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Challenges and opportunities in linking carbon sequestration, livelihoods and ecosystem service provision in drylands

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ABSTRACT

Changes in land use and management practices to store and sequester carbon are becoming integral to global efforts that both address climate change and alleviate poverty. Knowledge and evidence gaps nevertheless abound. This paper analyses the most pressing deficiencies in understanding carbon storage in both soils and above ground biomass and the related social and economic challenges associated with carbon sequestration projects. Focusing on the semi-arid and dry sub-humid systems of sub-Saharan Africa which are inhabited by many of the world's poor, we identify important interdisciplinary opportunities and challenges that need to be addressed, in order for the poor to benefit from carbon storage, through both climate finance streams and the collateral ecosystem service benefits delivered by carbon-friendly land management. We emphasise that multi-stakeholder working across scales from the local to the regional is necessary to ensure that scientific advances can inform policy and practice to deliver carbon, ecosystem service and poverty alleviation benefits.

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1. Introduction

Community-based land management projects within the voluntary carbon sector increasingly apply standards and protocols designed to reduce trade-offs and deliver multiple benefits across carbon storage, poverty alleviation, community empowerment, and biodiversity conservation dimensions (e.g. Plan Vivo and the Climate, Community and Biodiversity Alliance (CCBA) standards). However, accurate accounting methodologies underpinning the carbon components of such assessments are lacking due to an absence of scientific data, models, appropriate local monitoring methods and regional measurement protocols, particularly in drylands, where methods need to address inherent spatial and temporal dynamism (Schmidt et al., 2011; Reynolds et al., 2011). These challenges mean carbon sequestration gains (or prevented losses) are difficult to quantify, and need to be integrated with assessments of livelihood costs, benefits and trade-offs. The lack of coherent and credible science accessible to practitioners remains a significant obstacle to the development of integrated practice. This paper evaluates the current scientific knowledge and outlines the evidence gaps underpinning these challenges, identifying the most pressing deficiencies and promising ways forward. We focus largely on sub-Saharan Africa, a target area for many pro-poor environment-development initiatives and globally important region for carbon storage (Ciais et al., 2009). We specifically consider dryland ecosystems, where many of the world's poorest communities live (Middleton et al., 2011) and where knowledge and investment are lacking compared to tropical forest regions (Terrestrial Carbon Group, 2010). The paper's objectives are to:

(1) Identify the scientific and process-based knowledge gaps and methodological challenges in understanding the factors affecting carbon storage in dryland soils and above ground biomass.
(2) Reflect upon the links between carbon, the provision of other ecosystem services and livelihood impacts, considering the challenges in developing payment systems for carbon storage.
(3) Outline the key forward-looking interdisciplinary and multi-stakeholder opportunities to advance progress towards pro-poor, climate-smart development in the world's drylands.

We analyse a range of literature and practical experiences from across sub-Saharan Africa, drawing on academic, policy and practical insights of a broad multi-stakeholder group who attended a workshop held in Namibia in October 2010. The workshop goal was to evaluate current knowledge on the relationships between livelihoods, poverty, land use and carbon stores and fluxes through discussion and knowledge-sharing. The workshop also developed collaborative partnerships across countries and between researchers, non-governmental organisations (NGOs), private sector investors and government officials, providing further insight to the practical recommendations presented in this paper.

2. Knowledge gaps and methodological challenges in understanding dryland carbon storage

Efforts to increase the size of the terrestrial carbon store are perhaps most commonly associated with climate change mitigation. However, the presence of carbon in both soils and biomass is hugely beneficial for a range of ecosystem functions and services, assisting in the provision of adaptation options and the maintenance of natural resource-based livelihoods. Simply by being present, soil organic carbon (SOC) improves soil structural stability and water holding capacity (Holm et al., 2003). The decomposition of organic carbon generates further direct benefits through the recycling of nutrients and maintenance of soil fertility (Stursova and Sinsabaugh, 2008; Scholes et al., 2009). This, in turn, contributes to other supporting, provisioning and regulating services, particularly food and timber production. Climate change mitigation efforts linked to land use and land management generally seek to increase the amount of carbon stored in soils and biomass. However a trade-off exists in that to realise many livelihood and ecosystem service benefits from SOC requires its depletion (e.g. through crop production) and thus a net release of CO_2 (Janzen, 2006). Understanding how different land use and management systems can both maintain and enhance carbon storage and other ecosystem services (the ''hoard it or use it'' conundrum identified by Janzen (2006)), as well as identifying where the trade-offs between these goals are situated, are key research challenges, especially in relation to how SOC can be increased without suppressing decomposition rates so that nutrient cycling is not adversely affected (Powlson et al., 2011).

