree mortality. Although the sites modeled in this study are more semi-arid than arid (600 vs. 300 mm annual rainfall), the observation highlights the potential overriding effect that other ecosystem drivers such as fire, herbivory and competition may have on observed NEP.

The effect of changes in temperature and rainfall alone

For most VFTs, the effect of temperature and precipitation follow similar trends in terms of the direction of the change in carbon stocks and the magnitude of the change for a particular VFT. The effect of the degree of ecosystem water limitation and available soil nitrogen discussed above is again observed in the results of the influence of changes in rainfall and temperature alone. Dryer sites and those with higher soil N tend to show a larger positive response to both changes in rainfall and temperature.

There is no clear difference between the CSIRO and CCSR GCMs with respect to rainfall and temperature in our study sites over 20-30 year period of concern. The effect on changes in temperature and rainfall according to both GCMs appears to be similar for the majority of sites except for the two grassland locations. While a comprehensive review of agreement in GCM projections and general climate change projections for southern Africa is beyond the scope of this discussion, there is consensus among GCMs that the winter-rainfall western portion of the sub-continent will tend be dryer going forward while the summer-rainfall eastern component South Africa will tend be wetter (Midgley et al. 2007). Where projections are less clear and there is less agreement between GCMs is in the transition area between the winter- and summer-rainfall zones.

The Miombo Woodland in western Zambia and the Mopane, fine- and broad-leafed savanna sites located in the Kruger National Park in the South African Lowveld fall well within the summer rainfall zone where there is agreement between GCM models. However, the Wet Grassland site at Cathedral Peak on top of the Drakensburg Escarpment and the Dry Grassland site just west of the escarpment near Bloemfontein, fall broadly within transition zone between winter and summer rainfall. This transition zone is inherently climatically complex and covers a broad range and climatic zones and dynamics from wetter coastal belt over the high Drakensburg Mountain Escarpment to the drier higher altitude grasslands of the Free State Province.

In a similar manner to the comparison of GCM results, there is little difference in the outcomes of the A2 versus the B1 scenario for a particular GCM. The modeled change in carbon stocks under the B1 scenario tends to be slightly less than the A2 scenario, which is to be expected. Again in a similar manner to the GCM results, there is a larger difference for the grasslands sites, particularly CCSR GCM.

In Conclusion

For the suite of VFTs and sites modeled, projected changes in [CO2], temperature and rainfall appear to present little downside risk to the expected carbon outcomes of REDD and CS initiatives over the next 20 year period. If anything, projected changes in [CO2], temperature and rainfall could lead to up to a 30 – 40% increase in carbon sequestered over the lifetime of a project for the suite of sites modeled.

Of crucial importance to the land-use sector in general is the ability to assess potential downside risk in land-use change ventures. Estimating the risk that climate change presents to CS and REDD ventures as illustrated in this study, provides a valuable input into assessing the risk profile of land-use ventures and reducing sources of uncertainty in policy and funding decisions. It is however, only part of assessing the overall investment risk in CS and REDD activities. The potential risk that other biophysical variables present, such as fire and pests, need to be considered including how such variables may alter with a changing climate. In addition, biophysical risk analyses need to be integrated with economic models to assess the impact of biophysical risk on expenditure, revenues and the overall financial feasibility of the initiative (Aaheim et al. 2010).

Acknowledgements

The National Research Foundation, the Council for Scientific and Industrial Research and Stellenbosch University are thanked for contributing to this study. Dr Bill Parton is thanked for providing TK with Century Model tuition and Weather SA is thanked for providing climate data.

Figures and Tables

	Infertile	Fertile
Wet	Miombo woodlands	Moist grasslands
Dry	Mopane woodlands Broad-leaf savanna	Subtropical thicket Fine-leaf savanna Dry grasslands

Figure. 1 A diagram showing the separation of vegetation types in southern Africa based on plant available moisture and nutrient availability. Further subdivisions were driven by additional prevailing biophysical variables (for example, fire frequency).

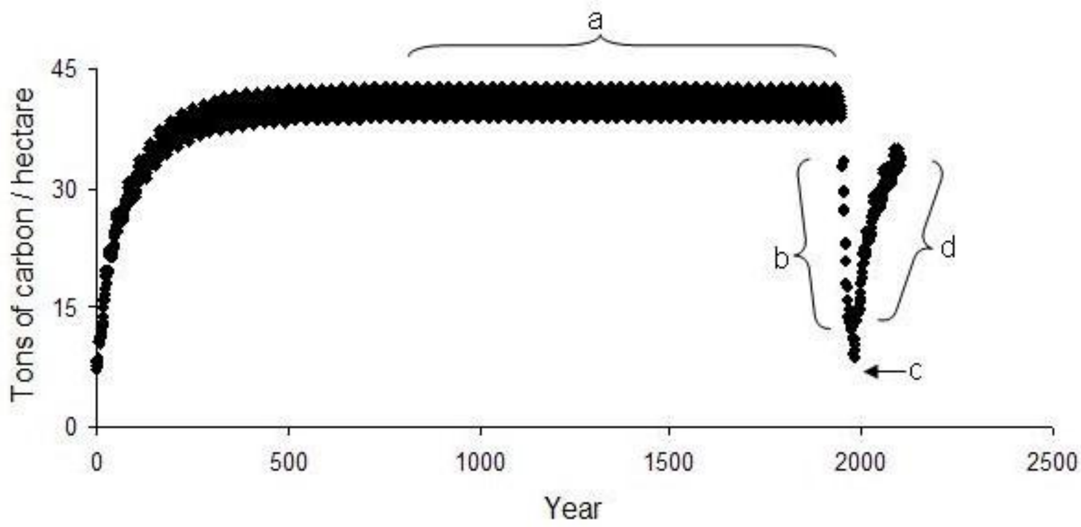

Figure. 2 An example of the results of a typical Century Ecosystem Program simulation run for sub-tropical thicket. a is the ‘equilibrium’ intact state, b is the period in which the carbon stocks are reduced to a ‘degraded state’ through substantial increases in herbivory or the harvesting of wood. From point c, the additional harvesting of wood is removed and the system is allowed to recover (section d). See Table 1, for values for a, c and d for each vegetation functional type.

Table 1 The equilibrium, degraded state and growth rate data obtained from published studies for each vegetation function type. State 'a', 'c', and phase 'd' refer to the corresponding labels in Figure. 2. The carbon stocks quoted are for above-ground carbon in tons of carbon per hectare.

Vegetation functional type	Study site	Equilibrium state 'a' - tons AG C / hectare	Degraded state 'c' - tons AG C / hectare	Restoration phase 'd' - % increase / year	Reference
Mopane	Letaba exclosure	11.5	3.3	2 - 3	Scholes 1987; Shackleton 1997
Broad-leaf savanna	Skukuza flux tower	9.5	1.9	2.0 - 2.7	Scholes 1987; Shackleton *et al.*, 1994; Shackleton 1997
Fine-leaf savanna	Skukuza flux tower	16	2.5	2 – 6.4	
Miombo woodlands	Mongu, Zambia	27.5	1	1 – 2	Chidumayo 1997
Subtropical thicket	Baviaanskloof, Eastern Cape	41	8	1 - 2	Mills *et al.* 2005; Lechmere-Oertel *et al.* 2005
Dry grasslands	Bloemfontein	0.6	~0.1	~50	Snyman and Fouche 1991
Moist grasslands	Cathedral Peak	1.1	~0.1	~50	Everson 1985

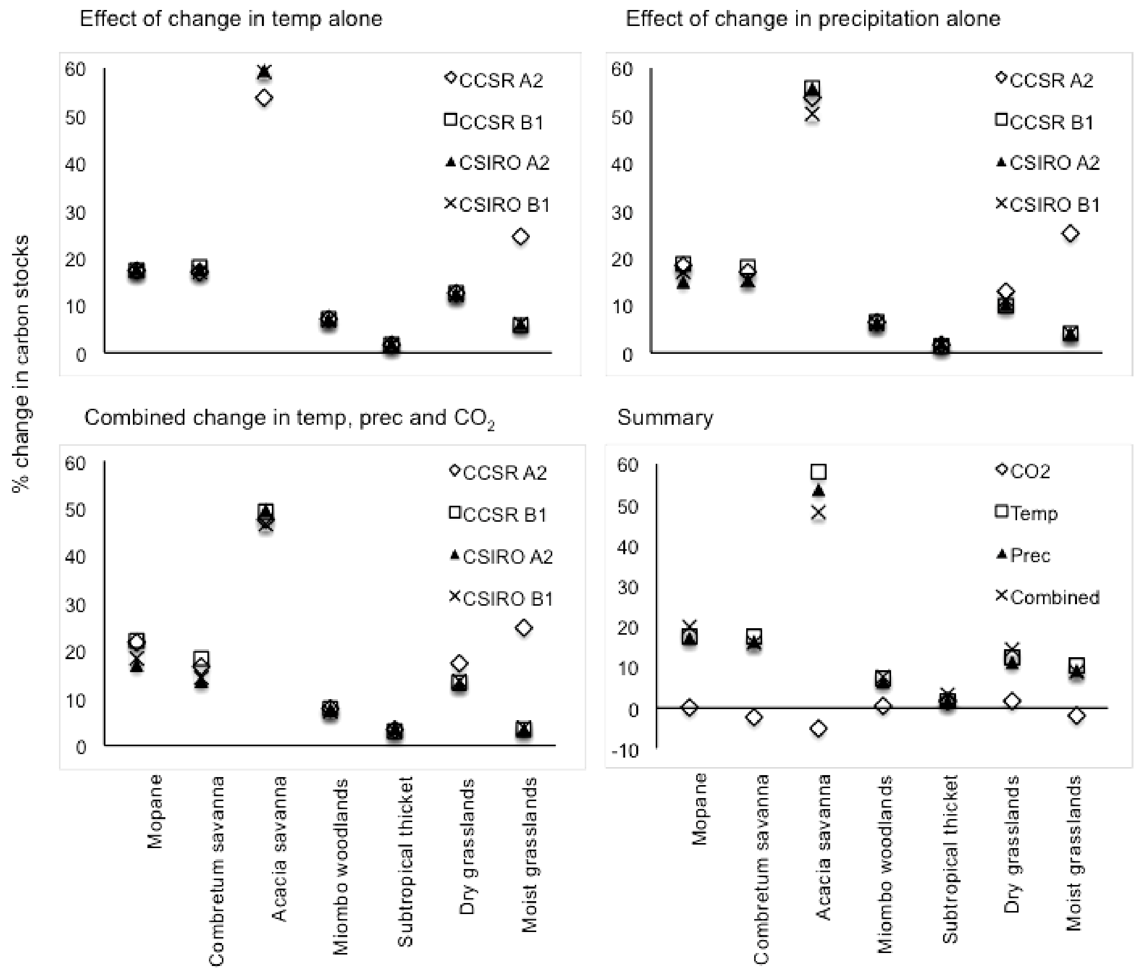

Figure 3. REDD activity scenario: The percentage change relative to ambient (control) conditions in aboveground carbon stocks over 20 years due to projected changes in [CO2], temperature and rainfall.

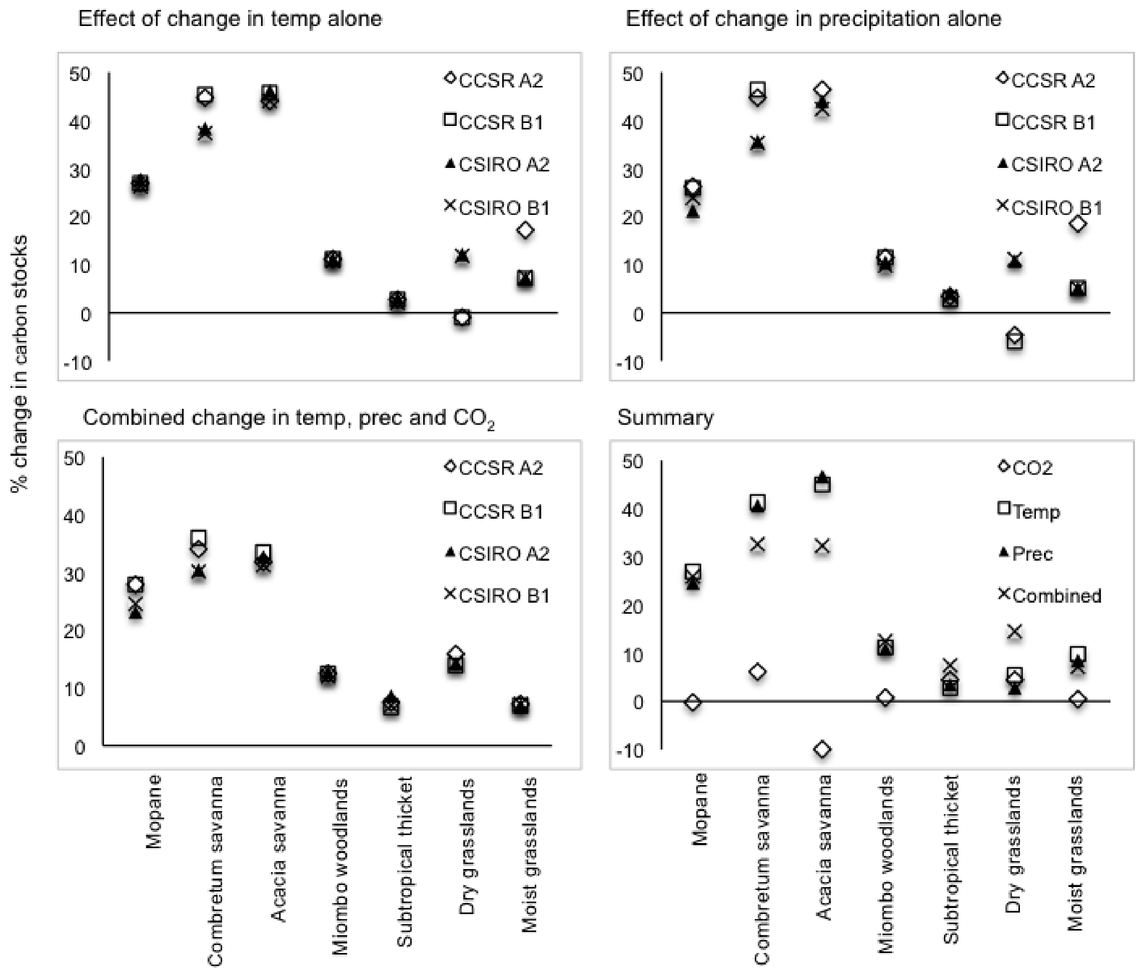

Figure 4. Carbon sequestration / restoration activity scenario: The percentage change relative to ambient (control) conditions in aboveground carbon stocks over 20 years due to projected changes in [CO2], temperature and rainfall.

Appendix 1: A brief attribute description of the study sites employed in this modeling study

Vegetation type	Study site	Rainfall	Temperature		Soil		Dominant plant species	Biomass (above ground)
		mm/year	Mean °C	Range °C	Type	Sand/silt/clay %		tons / hectare
Mopane	Letaba	506	23	7.5 - 32.3	Coarse loamy	80.5/10/9.5 Paterson and Steenkamp 2003	Colophospermum mopane	21.6 Scholes 1987
Broad-leaf savanna	Skukuza	572	22	2.4 - 35.9	Sandy	69/5/26 Scholes *et al.* 2001	Combretum apiculatum	18.9 Shackleton 1997
Fine-leaved savanna	Skukuza	572	22	2.4 - 35.9	Clayey	65/3/32 Scholes et al, 2001	Acacia nigrescens	28 (Inferred from) Scholes and Walker 1993
Miombo woodlands	Zambia - Mongu	956	23	6.5 - 39.2	Sandy	95/2/3 Pinheiro et al. 2001	Brachstegia spiciformis	55 Desanker et al. 1997
Subtropical thicket	Eastern Cape	413	19	5.1 - 34.5	Loamy	83/9/8 Lechmere-Oertel 2003	Portulacaria afra	80 Lechmere-Oertel 2003
Dry grasslands	Bloemfontein	560	16	- 4.9 - 34.1	Loamy	87/3/10 Snyman and Fouche 1991	Themeda - Cymbopogon	1.2 Snyman and Fouche 1991
Moist grasslands	Cathedral Peak	1324	14	2.1 - 24.9	Acidic, leached	23/24/53 Everson et al. 1998	Themeda triandra	2.3 Everson 1985

Chapter 7

Summary and Synthesis

Perhaps one of the more interesting but also challenging aspects of land use based climate change mitigation activities are their multi-disciplinary nature. Developing land use based activities in a particular area requires consideration of local socio-economics, social structures and governance, local and national politics, systems ecology, investment finance, national and international climate change policy and associated public funding and carbon market related revenues.

The objective of this thesis was to examine certain aspects that are particular to land use activities, as opposed to industrial or energy sector projects. While a comprehensive consideration of all additional aspects is beyond the scope of a single thesis, effort has been made to focus on five particular risk and feasibility aspects that are important to the realisation of REDD activities in southern Africa. It is hoped that the new insights and knowledge emerging from this thesis may inform practitioners ranging from international policy makers to local project implementers, as well as contribute to a growing understanding of how to appropriately consider, design and implement land use based climate change mitigation activities and ecosystem service investments more broadly.

A common theme through the individual papers of this dissertation is consideration of the risk and feasibility associated with REDD activities in general. A sound scientific rationale and clear demand exists for halting the degradation of intact indigenous ecosystems as well as restoring systems where possible. It provides benefits from a climate change mitigation and adaptation point of view as well as from the perspective of improving human livelihoods, the provision of ecosystem services and the long-term conservation of biodiversity.

In terms of new emerging knowledge, this dissertation contributes more broadly to assessing the viability of land use based climate change mitigation activities located in sub-Saharan Africa. In particular it contributes to our knowledge of the biophysical risk, financial feasibility and appropriate policy and implementation structures, associated with REDD activities.

Considering the potential influence of biophysical risk on expected outcomes

Chapter 5 and 6 considered biophysical risk factors that may affect the expected outcome of REDD activities and the volume of emission reduction units (ERUs) and associated income realised at the end of the initiative's lifetime (Chapter 1, Fig. 1). Such risk factors have the potential to significantly reduce the volume of ERUs awarded to the host-country or project-scale implementer, and therefore the attractiveness of land use based climate change mitigation activities to policy makers, stakeholders, land-owners and investors alike.

The review indicates that fire does not present a considerable risk to REDD initiatives in woodland, savanna and grassland systems located in sub-Saharan Africa. In such vegetation types, fires are typically in the form of a ground burn, as opposed to a crown fire, that has little effect on mature trees and the total carbon stock of the system. Consequently, a fire event leads to the release of small fraction of the total system carbon pool that generally recovers within in a few years. Furthermore, the implementing agent may be able to further reduce the risk of fire through management interventions such as early season burns that remove the grass fuel load and appropriate timber harvesting practices.

Where fire may present a significant risk to the outcome of REDD activities is in moist forest vegetation types that were seldom exposed to fire in the past, but due to climate change are now more fire-prone; these systems are typically too wet to burn due to the continual presence of moist conditions throughout the year. Changes in climate however may lead to the occurrence of unprecedented dry periods that provide a period in which fire may occur. Under these circumstances, fire may lead to the release of a significant fraction of the total carbon stock of the system. In addition, it may take several decades (at least) for carbon stocks to recover during which fire may be a particular risk due to the absence of a closed canopy and a moist micro-climate.

Whereas most GCMs project that the moist forests of central and west Africa will get wetter in the future and therefore the risk of fire may be low in such systems, elsewhere, particularly in certain areas of South America and South-East Asia, fire may present a significant and real threat to long-term existence of moist forests and associated carbon stocks (Scholze et al. 2006, Pechony and Shindell, 2010). It is therefore crucial that the risk of fire to carbon stocks under such circumstances is considered when developing national- and project scale activities.

The second paper to consider biophysical risk (Chapter 6) sought to assess the effect of projected changes in climate and atmospheric CO_2 on the outcome of reforestation and avoided deforestation activities. For the majority of vegetation types and sites modeled, projected climate change is likely to a lead to an increase in carbon stocks over a 20 year period compared to ambient conditions. Future predicted increases in atmospheric CO_2 are also likely to lead to a slight increase in carbon stocks for the majority of the sites modeled. The modeled decrease in carbon stocks for the nutrient-rich savanna vegetation type was not expected and requires further investigation. In a similar manner to the need to assess the fire risk, it would be prudent for the developers of national- and project-scale projects to assess the potential impact of predicted change in climate prior to the implementation of activities.

Together, the last two papers provide an initial, but important consideration of the potential impact of biophysical risk factors on the outcome of land use based climate change mitigation activities that are applicable to investment in ecosystem-based climate change adaptation activities as well as ecosystem services. Assessing the risk of fire for a specific location requires site-specific consideration of for example, vegetation type, successional stage, antecedent and future climate, current land-management, potential timber harvesting, the level of fragmentation of the landscape, the spatial extent of the initiative as well as the nature of the proposed REDD activity.

Furthermore, it may be interesting to focus on the secondary effects of climate change, through the potential effect of changes in climate on ecosystem drivers such as human-activities and fire. For example, whereas projected increases in moisture and temperature for the equatorial regions of east and central Africa may lead to increases in net primary productivity (NPP, Doherty et al. 2010), projected increases in rainfall variability over the same period could be hypothesized to increase the size of the grass fuel load in open woodland, savanna and grassland systems leading to an associated change in the nature of the 'fire trap'. This may in turn inhibit the recruitment of saplings and the survival of older tree specimens.

The influence of transaction costs on project feasibility

Whereas Chapters 5 and 6 focused on risk factors that may affect expected emission reductions and associated revenues, Chapter 4 focused on the influence of transaction costs on the financial feasibility of REDD activities (Chapter 1, Fig. 1). It was noted that transaction costs may be more than previously assumed. Especially for smaller-scale projects (<10,000ha), transaction costs in the order of US$500,000 to US$700,000 may present a significant roadblock to the realisation of REDD. If project development, implementation or incentive costs are added to this analysis, the cost of REDD may be considerably more than first presumed, especially for activities located outside moist forests.

Experience in developing REDD activities in sub-Saharan Africa over the past five years indicates that these high costs, in relation to relatively low returns, are some of the principal reasons why land use based climate change mitigation activities have not been widely implemented to date. While it is likely that a certain portion of the transaction costs incurred through the realisation of project-scale initiatives will be absorbed by proposed national-scale REDD implementation capacity and some efficiencies of scale may be realised, transaction costs still need to be considered when attempting to calculate the true cost of REDD and its efficiency and appropriateness compared to alternative climate change mitigation options.

In order to establish the true cost of REDD as well as mechanisms and policies that intend to make REDD more efficient and attractive to investors. Future research needs to include consideration of project identification, development and operational costs. In addition, the cost of developing, implementing and incentivising alternative livelihoods may be particularly significant in the populated landscapes of southern and eastern Africa. Furthermore, additional research is required to understand the relative costs and benefits of REDD implementation in different contexts. Chapter 4 focused on REDD activities located in the woodlands and savannas of sub-Saharan Africa. An avoided deforestation activity that, for instance, involves the halt of logging concession in a fairly uninhabited part of the Congo-Guinea moist forest belt, may have similar costs but considerably higher revenues. By comparison, a community-based agro-forestry project located in the dry Miombo woodlands of southern Africa may have considerably larger development, implementation and operational costs with relatively low revenues generated through the trade of ERUs.

Considering policy and capacity requirements for appropriate REDD implementation

In line with assessing the feasibility of- and risk associated with land use based climate change mitigation and adaptation activities, Chapter 2 and 3 focused on policy and implementation issues that are particularly important to the widespread realisation of REDD activities in Africa. The aim of Chapter 2 was to provide a sub-Saharan Africa perspective on key implementation and policy issues that are important to the long-term success of land use based climate change mitigation activities in the region. It not only focused on means of reducing permanence and leakage risk, but, in addition, on ways of improving the potential of REDD in terms of expanding the scope of activities and beneficiation by local residents. It noted that multi-criteria land use planning prior to implementation of REDD may provide a means of reducing permanence risk. This is especially pertinent in the context of sub-Saharan Africa where many countries are emerging from periods of relative isolation and where considerable mining, agricultural and infrastructure development is expected in the medium- to long term. Although the development of a 'final' land use plan for each host-nation may not be appropriate or practical, a clear decision making process on where to place REDD activities is required to address long-term permanence risk.

Furthermore, if REDD is to reach its full potential, there is need to expand the scope of recognised activities that reduce atmospheric GHG concentrations as well as the vegetation types in which activities are recognised. The inclusion of activities in non-forest vegetation types, for example 'reducing GHG emissions through avoided soil carbon degradation', may hold good potential to reduce anthropogenic climate change while providing local livelihood, ecosystem service and biodiversity benefits.

Furthermore, the need to reform land-tenure prior to the implementation of REDD is questioned. Land-tenure reform may not necessarily result in desired improvements in forest management, host-country governments may not necessarily want to reform land-tenure, the process may take a considerable amount of time, and a lack of formal land-tenure does not automatically imply that a land use based enterprise cannot be successful. There are good examples of private sector companies running successful, financially viable operations over the long-term (>30 years) that require implementation at large spatial scales in areas with no land-tenure under constitutional law. A lack of formal land-tenure therefore does not necessarily infer unacceptably high risk. The keys are clear resource tenure rather than land tenure, government support, ensuring that REDD is more lucrative than alternative land use options, and providing true incentives in the form of cash or employment to local residents. The implementing agent needs to create a local sense of value in natural systems otherwise residents are likely to pursue alternative, more attractive land use options.

Chapter 3 expands on this theme of creating appropriate local incentives. The Chapter focused on the appropriateness of local residents undertaking REDD monitoring requirements. The key message is that community-based forest carbon monitoring provides a cost-effective yet robust means of estimating biomass carbon stocks. Experience in attempting to quantify carbon stocks in relatively heterogeneous open woodland and savanna vegetation types indicates that ground based surveys provide a cost-efficient means of reducing the error in the mean carbon stock estimate. Cost efficiencies are increased if locally recruited staff undertake monitoring operations at a fraction of the cost of international or even national consultants.

A relatively simple survey and data transfer methodology has been shown to provide data that are adequate and robust. At a pragmatic level, national-scale implementation will require a large number of monitors and facilitators. The organisation is similar to agricultural outreach officers who monitor, advise and co-ordinate activities in remote rural areas. From a pure logistics point of view, relying on external consultants is not practical. In addition, non-biomass metrics need to be monitored, such as socio-economic indicators, biodiversity, deforestation drivers and agents, and the manner in which deforestation drivers are being reduced. The measurement of such metrics tends to be time-intensive and require knowledge of local languages and formalities. It therefore makes sense for monitoring activities to be undertaken by locally-recruited staff.

Of equal importance to estimating forest carbon stocks is the need to create local incentives, employment opportunities and a local sense of ownership in intact forest systems. A key issue regarding permanence risk is the acceptance and ownership of REDD activities at both a government and local level. Monitoring, the creation of local enterprises (for example bee farming by the indigenous community) as well as ecosystem service management through the public-private partnerships as advocated in Chapter 2, may provide a means of creating crucial direct benefits and value at a local level.

Concluding thoughts and future research needs

In the period of writing this dissertation, land use based climate change mitigation has moved from a fringe consideration in international climate change policy to one of the dominant policy issues and one of the only issues that most parties can agree on. Especially since the 13th Conference of Parties meeting in Indonesia, consideration of land use based climate change mitigation has moved from the limited inclusion of project-scale afforestation and reforestation activities within the CDM to the Kyoto Protocol, to the prominent prioritisation of national-scale implementation across all countries with significant forest cover. At the same time, considerable financial and institutional resources have being allocated to the realisation of REDD. National-scale REDD implementation plans are being developed that include the development of supporting policy, legalities, national institutional capacity, monitoring capacity, and so forth.

In the same period, the science and knowledge required to appropriately respond to climate change has progressed enormously. Certain aspects, such as the development of supporting climate change policy at a national- and international level and how to appropriately approach national-scale implementation has moved forward considerably (see Angelsen et al. 2009, Streck et al. 2009 as an example). Knowledge of terrestrial carbon stocks and fluxes has also progressed significantly. In the past few years, a number of global or continental carbon stock maps have been published and are in continual development (for example, Baccini et al. 2008, Asner et al. 2010).

Considerable science and knowledge gaps still however exist. When it comes to attempting to grapple with the implications and the development of responses to climate change and global change more broadly, many scientists and field practitioners believe we are only starting to understand the enormity of what is required. The results of this dissertation, as with most research, leads to new, exciting but important research questions that need to be considered.

Chapters 4, 5 and 6 of this dissertation are among the first considerations of the effect of climate change and fire on the outcome of REDD activities as well as aspects of the financial feasibility thereof. Stemming from this work, combining these aspects in a joint biophysical-socio-economic model may provide valuable insights into the nature of the relationships between components. Such a model could be used to explore questions such as: How will predicted changes in climate affect fire regimes and associated ecosystem structure and function? Initial analysis on the impact of climate change on rangeland and woodland systems in East Africa (Doherty et al. 2010), predict that increases are likely to lead to an increase in tree cover. The model used however, does not include the impact of fire. An alternative hypothesis may therefore be that an increase in rainfall in rangeland systems may lead to an increase in the grass fuel load at the end of the wet season, which would lead to an increase in fire intensity, which in turn may inhibit tree recruitment and survival. A rebuttal of this argument is that people with livestock inhabit much of the Miombo woodlands of East Africa and early season burning and grazing are likely to limit the grass fuel-load.

To tease apart and better understand these relationships and the nature of climate change impacts, future models should combine these biophysical- and human drivers to better understand the nature of climate change impacts. A combined model may also provide better estimates of the net atmospheric effect of climate change mitigation activities as well as the true costs and benefits in terms of financially returns, livelihood benefits and the net effect of the intervention on ecosystem services.

A second area that requires further investigation is an assessment of the true cost of REDD activities. The majority of cost analyses to date have focused on opportunity costs, with little consideration of project identification, development and administration costs. Moreover, opportunity cost studies to date appear to solely focus on revenues lost from the sale of cash-crops due to not farming an area of land. This assumption may hold for large-scale commercial farmers, but it is questionable when considering subsistence farmers or small-scale cash-crop farmers. Such analysis do not include consideration of traditional ties to the land, the fact that the land is also the farmer’s home and that the farmer’s family derives a suite of ecosystem services from the land and surrounding area in terms of water, food, energy, fiber, housing material and so forth. Whether farmers will move from their land for the market price obtained for crops is doubtful. New, innovative means of estimating the value of land and willingness to change livelihoods may therefore be required.

In addition to opportunity costs and the transaction costs explored in Chapter 4, the cost of project identification, development and management need to be included in order to estimate the real cost of REDD activities. Identification costs include carbon stock as well as deforestation driver, socio-economic, biodiversity and environmental impact assessments. They include the cost of running an office and logistics and the cost of protracted stakeholder engagement and negotiations. Development costs include the expense of developing alternative land use plans, alternative livelihoods strategies, and potentially improved silviculture and fire management plans. To implement truly accepted and sustainable activities, the project implementer needs to understand the socio-economic context and land use drivers of the specific area comprehensively before developing concepts to address deforestation and ecosystem degradation. The residents need to develop a sense of trust and understanding with the project developer before committing to a certain type of land use for at least the next 20-30 years. These processes take time and money, which are not currently being considered in REDD cost estimates. Experience in developing REDD activities in southern and East Africa indicates that where the main drivers of deforestation in an area are the expansion of small-scale subsistence agriculture and the production of charcoal. The time and expense of developing alternative land use practices, livelihoods and compensation mechanisms in a truly participatory manner may be considerable.

To date, many of these expenses have been absorbed by conservation and development NGOs who generally fund such activities through alternative sources. While this has been done with the best intentions to get REDD activities to work and implemented, it does hide the true cost of REDD activities and the real price that should be paid for generated ERUs. While national-scale implementation may result in the realisation of certain cost-efficiencies, the cost of systematically developing and implementing a national-scale program include a vast network of extension officers and associated logistics and support may be significant higher than current estimates.

Lastly, the development of national-scale REDD capacity is likely to take at least five years before the start of implementation on the ground (www.forestcarbonpartnership.org). In addition to the early small-scale pilot projects being developed through the process of creating national scale capacity, it may be prudent to expedite the early implementation of REDD in larger areas that are clearly marginal in terms of alternative land use choices such as commercial agriculture. The mountains of the sub-region are an example of such an area. The mountains of the Eastern Arc and northern Mozambique are areas that may not be suitable for commercial agriculture but do provide significant climate regulation, water and soil management services in a region where demand for water is predicted to increase significantly. It may therefore be appropriate to prioritise such areas for early implementation within national REDD programs.

References

Aaheim A., Chaturvedi R. K. & Sagadevan A. A. (2010) Integrated modeling approaches the analysis of climate change impacts on forests and forest management *Mitigation and Adaptation Strategies for Global Change* DOI: 10.1007/s11027-010-9254-x.

Abegaz B. (2005) Multilateral development aid for Africa. *Economic Systems* 29, 433-54.

Accatino F., DeMichele C., Vezzoli R., Donzelli D. & Scholes R. J. (2010) Tree-grass co-existence in savanna: Interactions of rain and fire. . *Journal of Theoretical Biology* 267, 235-42.

Ali J., Benjaminsen T. A., Hammad A. A. & Dick ÿ. B. (2005) The road to deforestation: An assessment of forest loss and its causes in Basho Valley, Northern Pakistan. *Global Environmental Change Part A* 15, 370-80.

Ananda J. & Herath G. (2008) Multi-attribute preference modeling and regional land-use planning. *Ecological Economics* 65, 325-35.

Ananda J. & Herath G. (2009) A critical review of multi-criteria decision making methods with special reference to forest management and planning. *Ecological Economics* 68, 2535-48.

Angelsen A., Brown S., Loisel C., Peskett L., Streck C. & Zarin D. (2009) Reducing emissions from deforestation and forest degradation (REDD): An options assessment report. Meridian Institute - www.redd-oar.org.

Antinori C. & Sathaye J. A. (2007) Assessing transaction costs of project-based greenhouse gas emissions trading. Ernest Orlando Lawrence Berkeley National Laboratory, Berkeley, LBNL-57315.

Antle J. M. & Diagana B. (2003) Creating incentives for the adoption of sustainable agricultural practices in developing countries: The role of soil carbon sequestration. *American Journal of Agricultural Economics* 85, 1178-84.

Archibald S., Nickless A., Govender N., Scholes R. J. & Lehsten V. (2010a) Climate and the inter-annual variability of fire in southern Africa: a meta-analysis using long-term field data and satellite-derived burnt area data. *Global Ecology and Biogeography* 19, 794-809.

Archibald S., Nickless A., Scholes R. J. & Schulze R. (2010b) Methods to determine the impact of rainfall on fuels and burned area in southern African savannas. *International Journal of Wildland Fire* 19, 774-82.

Archibald S., Roy D. P., Van Wilgen B. W. & Scholes R. J. (2009) What limits fire? An examination of drivers of burnt area in Southern Africa. *Global Change Biology* 15, 613-30.

Asner G. P., Powell G. V. N., Mascaro J., Knapp D. E., Clark J. K., Jacobson J., Kennedy-Bowdoin T., Balaji A., Paez-Acosta G., Victoria E., Secada L., Valqui M. & Flint Hughes R. (2010) High-resolution forest carbon stocks and emissions in the Amazon. *Proceedings of the National Academy of Sciences* early release: http://www.pnas.org/cgi/doi/10.1073/pnas.1004875107.

Atwood D. A. (1990) Land registration in Africa: The impact on agricultural production. *World Development* 18, 659-71.

Baccini A., Laporte N., Goetz S., Sun M. & Dong H. (2008) A first map of Tropical Africa's above-ground biomass derived from satellite imagery. *Environmental Research Letters*, 045011.

Baer P. (2003) An issue of scenarios: carbon sequestration as investment and the distribution of risk. *Climatic Change* 59, 283-91.

Balling R. C., Meyer G. A. & Wells S. G. (1992) Relation of surface climate and burned area in Yellowstone National Park. *Agricultural and Forest Meteorology* 60, 285-93.

Batjes N. H. (2008) Mapping soil carbon stocks of Central Africa using SOTER. *Geoderma* 146, 58-65.

Beerling D. J. & Osborne C. P. (2006) The origin of the savanna biome. *Global Change Biology* 12, 2023-31.

Bekessy S. A. & Wintle B. A. (2008) Using carbon investment to grow the biodiversity bank. *Conservation Biology* 22, 510-3.

Bellassen V. & Gitz V. (2008) Reducing emission from deforestation and degradation in Cameroon - assessing costs and benefits. *Ecological Economics* 68, 336-44.

Bhishma P. S., Pandey S. S. & al. e. (2010) Forest Carbon Stock Measurement: Guidelines for Measuring Carbon Stocks in Community-managed Forests. ANSAB, FECOFUN, ICIMOD, and NORAD. , Kathmandu.

Bizikova L., Robinson J. & Cohen S. (2007) Linking climate change and sustainable development at the local level. *Climate Policy* 7, 271-7.

Bodie Z., Kane A. & Marcus A. J. (1996) *Investments*. IRWIN, London.

Boko M., Niang I., Nyong A., Vogel C., Githeko A., Medany M., Osman-Elasha B., Tabo R. & Yanda P. (2007) Africa. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. (eds M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden and C. E. Hansen) pp. 433-67. Cambridge University Press, Cambridge.

Bond W. J. & Keeley J. E. (2005) Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems *Trends in Ecology and Evolution* 20, 387-94.

Bond W. J. & Midgley G. F. (2000) A proposed CO2 controlled mechanism of woody plant invasion in grasslands and savannas. *Global Change Biology* 6, 865-9.

Bond W. J., Midgley G. F. & Woodward F. I. (2003a) The importance of low atmospheric CO2 and fire in promoting the spread of grasslands and savannas. *Global Change Biology* 9, 973-82.

Bond W. J., Midgley G. F. & Woodward F. I. (2003b) What controls South African vegetation - climate or fire? *South African Journal of Botany* 69, 79-91.

Bond W. J., Woodward F. I. & Midgley G. F. (2005) The global distribution of ecosystems in a world without fire. *New Phytologist* 165, 525-38.

Bowman D. M. J. S., Balch J. K., Artaxo P., Bond W. J., Carlson J. M., Cochrane M. A., D'Antonio C., DeFries R. S., Doyle J. C., Harrison S. P., Johnston F. H., Keeley J. E., Krawchuk M. A., Kull C. A., Marston J. B., Moritz M. A., Prentice I. C., Roos C. I., Scott A. C., Swetnam T. W., van der Werf G. R. & Pyne S. J. (2009) Fire in the earth system. *Science* 324, 481-4.

Boyd E., Gutierrez M. & Chang M. (2007) Small-scale forest carbon projects: Adapting CDM to low-income communities. *Global Environmental Change* 17, 250-9.

Brain S. K. & Sillen A. (1988) Evidence from the Swartkrans cave for the earliest use of fire. *Nature* 336, 464-66.

Brown S., Hall M., Andrasko K., Ruiz F., Marzoli W., Guerrero G., Masera O., Dushku A., DeJong B. & Cornell J. (2007) Baselines for land-use change in the tropics: application to avoided deforestation projects. Original Paper. *Mitigation and Adaptation Strategies for Global Change.* 12, 1001-26.

Burgess N. D., Bahane B., Clairs T., Danielsen F., Dalsgaard S., Funder M., Hagelberg N., Harrison P., Haule C., Kabalimu K., Kilahama F., Kilawe K., Lewis S. L., Lovett J. C., Lyatuu G., Marshall A. R., Meshack C., Miles L., Milledge S. A. H., Munishi P. K. T., Nashanda E., Shirima D., Swetnam R. D., Willcock S., Williams A. & Zahabu E. (2010) Getting ready for REDD+ in Tanzania: a case study of progress and challenges. *Flora and Fauna International* 44, 339-51.

Cacho O. & Lipper L. (2006) Abatement and transaction costs of carbon-sink project involving small holders. p. 22, ESA Working Paper No. 06-13 (www.fao.org/es/esa), FAO, Rome. .

Cacho O., Marshall G. R. & Milne M. (2005) Transaction and abatement costs of carbon-sink projects in developing countries. *Environment and Development Economics* 10, 597-614.

Cacho O., Wise R. & MacDicken K. (2004) Carbon monitoring costs and their effect on incentives to sequester carbon through forestry. *Mitigation and adaptation strategies for global change*, 273-93.

Cairns M. A., Brown S., Helmer E. H. & Baumgardner G. A. (1997) Root biomass allocation in the world's upland forests. *Oecologia* 111, 1-11.

Cairns R. D. & Lasserre P. (2004) Reinforcing economic incentives for carbon credits for forests. *Forest policy and economics* 6, 321-8.

CAIT. (2009) Climate Analysis Indicators Tool (CAIT) Version 6.0 (Washington, DC: World Resources Institute, http://cait.wri.org/).

Campbell B. M. (1996) The Miombo in Transition: Woodlands and Welfare in Africa. Center for International Forestry Research, Jakarta, Indonesia.

Campbell B. M., Costanza R. & van den Belt M. (2000) Introduction, overview and synthesis: Special section: Land use options in dry tropical woodland ecosystems in Zimbabwe. *Ecological Economics* 33, 341-51.

Capalbo S. M., Antle J. M., Mooney S. & Paustian K. (2004) Sensitivity of carbon sequestration costs to economic and biological uncertainties. *Environmental Management* 33, 238-51.

Carlson K. M. & Curran L. M. (2009) REDD pilot project scenarios: are costs and benefits altered by spatial scale? *Environmental Research Letters* 4, 1-3.

Casey R. & Williams M. (2010) How does fire intensity and frequency affect miombo woodland tree populations and biomass. *Ecological Applications* Early online edition.

CCAP. (2009) The Center for Clean Air Policy. Norway's proposal to auction assigned amount units: implementation options. Washington DC.

CCBA. (2005) Climate, Community and Biodiversity Project Design Standards (First Edition): at www.climate-standards.org. p. 40. CCBA, Washington DC.

Chance D. M. (2003) *Analysis of derivatives for the CFA program*. United Book Press, Baltimore.

Chazdon R. L. (2008) Beyond deforestation: restoring forests and ecosystem services on degraded lands. *Science* 320, 1458-60.

Chhatre A. & Agrawal A. (2009) Trade-offs and synergies between carbon storage and livelihood benefits from forest commons. *Proceeding of the National Academy of Sciences* http://www.pnas.org/content/early/2009/10/05/0905308106.

Chidumayo E. N. (1990) Above-ground woody biomass structure and productivity in a Zambezian woodland. *Forest ecology and management* 36, 33-46.

Chidumayo E. N. (1997) *Miombo ecology and management: an introduction*. Stockholm Environmental Institute, Stockholm.

Chomitz K. M. (2002) Baseline, leakage and measurement issues: how do forestry and energy projects compare? *Climate Policy* 2, 35-49.

Chomitz K. M. (2006) Policies for national-level avoided deforestation programs: a proposal for discussion.

Christie S. I. & Scholes R. J. (1995) Carbon storage in Eucalyptus and pine plantations in South Africa. *Environmental Monitoring and Assessment* 38, 231-41.

Clarke J., Cavendish W. & Coote C. (1996) Rural households and miombo woodlands: use, value and management. In: *The miombo in transition: woodlands and welfare in Africa* (ed B. Campbell). Center for International Forestry Research, Bogor, Indonesia.

Cleaver K. M. (1996) A strategy to develop agriculture in sub-Saharan Africa and a focus for the World Bank. In: *World Bank Technical Paper: No 203*, World Bank, Washington DC.

Cleaver K. M. & Donovan W. G. (1995) Agriculture, poverty, and policy reform in sub-Saharan Africa. In: *World Bank Discussion Papers No.280*, World Bank, Washington DC.

Clerici N. (2006) Monitoring and assessing fire impacts and land-cover change in tropical and subtropical ecosystems using satellite remote sensing and GIS techniques: PhD thesis. In: *European Union: Joint Research Centre*.

Clewell A. F. & Aronson J. (2006) Motivations for the restoration of ecosystems. *Conservation Biology*.

Climate Focus. (2008) COMESA: African Bio-Carbon Initiative: Background Document. www.comesa.int, http://climatefocus.com/news/index.html#BioCarb.

Cochrane M. A. & Laurance W. F. (2002) Fire as a Large-Scale Edge Effect in Amazonian Forests. *Journal of Tropical Ecology* 18, 311-25.

Coetsee C., February E. C. & Bond W. J. (2008) Nitrogen availability is not affected by frequent fire in a South African savanna. *Journal of Tropical Ecology* 24, 647-54.

Combes Motel P., Pirard R. & Combes J. L. (2009) A methodology to estimate impacts of domestic policies on deforestation: Compensated Successful Efforts for "avoided deforestation" (REDD). *Ecological Economics* 68, 680-91.

Conrad S. G. & Soloman A. M. (2009) Effects of wildland fire on regional and global carbon stocks in a changing environment. *Developments in Environmental Science* 8, 109-38.

Cotula L. & Mayers J. (2009) Tenure in REDD - Start-point or afterthought? . In: *Natural Resource Issues No. 15*. International Institute for Environment and Development, London.

Crittenden J. & Wilder M. (2008) Bringing the forest to market: Structuring avoided deforestation projects. *The Carbon and Climate Law Review* 3, 273-82.

de Ronde C., Everson T. M. & Everson C. S. (2004a) Fire management in rural areas and industrial forestry plantations. In: *Wildland fire management handbook for sub-Saharan Africa* (eds J. G. Goldammer and C. de Ronde). www.compress.co.za, www.oneworldbooks.com, Johannesburg.

Defra. (2010) 2010 Guidelines to Defra / DECC's GHG conversion factors for company reporting. Department of Energy and Climate Change and Department of Environment, Food and Rural Affairs, United Kingdom.

Denman K. L., Brasseur G., Chidthaisong A., Ciais P., Cox P. M., Dickinson R. E., Hauglustaine D., Heinze C., Holland E., Jacob D., Lohmann U., Ramachandran S., da Silva Dias P. L., Wofsy S. C. & Zhang X. (2007) Couplings between changes in the climate system and biogeochemistry. In: *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (eds S. Soloman, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor and H. L. Miller) p. 90, Cambridge.

Desanker P. V., Frost P. G. H., Justice C. O. & Scholes R. J. (1997) IGBP Report 41: The Miombo Network: Framework for a terrestrial transect study of land-use and land-cover change in the Miombo ecosystems of Central Africa. p. 110. International Geosphere-Biosphere Programme, Stockholm.

Doherty R. M., Sitch S., Smith B., Lewis S. L. & Thornton P. K. (2010) Implications of future climate and atmospheric CO2 content for regional biogeochemisty, biogeography and ecosystem services across East Africa. *Global Change Biology* 16, 617-40.

Ebeling J. & Yasue M. (2008) Generating carbon finance through avoided deforestation and its potential to create climatic, conservation and human development benefits. *Philosophical transactions of the Royal Society: B* 363, 1917-24.

ECCM. (2007) Establishing mechanisms for payments for carbon environmental services in the Eastern Arc Mountains, Tanzania. p. 46. Edinburgh Centre for Carbon Management, Edinburgh.

EcoSecurities. (2002) Clean Development Mechanism (CDM): simplified modalities and procedures for small-scale projects. EcoSecurities.

Ellery W. N., Scholes R. J. & Mentis M. T. (1991) An initial approach to predicting the sensitivity of the South African grassland biome to climate change. *South African Journal of Science* 87, 499-503.

Engelbrecht F. A., McGregor J. L. & Engelbrecht C. J. (2008) Dynamics of the conformal-cubic atmospheric model projected climate-change signal over southern Africa. *Journal of Climatology* DOI: 10/1002/joc.1742.

Ethiopia. (2010) Federal Democratic Republic of Ethiopia, Readiness Preparation Proposal (R-PP), World Bank Forest Carbon Partnership Facility. (ed www.forestcarbonpartnership.org).

Everson C. S. (1985) Ecological effects of fire in the montane grasslands of Natal. In: *Department of Grassland Science, Faculty of Agriculture* p. 236. University of Natal, Pietermaritzburg.

Everson C. S., Molefe G. L. & Everson T. M. (1998) Monitoring and modeling components of the water balance in a grassland catchment in the summer rainfall area of South Africa: WRC Report No 493/1/98. p. 138. Water Research Commission, Pretoria.

FAO. (2006) Global forest resources assessment 2005: Progress towards sustainable forest management. In: *FAO Forestry Paper 147*. Food and agriculture organization of the United Nations Rome.

FAO. (2008) Understanding forest tenure in Africa: opportunities and challenges for forest tenure diversification. In: *Forestry Policy and Institutions Working Paper No. 19.* , Rome.

Farley K. A., Jobbagy E. G. & Jackson R. B. (2005) Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology* 11, 1-12.

Faures J.-M. & Santini G. (2008) Water and the rural poor: Interventions for improving livelihoods in sub-Saharan Africa. FAO Land and Water Division. Food and Agriculture Organization of the United Nations (FAO) and IFAD, Rome.

Fearnside P. M. (2000) Uncertainty in land-use change and forestry sector mitigation options for global warming: plantation silviculture versus avoided deforestation. *Biomass and Bioenergy* 18, 457-68.

Fearnside P. M. (2002) Why a 100-Year Time Horizon should be used for GlobalWarming Mitigation Calculations *Mitigation and adaptation strategies for global change* 7, 19-30.

Felzer B., Kicklighter D. W., Melillo J. M., Wang C., Zhuang Q. & Prinn R. (2004) Effects of ozone on net primary production and carbon sequestration in the conterminous United States using a biogeochemistry model. *Tellus B* 56, 230-48.

Fischlin A., Midgley G. F., Price J. T., Leemans R., Gopal B., Turley C., Rounsevell M. D. A., Dube O. P., J. T. & Velichko A. A. (2007) Ecosystems, their properties, goods and services.Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change,. (eds M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden and C. E. Hanson), Cambridge University Press, Cambridge. 211-272.

Frost P. (1996) The ecology of miombo woodlands. In: *The miombo in transition: woodlands and welfare in Africa* (ed B. Campbell). Center for International Forestry Research, Bogor, Indonesia.

Geldenhuys C. J. (1994) Bergwind fires and the location pattern of forest patches in the southern Cape landscape, South Africa. *Journal of Biogeography* 21, 49-62.

Gerten D., Luo Y., le Maire G., Parton W. J., Keough C., Dukes J. S., Hanson P. J., Knapp A. A. K., Linder S., Nepstad D. C., Rustad L. & Sowerby A. (2008) Modelled effects of precipitation on ecosystem carbon and water dynamics in different climatic zones. *Global Change Biology* 14, 2365-79.

Gitay H. & Noble I. (1997) What are functional types and how should we seek them? In: *Plant functional types: their relevance to ecosystem properties and global change* (eds T. M. Smith, H. H. Shugart and F. I. Woodward). Cambridge University Press, Cambridge.

Grieg-Gran M. (2006) The cost of avoiding deforestation: report prepared for the Stern Review of the Economics of Climate Change. p. 18. IIED: International Institute for Environment and Development, London.

Gullison R. E., Frumhoff P. C., Canadell J. G., Field C. B., Nepstad D. C., Hayhoe K., Avissar R., Curran L. M., Friedlingstein P., Jones C. D. & Nobe C. (2007) Tropical forests and climate policy. *Science* 316, 985.

Guo Z., Xiao X., Gan Y. & Zheng Y. (2001) Ecosystem functions, services and their values - a case study in Xingshan County of China. *Ecological Economics* 38, 141-54.

Guy P. R. (1989) The Influence of Elephants and Fire on a Brachystegia-Julbernardia Woodland in Zimbabwe. *Journal of Tropical Ecology* 5, 215-26.

Hall D. O., Ojima D. S., Parton W. J. & Scurlock J. M. O. (1995) Response of temperate and tropical grasslands to CO2 and climate change. *Journal of Biogeography* 22, 537-47.

Hamilton K., Sjardin M., Marcello T. & Xu G. (2008) Forging a frontier: state of the voluntary carbon markets 2008. p. 77pp. New Energy Finance and the Forest Trends Association, New York

Hanan N. P., Sea W. B., Dangelmayr G. & Govender N. (2008) Do fires in savannas consume woody biomass? A comment on approaches to modeling savanna dynamics. *The American Naturalist* 171, 851-6.

Heelemann S., Proches S., Rebelo A. G., Van Wilgen B. W., Porembski S. & Cowling R. M. (2008) Fire season effects on the recruitment of non-sprouting serotinous Proteaceae in the eastern (bimodal rainfall) fynbos biome, South Africa. *Austral Ecology* 2008, 119-27.

Higgins S. I., Bond W. J. & Trollope S. W. (2000) Fire, sprouting and variability: a recipe for grass-tree coexistance in savanna. *Journal of Ecology* 88, 213-29.

Higgins S. I., Shackleton C. M. & Robinson E. R. (1999) Changes in woody community structure and composition under contrasting landuse systems in a semi-arid savanna, South Africa. *Journal of Biogeography* 26, 619-27.

Hoffmann W. A., Orthen B. & Nascimento P. K. V. d. (2003) Comparative fire ecology of tropical savanna and forest trees. *Functional Ecology* 17, 720-6.

Holdo R. M. (2007) Elephants, fire, and frost can determine community structure and composition in Kalahari woodlands. *Ecological Applications* 17, 558-68.

Holling C. S. (1973) Resilience and stability of ecological systems *Annual Review of Ecology and Systematics* 4, 1-23.

Hooper D. U., Solan M., Symstad A., Diaz S., Gessner M. O., Buchmann N., Degrange V., Grime P., Hulot F., Mermillod-Blondin F., Roy J., Spehn E. & van Peer L. (2002) Species diversity, functional diversity and ecosystem functioning: Chapter 17. In: *Biodiversity and ecosystem functioning: Synthesis and perspectives* (eds M. Loreau, S. Naeem and P. Inchausti) pp. 195-207. Oxford University Press, Oxford.

IPCC. (2003) Good practice guidance for land use, land-use change and forestry. (eds J. Penman, M. Gytarsky, T. Hiraishi, T. Krug, D. Kruger, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe and F. Wagner). Intergovernmental Panel on Climate Change, http://www.ipcc-nggip.iges.or.jp, Kanagawa, Japan.

IPCC. (2007) Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. (eds S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor and H. L. Miller) p. 996. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Jackson R. B., Canadell J., Ehleringer J. R., Mooney H. A., Sala O. E. & Schulze E. D. (1996) A Global Analysis of Root Distributions for Terrestrial Biomes. *Oecologia* 108, 389-411.

James D. (2006) *Gaining ground?: Rights and Property in South African Land Reform*. Routledge-Cavendish, London.

Jindal R., Swallow B. & Kerr J. T. (2008) Forestry-based carbon sequestration project in Africa: Potential benefits and challenges. *Natural Resources Forum* 32, 116-30.

Johns T., Merry F., Stickler C., Nepstad D., Laporte N. & Goetz S. (2008) A three-fund approach to incorporating government, public and private forest stewards into a REDD funding mechanism. *International Forestry Review* 10, 458 - 64.

Johnson D. W. (2006) Progressive N limitation in forests: Review and implications for long-term responses to elevated CO2. *Ecology* 81, 64-75.

Justice C. O., Giglio L., Korontzi S., Owens J., Morisette J. T., Roy D. P., Descloitres J., Alleaume S., Petitcolin F. & Kaufman Y. (2002) The MODIS fire products. *Remote Sensing of Environment* 83, 244-62.

Kamelarczyk K. B. F. (2009) Carbon stock assessment and modeling in Zambia: A UN-REDD programme study. In: *UN-REDD Programme, MRV Working Paper 4*. Food and Agriculture Organization of the United Nations.

Kennedy A. D. & Potgieter A. L. F. (2003) Fire season affects size and architecture of *Colophospermum mopane* in southern African savannas. *Plant Ecology* 167, 179-92.

Kenya. (2010) Kenya, Readiness Preparation Proposal (R-PP), World Bank Forest Carbon Partnership Facility. (ed www.forestcarbonpartnership.org).

Kgope B. S., Bond W. J. & Midgley G. F. (2010) Growth responses of African savanna trees implicate atmospheric [CO2] as a driver of past and current changes in savanna tree cover. *Austral Ecology* 35, 451-63.

Kiang J. E. & Eltahir E. A. B. (1999) Role of ecosystem dynamics in biosphere-atmosphere interaction over the coastal region of West Africa. *Journal of Geophysical Research* 104, 31173-89.

Kimball B. A., Mauney J. R., Nakayama F. S. & Idso S. B. (1993) Effects of increasing atmospheric CO2 on vegetation. *Plant Ecology* 104-105, 65-75.

Kindermann G., Obersteiner M., Sohngen B., Sathaye J. A., Andrasko K., Rametsteiner E., Schlamadinger B., Wunder S. & Beach R. (2008) Global cost estimates of reducing carbon emissions through avoided deforestation. *Proceedings of the National Academy of Sciences* 105.

King D. M. (2004) Trade-based carbon sequestration accounting. *Environmental Management* 33, 559-71.

Knight F. H. (1921) Risk, Uncertainty and Profit. Houghton Mifflin, Boston.

Knoke T., Stimm B. & Weber M. (2008) Tropical farmers need productive alternatives. *Nature* 452, 934.

Knowles T. & Pienaar E. (2010) Enkangala ecosystem services review: An assessment of the effect of historical grazing and fire regimes on below-ground carbon stocks in temperature grassland systems. (Submitted to WWF South Africa).

Knowles T., von Maltitz G. & Makhado R. (2008) The uKhahlamba - Drakensberg Park World Heritage Site: Carbon sequestration feasibility and implementation assessment. CSIR Report for Ezemvelo KZN Wildlife. CSIR, Pretoria, South Africa.

Koenig R. (2008) Critical time for African rainforests. *Science* 320, 1439-41.

Kowero G., Campbell B. M. & Sumailia U. R. (2003) Policies and Governance Structures in Woodlands of Southern Africa. Center for International Forestry Research, Jakarta, Indonesia.

Krey M. (2004) Transaction costs of CDM projects in India - an empirical survery (HWWA Report 238). p. 132. HWWA - Hamburg, Hamburg.

Krey M. (2005) Transaction costs of unilateral CDM projects in India - results from an empirical survey. *Energy Policy* 33, 2385-97.

Kutsch W. L., Hanan N., Scholes R. J., McHugh I., Kubheka W., Eckhard H. & Williams C. (2008) Response of carbon fluxes to water relations in a savanna ecosystem in South Africa. *Biogeosciences* 5, 1797-808.

Laurikka H. & Springer U. (2003) Risk and return of project-based climate change mitigation: a portfolio approach. *Global Environmental Change* 13, 207-17.

Laws R. M. (1970) Elephants as Agents of Habitat and Landscape Change in East Africa. *Oikos* 21, 1-15.

Laye O. & Gaye I. (2008) Introduction: Driving forward sustainable development agendas in Africa. *Natural Resources Forum* 32, 90-2.

Lechmere-Oertel R. G., Kerley G. I. H. & Cowling R. M. (2005) Patterns and implications of transformation in semi-arid succulent thicket, South Africa. *Journal of Arid Environments* 62, 459-74.

Little G. (2006) Accelerating the implementation of the Clean Development Mechanism in South African Industry. In: *Gordon Institute of Business Science* p. 168. University of Pretoria, Pretoria.

Luo Y., Gerten D., le Maire G., Parton W. J., Weng E., Zhou X., Keough C., Beier C., Ciais P., Cramer W., Dukes J. S., Emmett B., Hanson P. J., Knapp A. A. K., Linder S., Nepstad D. C. & Rustad L. (2008) Modeled interactive effects of precipitation, temperature, and [CO2] on ecosystem carbon and water dynamics in different climatic zones. *Global Change Biology* 14, 1986-99.

Luo Y., Hui D. & Zhang D. (2006) Elevated CO2 stimulates net accumulations of carbon and nitrogen in land ecosystems: a meta-analysis. *Ecology* 87, 53-63.

Luoga E. J., Witkowski E. T. F. & Balkwill K. (2004) Regeneration by coppicing (resprouting) of miombo (African savanna) trees in relation to land use. *Forest Ecology and Management* 189, 23-35.

Manson A. D., Jewitt D. & Short A. D. (2007) Effects of season and frequency of burning on soils and landscape functioning in a moist montane grassland. *African Journal of Range and Forage Science* 24, 9-18.

Marland G., Fruit K. & Sedjo R. (2001) Acccounting for sequestered carbon: the question of permanence. *Environmental Science and Policy* 4, 259-68.

May R. M. (1977) Thresholds and breakpoints in ecosystems with a multiplicity of stable states. *Nature* 269, 471-7.

Mbilinyi B. P., Malimbwi R. E., Shemwetta D. T. K., Songorwa, Zahabu E., Katani J. Z. & Kashaigili J. (2006) Forest area baseline for the Eastern Arc Mountains. Conservation and Management of the Eastern Arc Mountain Forests, Forestry and Beekeeping Division, Dar es Salaam.

Mellilo J. M., McGuire A. D., Kicklighter D. W., Moore B., Vorosmarty C. J. & Schloss A. L. (1993) Global climate change and terrestrial net primary production. *Nature* 363, 234 - 40.

Michaelowa A. & Jotzo F. (2005) Transaction costs, institutional rigidities and the size of the Clean Development Mechanism. *Energy Policy* 33, 511 - 23.

Michaelowa A., Stronzik M., Eckermann F. & Hunt A. (2003) Transaction costs of the Kyoto mechanisms. *Climate Policy* 3, 261-78.

Mickels-Kokwe G. (2006) Small-scale woodland-based enterprises with outstanding economic potential: the case of honey in Zambia. p. 82, Center for International Forestry Research, Bogor, Indonesia.

Midgley G. F., Chapman R. A., Mukheibir P., Tadross M., Hewitson B., Wand S., Schulze R. E., Lumsden T., Horan M., Warbuton M., Kgope B., Knowles T., Abayomi A., Ziervogel G., Cullis R. & Theron A. (2007) Assessing impacts, vulnerability and adaptation in key South African sectors: A background study for the Long Term Mitigation Scenarios assessment. Energy Research Centre, Cape Town.

Midgley J. J., Lawes M. J. & Chamaillé-Jammes S. (2010) Turner Review No. 19. Savanna woody plant dynamics: the role of fire and herbivory, separately and synergistically. *Australian Journal of Botany* 58, 1-11.

Miles L. & Kapos V. (2008) Reducing greenhouse gas emissions from deforestation and forest degradation: Global land-use implications. *Science* 320, 1454-5.

Miles L. & Valerie K. (2008) Reducing greenhouse gas emissions from deforestation and forest degradation: Global land-use implications. *Science* 320, 1454-5.

Milledge S. & Elibariki R. (2005) The status of logging in southern Tanzania, A technical report submitted to the WWF East Africa Coastal Forests Ecoregion Programme. TRAFFIC East/Southern Africa, Dar es Salaam.

Mills A. J. & Cowling R. M. (2004) Rate of carbon sequestration at two thicket restoration sites in the Eastern Cape, South Africa. *Restoration Ecology* 14, 38-49.

Mills A. J., Cowling R. M., Fey M. V., Kerley G. I. H., Donaldson J., Lechmere-Oertel R. G., Sigwela A. M., Skowno A. & Rundel P. (2005) Effects of goat pastoralism on ecosystem carbon storage in semiarid thicket, Eastern Cape, South Africa. *Austral Ecology* 30, 797-804.

Milne G. (1935) Some suggested units of classification and mapping, particularly for East African soils. *Soil Research* 4, 183-98.

Mlambo D. & Mapaure I. (2006) Post-fire resprouting of Colophospermum mopane saplings in a southern African savanna. *Journal of Tropical Ecology* 22, 231-4.

Mlambo D., Nyathi P. & Mapaure I. (2005) Influence of Colophospermum mopane on surface soil properties and understorey vegetation in a southern African savanna. *Forest Ecology and Management* 212, 394-404.

Mokany K., Raison R. J. & A.S. P. (2006) Critical analysis of root : shoot ratios in terrestrial biomes. *Global Change Biology* 12, 84-96.

Mooney S., Antle J. M., Capalbo S. M. & Paustian K. (2004) Influence of project scale and carbon variability on the costs of measuring soil carbon credits. *Environmental Management* 33, 252-63.

Moura-Costa P. (2009) Compensation for carbon stock maintenance in forests as an alternative to avoiding carbon flows. Oxford Centre for Tropical Forests, Environmental Change Institute, University of Oxford, UK.

Myers E. (2007) Policies to Reduce Emissions from Deforestation and Degradation (REDD) in Tropical Forests: An examination of the issues facing the incorporation of REDD into market-based climate policies. p. 83. Resources for the Future, www.rff.org, RFF DP 07-50, Washington, D.C.

Nakicenovic N., Alcamo J., Davis G., De Vries B., Fenhann J., Gaffin S., Gregory K., Grubler A., Yong Jung T., Kram T., Lebre La Rovere E., Michaelis L., Mori S., Morita T., Pepper W., Pitcher H., Price L., Riahi K., Roehrl A., Rogner H.-H., Sankovski A., Schlesinger M., Shukla P., Smith S., Swart R., Van Rooijen S., Victor N. & Dadi Z. (2000) *Special Report of Emissions Scenarios: a special report of working group III of the IPCC*. Cambridge University Press, *Executive summary available on http://www.ipcc.ch/pub/reports.htm*, Cambridge.

Noble I. (2003) Carbon finance and rangeland management. In: *VIIth International Rangelands Conference* (eds N. Allsopp, A. R. Palmer, S. J. Milton, K. P. Kirkman, G. I. H. Kerley, C. R. Hurt and C. J. Brown), Durban, South Africa.

Ojima D. S., Parton W. J., Coughenour M. B., Scurlock J. M. O., Kirchner T., Kittel T. G. F., Hall D. o., Schimel D. S., Garcia Moya E., Gilmanov T. G., Seastedt T., Kamnalrut A., Kinyamario J. I., Long S. P., Menaut J.-C., Sala O. E., Scholes R. J. & van Veen J. A. (1996) Impact of climate and atmospheric carbon dioxide changes on grasslands of the world. In: *Global change effects on coniferous forests and grasslands* (eds A. J. Breymeyer, D. O. Hall, J. M. Melilo and G. I. Agren). John Wiley & Sons Ltd. .

Ormeno E., Cespedes B., Sanchez I. A., Velasco-Garcia A., Moreno J. M., Fernandez C. & Baldy V. (2009) The relationship between terpenes and flammability of leaf litter. *Forest Ecology and Management* 257, 471-82.

Osborne T. & Kiker C. (2005) Carbon offsets as an economic alternative to large-scale logging: a case study in Guyana. *Ecological Economics* 52, 481-96.

Ostrom E. (1990) *Governing the Commons: The Evolution of Institutions for Collective Actions*. Cambridge University Press, Cambridge.

Ostrom, E. (2007) A diagnostic approach for going beyond panaceas. *Proceedings of the National Academy of Sciences* 104, 15181–15187.

Ostrom E. (2009) A general framework for analyzing sustainability of socio-ecological systems. *Science* 325.

Otter L. B., Scholes R. J., Dowty P. R., Privette J. L., Caylor K. K., Ringrose S., Mukelabai M., Frost P., Hanan N., Totolo O. & Veenendaal E. M. (2002) The Southern African Regional Science Initiative (SAFARI 2000): wet season campaigns. *South African Journal of Science* 98, 131-7.

Page S. E., Siegert F., Rieley J. O., Boehm H.-D. V., Jaya A. & Limin S. (2002) The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature* 420, 61-5.

Parton W. J., Scurlock J. M. O., Ojima D. S., Gilmanov T. G., Scholes R. J., Schimel D. S., Kirchner T., Menaut J.-C., Seastedt T., Garcia E. G., Kamnalrut A. & Kinyanario J. I. (1993) Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Geochemical Cycles* 7, 785-809.

Paterson D. G. & Steenkamp P. I. (2003) Soil survey of the Letaba exclosure: ISCW Report No. GW/A/2003/38. p. 13. Agricultural Research Council, Institute for Soil, Climate and Water, Pretoria.

Pechony O. & Shindell D. T. (2010) Driving forces of global wildfires over the past millennium and the forthcoming century. *Proceedings of the National Academy of Sciences* 107, 19167-70.

Peng C. & Apps M. J. (1999) Modeling the response of net primary productivity (NPP) of boreal forest ecosystems to changes in climate and fire disturbance regimes. *Ecological Modelling* 122, 175-93.

Penman J., Gytarsky M., Hiraishi T., Krug T., Kruger D., Pipatti R., Buendia L., Miwa K., Ngara T., Tanabe K. & Wagner F. (2003) Good practice guidance for land use, land-use change and forestry. Intergovernmental Panel on Climate Change.

Peters-Guarin G. & McCall M. K. (2010) CCF for REDD. Using CyberTracker for Mapping and Visualising Community Forest Management. . Enschede, University of Twente, ITC; and Morelia: UNAM, CIGA. http://www.communitycarbonforestry.org.

Pierce J. L., Meyer G. A. & Timothy Jull A. J. (2004) Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. *Nature* 432, 87-90.

Pinheiro A., Privette J. L., Tucker C. J., Entekhabi D. & Berry J. A. (2001) Assessing the relationship between temperature and soil moisture in southern Africa. *Remote sensing and hydrology*, 1-6.

Place F. (2009) Land Tenure and Agricultural Productivity in Africa: A Comparative Analysis of the Economics Literature and Recent Policy Strategies and Reforms. *World Development* 37, 1326-36.

Platteau J.-P. (1995) Reforming land rights in sub-Saharan Africa: Issues of efficiency and equity. In: *United Nations Research Institute for Social Development* p. pp44, Geneva, Switzerland.

Porteous D. (2006) The enabling environment for mobile banking in Africa. p. 57. Commissioned by the Department for International Development (DFID), Bankable Frontier Associates, www.bankablefrontier.com, Boston.

Potvin C., Chapin F. S., Gonzalez A., Leadley P., Reich P. B. & Roy J. (2007) Plant biodiversity and responses to elevated carbon dioxide. In: *Terrestrial ecosystems in a changing world* (eds J. Canadell, D. Pataki and L. Pitelka). Springer, New York

Potvin C., Guay B. & Pedroni L. (2008) Is reducing emissions from deforestation financially feasible? A Panamanian case study. *Climate Policy* 8, 23-40.

RECOFTC. (2010) Forests and Climate Change after Copenhagen. An Asia-Pacific Perspective. . RECOFTC, Bangkok

Reilly F. K. & Brown K. C. (2003) *Investment Analysis and Portfolio Management*. Thomson South-Western, United States.

Ribeiro N. S., Shugart H. H. & Washington-Allen R. (2008) The effects of fire and elephants on species composition and structure of the Niassa Reserve, northern Mozambique. *Forest Ecology and Management* 255, 1626-36.

Richards K. R. & Stokes C. (2004) A review of forest carbon sequestration cost studies: a dozen years of research. *Climatic Change* 63, 1-48.

Ringius L. (2002) Soil Carbon Sequestration and the CDM: Opportunities and Challenges for Africa. *Climatic Change* 54, 471-95.

Rosenzweig C., Karoly D., Vicarelli M., Neofotis P., Wu Q., Casassa G., Menzel A., Root T. L., Estrella N., Seguin B., Tryjanowski P., Liu C., Rawlins S. & Imeson A. (2008) Attributing physical and biological impacts to anthropogenic climate change. *Nature* 453, 353-8.

Rubinstein M. (2002) Markowitz's "Portfolio Selection": A fifty-year retrospective. *Journal of Finance* 112, 1041-5.

Running S. W. (2006) Climate Change: Is Global Warming Causing More, Larger Wildfires? *Science* 313, 927-8.

Sankaran M., Hanan N. P., Scholes R. J., Ratnam J., Augustine D. J., Cade B. S., Gignoux J., Higgins S. I., Le Roux X., Ludwig F., Ardo J., Banyikwa F., Bronn A., Bucini G., Caylor K. K., Coughenour M. B., Diouf A., Ekaya W., Feral C. J., February E., Frost P. G. H., Hiernaux P., Hrabar H., Metzger K. L., Prins H. H., Ringrose S., Sea W., Tews J., Worden J. & Zambatis N. (2005) Determinants of woody cover in African savannas. *Nature* 438, 846-9.

Santilli M., Moutinho P., Schwartzman S., Nepstad D. C., Curran L. M. & Nobre C. A. (2005) Tropical deforestation and the Kyoto Protocol. *Climatic Change* 71, 267-76.

Scheffer M. & Carpenter S. R. (2003) Catastrophic regime shifts in ecosystems: linking theory to observation *Trends in Ecology and Evolution* 18, 648-57.

Scheiter S. & Higgins S. I. (2009) Impacts of climate change on the vegetation of Africa: an adaptive dynamic vegetation modelling approach. *Global Change Biology* 15, 2224-46.

Scholes M. C., Powlson D. & Tian G. (1997) Input control of organic matter dynamics. *Geoderma* 79, 25-47.

Scholes M. C., Scholes R. J., Otter L. & Woghiren A. J. (2003) Biochemistry: the cycling of elements. In: *The Kruger Experience: Ecology and management of savanna heterogeneity* (eds J. T. Du Toit, K. H. Rogers and H. C. Biggs). Island Press, London.

Scholes R. J. (1987) Response of three semi-arid savannas on contrasting soil to the removal of the woody component. In: *Faculty of Science* p. 297pg. University of the Witwatersrand, Johannesburg.

Scholes R. J. (1990) The regrowth of *Colophospermum mopane* following clearing. *Journal of the Grassland Society of southern Africa* 7, 147-51.

Scholes R. J. (2003) Convex relationships in ecosystems containing mixtures of trees and grasses. *Journal of Environmental and Resource Economics* 26.

Scholes R. J. (2009) Syndromes of dryland degradation in southern Africa. *African Journal of Range and Forage Science* 26, 113-25.

Scholes R. J. & Archer S. R. (1997) Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics* 28, 517-44.

Scholes R. J. & Biggs R. (2004) Ecosystem services in southern Africa: a regional assessment - A contribution to the Millennium Ecosystem Assessment, prepared by the regional-scale team of the Southern African Millennium Ecosystem Assessment. CSIR, Pretoria, South Africa. .

Scholes R. J., Gureja N., Giannecchinni M., Dovie D., Wilson B., Davidson N., Piggott K., McLoughlin C., van der Velde K., Freeman A., Bradley S., Smart R. & Ndala S. (2001) The environment and vegetation of the flux measurement site near Skukuza, Kruger National Park. *Koedoe* 44, 73-83.

Scholes R. J. & Hall D. O. (1996) The carbon budget of tropical savannas, woodlands and grasslands. In: *Global change: Effects on coniferous forests and grasslands* (eds A. I. Breymeyer, D. O. Hall, J. M. Mellilo and G. I. Agren). John Wiley and Sons Ltd, London.

Scholes R. J., Kendall J. & Justice C. O. (1996) The quantity of biomass burned in southern Africa. *Journal of Geophysical Research* 101, 23667 - 76.

Scholes R. J., Pickett G., Ellery W. N. & Blackmore A. C. (1997) Plant functional types in African savannas and grasslands. In: *Plant functional types: their relevance to ecosystem properties and global change* (eds T. M. Smith, H. H. Shugart and F. I. Woodward). Cambridge University Press, Cambridge.

Scholes R. J. & Walker B. H. (1993) *An African savanna: synthesis of the Nylsvley study*. Cambridge University Press, Cambridge.

Scholze M., Knorr W., Arnelll N. W. & Prentice C. (2006) A climate-change risk analysis for world ecosystems. *Proceedings of the National Academy of Sciences* 103, 13116-20.

Sea W. B. & Hanan N. (2011) Self-thinning and tree competition in savannas. *Biotropica* DOI: 10.1111/j.1744-7429.2011.00789.x.

Shackleton C., Griffin N. J., Banks D. I., J.M M. & Shackleton S. E. (1994) Community structure and species composition along a disturbance gradient in a communally managed South African savanna. *Vegetatio* 115, 157-67.

Shackleton C. M. (1997) The prediction of woody productivity in the savanna biome, South Africa. In: *Faculty of Science* p. 210pp. University of the Witwatersrand, Johannesburg.

Shackleton C. M., Shackleton S. E., Buiten E. & Bird N. (2007) The importance of dry woodlands and forests in rural livelihoods and poverty alleviation in South Africa. *Forest policy and economics* 9, 558 - 77.

Shea R. W., Shea B. W., Kauffman J. B., Ward D. E., Haskins C. I. & Scholes M. C. (1996) Fuel biomass and combustion factors associated with fires in savanna ecosystems of South Africa and Zambia. *Journal of Geophysical Research* 101, 23551-68.

Skutsch M. (2005) Reducing carbon transaction costs in community-based forest management. *Climate Policy* 5, 433-43.

Skutsch M. (2010) *Community Forest Monitoring for the Carbon Market: Opportunities under REDD*. Earthscan, London.

Skutsch M., Bird N., Trines E., Dutschke M., Frumhoff P., de Jong B. H. J., van Laake P., Masera O. & Murdiyarso D. (2007) Clearing the way for reducing emissions from tropical deforestation. *Environmental Science & Policy* 10, 322-34.

Smit G. N. (2004) An approach to tree thinning to structure southern African savannas for long-term restoration from bush encroachment. *Journal of Environmental Management* 71, 179-91.

Smith P., Martino D., Cai Z., Gwary D., Janzen H. H., Kumar P., McCarl B. A., Ogle S. M., O'Mara F., Rice C., Scholes R. J. & Sirotenko O. (2007) Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. (eds B. Metz, O. R. Davidson, R. Bosch, R. Dave and M. L.), Cambridge University Press, Cambridge, United Kingdom.

Snyman H. A. & Fouche H. J. (1991) Production and water-use effeciency of semi-arid grasslands of South Africa as affected by veld condition and rainfall. *Water SA* 17, 263-8.

Song C. & Woodcock C. E. (2003) A regional forest ecosystem carbon budget model: impacts of forest age structure and landuse history. *Ecological Modelling* 164, 33-47.

SSN. (2006) Renewable energy and energy efficiency in Africa: Carbon finance guide. p. 37. South South North (www.southsouthnorth.org), Tanzania and Mozambique.

Stern N. (2006) The Economics of Climate Change. p. 712. Cabinet Office - HM Treasury (www.hm-treasury.gov.uk), Cambridge.

Strassburg B., Turner R. K., Fisher B., Schaeffer R. & Lovett A. (2009) Reducing emissions from deforestation--The "combined incentives" mechanism and empirical simulations. *Global Environmental Change* 19, 265-78.

Streck C., Gomez-Echeverri L., Gutman P., Loisel C. & Werksman J. (2009) REDD+ Instiutional Options Assessment: Developing an efficient, effective and equitable institutional framework for REDD+ under the UNFCCC. p. 30pp. Meridian Institute - www.redd-oar.org.

Streck C. & O'Sullivan R. (2008) Harnessing carbon finance for Africa's rural poor. p. 13pp. Climate Focus - http://www.climatefocus.com/, Rotterdam.

Subak S. (2003) Replacing carbon lost from forests: an assessment of insurance, reserves, and expiring credits. *Climate Policy* 3, 107-22.

Swart R. & Raes F. (2007) Making integration of adapation and mitigation work: mainstreaming into sustainable development policies? *Climate Policy* 7, 288-303.

Tacconi L. (2009) Compensated successful efforts for avoided deforestation vs compensated reductions. *Ecological Economics* 68, 2469-72.

Tanzania. (2010) United Republic of Tanzania, Readiness Preparation Proposal (R-PP), World Bank Forest Carbon Partnership Facility. (ed www.forestcarbonpartnership.org).

TEEB. (2010) The economics of ecosystems and biodiversity: Mainstreaming th economics of nature: A synthesis of the approach, conclusions and recommendations of the TEEB.

Theron L. J. (2009) Carbon stock quantification: training and field manual. Peace Parks Foundation Climate Change Programme, Stellenbosch.

Thomas C. D., Cameron A., Green R. E., Bakkenes M., Beaumont L. J., Collingham Y. C., Erasmus B. F. N., Ferreira de Siqueira M., Grainger A., Hannah L., Hughes L., Huntley B., van Jaarsveld A. S., Midgley G. F., Miles L., Ortega-Huerta M. A., Peterson A. T., Phillips O. L. & Williams S. E. (2004) Extinction risk from climate change. *Nature* 427, 145-8.

Thompson I., Mackey B. G., McNulty S. & Mosseler A. (2009) Forest resilience, biodiversity and climate change. A synthesis of the biodiversity / resilience / stability relationship in forest ecosystems. In: *Secretariat of the Convention on Biological Diversity: Technical Series no.43* p. pp 67, Montreal.

Thonicke K., Venevsky S., Sitch S. & Cramer W. (2001) The role of fire disturbance for global vegetation dynamics: coupling fire into a Dynamic Global Vegetation Model. *Global Ecology and Biogeography* 10, 661-77.

Thornton P. K., Jones P. G., Owiyo T., Krusha R. L., Herrero M., Kristjanson P., Notenbaert A., Bekele N., Omolo A., Orindi V., Otiende B., Ochieng A., Bhadwal S., Anantram K., Nair S., Kumar V. & Kulkar U. (2006) Mapping climate vulnerability and poverty in Africa. Report to the Department for International Development. ILRI, Nairobi, Kenya.

Torres A. B., Marchant R., Lovett J. C., Smart J. C. R. & Tipper R. (2009) Analysis of the carbon sequestration costs of afforestation and reforestation agroforestry practices and the use of cost curves to evaluate their potential for implementation of climate change mitigation. *Ecological Economics* In Press, Corrected Proof.

Trodd N. M. & Dougill A. J. (1998) Monitoring vegetation dynamics in semi-arid African rangelands: Use and limitations of Earth observation data to characterize vegetation structure. *Applied Geography* 18, 315-30.

Trollope S. W., Trollope L. A., Potgieter A. L. F. & Zambatis N. (1996) SAFARI-92 characterization of biomass and fire behavior in the small experimental burns of the Kruger National Park. *Journal of Geophysical Research* 101, 531-23.

Trollope W. S. W. (2007) Fire - a key factor in the ecology and management of African grasslands and savannas. In: *Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems* (eds R. E. Masters and K. E. M. Galley), Tall Timbers Research Station, Tallahassee, Florida, USA.

Trollope W. S. W., de Ronde C. & Geldenhuys C. J. (2004) Fire behaviour. In: *Wildland fire management handbook for sub-Saharan Africa* (eds J. G. Goldammer and C. de Ronde). www.compress.co.za, www.oneworldbooks.com, Johannesburg.

Trollope W. S. W. & Trollope L. A. (1996) Fire in African savanna and other grazing ecosystems. In: *Forest Fire and Global Change* p. 15, Shushenkoye, Russian Federation.

Tschakert P., Huber-Sannwald E., Ojima D. S., Raupach M. R. & Schienke E. (2008) Holistic, adaptive management of the terrestrial carbon cycle at local and regional scales. *Global Environmental Change* 18, 128-41.

Turpie J. K., Marais C. & Blignaut J. (2008) The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics* 65, 788 - 98.

Tyler E. (2006) CDM for small, sustainable projects: where is the value added? Ecosystem Marketplace [online]: http://www.ecosystemmarketplace.com - Opinion section, California, USA.

UNDP. (2003) The Clean Development Mechanism: a user's guide. p. 20. United Nations Development Programme, New York.

UNFCCC. (2009) Decision 4/CP.15. Methodological guidance for activities relating to reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries. United Nations: FCCC/CP/2009/11/Add.1 http://unfccc.int/resource/docs/2009/cop15/eng/11a01.pdf#page=11.

UNFCCC (1998) Kyoto Protocol to the United Nations Framework Convention on Climate Change. At: http://unfccc.int/kyoto_protocol/items/2830.php.

van der Werf G. R., Dempewolf J., Trigg S. N., Randerson J. T., Kasibhatla P. S., Giglio L., Murdiyarso D., Peters W., Morton D. C., Collatz G. J., Dolman A. J. & DeFries R. S. (2008) Climate regulation of fire emissions and deforestation in equatorial Asia. *Proceedings of the National Academy of Sciences* 105, 20350-5.

van der Werf G. R., Randerson J. T., Collatz G. J., Giglio L., Kasibhatla P. S., Arellano Jr A. F., Olsen S. C. & Kasischke E. S. (2004) Continental -scale partitioning of fire emissions during 1997 to 2001 El Nino / La Nina Period. *Science* 73, 73-6.

van Jaarsveld A. S., Biggs R., Scholes R. J., Bohensky E., Reyers B., Lynam T., Musvoto C. & Fabricius C. (2005) Measuring conditions and trends in ecosystem services at multiple scales: the Southern African Millennium Ecosystem Assessment (SAfMA) experience. *Philosophical transactions of the Royal Society: B* 360, 425-41.

van Kooten G. C., Eagle A. J., Manley J. & Smolak T. (2004) How costly are carbon offsets? A meta-analysis of carbon forest sinks. *Environmental Science & Policy* 7, 239-51.

van Laake P., Skutsch M. & McCall M. K. (2009) Chapter 3.4: Data Collection at Local / National Level. . In: *GOFC-GOLD 2009. A Sourcebook of Methods and Procedures for Monitoring and Reporting Anthropogenic Greenhouse Gas Emissions and Removals. GOFC-GOLD Report Version COP 15-1,* . GOFC-GOLD Project Office, Natural Resources Canada). , Alberta.

van Langevelde F., van de Vijver C. A. D. M., Kumar L., van de Koppel J., de Ridder N., van Andel J., Skidmore A. K., Hearne J. W., Stroosnijder L., Bond W. J., Prins H. H. T. & Rietkerk M. (2003) Effects of fire and herbivory on the stability of savanna ecosystems. *Ecology* 84, 337-50.

van Wilgen B. W. (1982) Some effects of post-fire age on the above-ground plant biomass of fynbos (macchia) vegetation in South Africa. *Journal of Ecology* 70, 217-25.

Van Wilgen B. W., Higgins K. B. & Bellstedt D. U. (1990) The role of vegetation structure and fuel chemistry in excluding fire from forest patches in the fire-prone fynbos shrublands of South Africa. *Journal of Ecology* 78, 210-22.

van Wilgen B. W. & Scholes R. J. (1997) The vegetation of fire regimes of southern hemisphere Africa. In: *Fire in southern African savannas: ecological and atmospheric perspectives* (eds B. W. van Wilgen, M. O. Andreae, J. G. Goldammer and J. A. Lindesay) p. pp256. Witwatersrand University Press, Johannesburg.

van Wilgen B. W., Trollope W. S. W., Biggs H. C., Potgieter A. L. F. & Brockett B. H. (2003) Fire as a driver of ecosystem variability. In: *The Kruger Experience: Ecology and Management of Savanna Heterogeneity* (eds J. du Toit, H. Biggs and K. Rogers). Island Press, London.

Veldman J. W., Mostacedo B., PeÒa-Claros M. & Putz F. E. (2009) Selective logging and fire as drivers of alien grass invasion in a Bolivian tropical dry forest. *Forest Ecology and Management* 258, 1643-9.

Venter F. J. (1990) A classification of land for management purposes in the Kruger National Park. University of South Africa Johannesburg.

Verplanke J. & Zahabu E. (2009) A Field Guide for Assessing and Monitoring Reduced Forest Degradation and Carbon Sequestration by Local Communities. . University of Twente, K:TGAL programme, www.communitycarbonforestry.org.

Vignola R., Locatelli B., Martinez C. & Imbach P. (2009) Ecosystem-based adaptation to climate change: what role for policy-makers, society and scientists? *Mitigation and adaptation strategies for global change* 14, 691-6.

Visseren-Hamakers I. J. & Glasbergen P. (2007) Partnerships in forest governance. *Global Environmental Change* 17, 408-19.

Walker B. & Meyers J. A. (2004) Thresholds in ecological and social-ecological systems: a developing database. *Ecology and Society* 9, 3. URL: http://www.ecologyandsociety.org/vol9/iss2/art3/.

Walker B. H., Kinzig A. P. & Langridge J. (1999) Plant attribute diversity, resilience, and ecosystem function: the nature and significance of dominant and minor species. *Ecosystems* 2, 95-113.

Walker S. M. & Desanker P. V. (2004) The impact of land use on soil carbon in Miombo Woodlands of Malawi. *Forest ecology and management* 203, 345-60.

Watson R. T., Noble I. R., Bolin B., Ravindranath N. H., Verado D. J. & Dokken D. J. (2000) *Special Report on Land Use, Land-Use Change and Forestry*. Cambridge University Press, Cambridge.

Watson R. T., Zinyowera M. C. & Moss R. H. (1997) The regional impacts of climate change: an assessment of vulnerability: IPCC. p. pp 16, IPCC, Geneva, Switzerland.

Westerling A. L., Hidalgo H. G., Cayan D. R. & Swetnam T. W. (2006) Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science* 313, 940-3.

Williams M., Ryan C. M., Rees R. M., Sambane E., Fernando J. & Grace J. (2008) Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique. *Forest Ecology and Management* 254, 145-55.

Winkler H. (2008) Measurable, reportable and verifiable: the keys to mitigation in the Copenhagen deal. *Climate Policy* 8, 534-47.

Winkler H., Howells M. & Baumert K. (2007) Sustainable development policies and measures: institutional issues and electrical efficiency in South Africa. *Climate Policy* 7, 212-29.

Wise R., von Maltitz G., Scholes R. J., Elphinstone C. & Koen R. (2009) Estimating carbon in savanna ecosystems: rational distribution of effort. *Mitigation and adaptation strategies for global change* 14, 579-604.

Woghiren A. J. (2002) Nitrogen characterisation of the savanna flux site at Skukuza, Kruger National Park. In: *Faculty of Science* p. 142. University of the Witwatersrand, Johannesburg.

World Bank. (2009) BioCarbon Fund Project. Uganda: Nile Basin Reforestation. http://wbcarbonfinance.org/Router.cfm?Page=Projport&ProjID=9644.

WRI. (2008) World Resources Institute, Climate Analysis Indicators Tool (CAIT) Version 5.0, http://cait.wri.org/.

Zahabu E. (2008) Sinks and Sources: a Strategy to Involve Forest Communities in Tanzania in Global Climate Policy. University of Twente, PhD Thesis.

Zahabu E., Skutsch M., Sosvele H. & Malimbwi R. E. (2007) Reduced emissions from deforestation and degradation. *African Journal of Ecology* 45, 451-3.

Zepp R. G., Callaghan T. V. & Erickson D. J. (2003) Interactive effects of ozone depletion and climate change on biogeochemical cycles. *Photochemistry and photobiology* 2, 51-61.

Zheng X. & Eltahir E. A. B. (1998) The role of vegetation in the dynamics of West African monsoons. *Journal of Climate* 11, 2078-97.