Many of the knowledge gaps in understanding dryland carbon storage stem from a lack of empirical data and scientific evidence, which limits the utility of scientific knowledge for research users such as policy makers and NGOs. Measurement challenges restrict the number of studies focusing on processes and trade-offs in drylands, impeding development of accurate carbon accounting methodologies. Incomplete knowledge of carbon cycles makes it difficult to up-scale plot or field-level studies to inform regional or global model development, hindering accurate prediction of how land, non-carbon ecosystem services and livelihoods may be affected by climatic, environmental and other changes. Parallel is the need to draw together understanding from different disciplinary bases to develop applied research, grounded in sound science, to deliver policy-relevant outcomes of practical value.

In this section we outline the key data gaps and research needs in relation to these challenges for below ground soil organic carbon (SOC) stores and fluxes (Section 2.1); and above ground biomass (AGB) stores and associated fluxes (Section 2.2).

2.1. *Below ground carbon: soil organic carbon (SOC) stores and fluxes*

The need to include SOC storage in payment schemes is long recognised (Lal, 2004), but only simple models are used at present, based on changes in soil organic matter (SOM) measurements through time (e.g. Wildlife Works Carbon, 2010). A greater range and depth of field data are essential to enable monitoring of changes in SOC storage (Powlson et al., 2011) and the development of a new generation of soil carbon models (Schmidt et al., 2011). These need to be linked to the development of methodologies that local communities can use to monitor SOC. In this section we identify sampling and measurement challenges and outline preliminary monitoring opportunities that offer scope to significantly advance understanding of SOC processes and fluxes. This acts as a guide to developing SOC budgets that can be linked to different climate, land use and land management futures, and requires the integration of insights from soil science, microbiology and environmental modelling.

The size of terrestrial OC stores is determined by the balance between inputs from primary production and outputs principally from gaseous losses to the atmosphere due to SOM decomposition and abiotic photo-oxidation of litter. Further losses associated with the erosion of surface sediments or litter are minimal. Typically, erosion through aeolian or fluvial processes represents a local redistribution of sediment, leading to accumulations in depositional areas, e.g. around bushes (Schlesinger et al., 1990; Dougill and Thomas, 2004). As such, it is the changes in primary productivity and/or decomposition rates that primarily affect the amount of OC stored in soils (Schmidt et al., 2011). SOC depletion occurs as it is mineralised and respired as CO_2 by heterotrophic soil microbes metabolising carbon substrates (Luo and Zhou, 2006). Changes in land management practices (e.g. reduced tilling, reduced grazing and prevention of deforestation) can reduce heterotrophic respiration losses, preserving the SOC store (e.g. Cao et al., 2004). However, scientific evidence gaps limit our ability to include SOC stores and fluxes in the valuation of benefits accruing from land management practices and reduce the accuracy of future predictions of SOC store changes under different land management and climatic scenarios. This makes it difficult to assure investors that the anticipated carbon sequestration will be delivered (Versi, 2009).

Three factors underpin this uncertainty:

(1) Insufficient data on the amount, distribution and form of SOC.
(2) Few empirical data from drylands that can be used to calibrate and validate soil respiration models and predict the effects of climate and land use on SOC losses through respiration.
(3) Limited awareness of the unique factors and processes affecting SOC in drylands.

Each of these is considered below, highlighting areas of research innovation that provide significant opportunity to advance scientific understanding.

2.1.1. *Insufficient data on SOC amount, distribution and form*

Across dryland sub-Saharan Africa, reliable SOC data are lacking. Despite mapping and quantification of regional-scale SOC, data remain at a coarse resolution. The Food and Agriculture Organisation's global terrestrial carbon map amalgamates data from the harmonized world soil database (ISRIC, 2009) with above and below ground biomass to show the distribution of vegetation and SOC to 1 m depth at a 1 km^2 resolution (Scharlemann et al., 2009). However, variability in SOC concentrations, even within farms and fields, is high, particularly where organic manures are applied preferentially to soils closest to homesteads (e.g. Giller et al., 2009). There are also significant variations in soil texture and associated SOC linked to topographic variability (tens and hundreds of metres). A pre-requisite to reliable carbon accounting and assessment of links between SOC and other ecosystem services are accurate data on SOC stores at these finer scales.

While soil property databases provide spatial SOC information, sampling protocols typically take composite samples from 0–30 cm and 30–100 cm (Walsh and Vågen, 2006). This facilitates efficient and cost-effective characterisation of mesic soils from landscapes with clear differentiation in organic content at the A/B horizon interface. However, in drylands there is little horizonisation and SOC concentrates close to the surface, often within a surface biological crust (Belnap and Lange, 2003), so alternative sampling is required to deliver accurate SOC measurements. The importance of more detailed depth profile measurements is stressed by Powlson et al. (2011), especially in relation to the impacts of changes in agricultural practices such as moves towards zero tillage. Some studies have shown that increases in surface layer storage may be partly offset by decreases in SOC storage below the depth of tillage (e.g. Machado et al., 2003) and the implications of such findings for carbon payment initiatives are now being questioned (e.g. Gattinger et al., 2011).

A further pre-requisite is accurate data on the nature and composition of SOC stores, and on SOC decomposition processes across different soil moisture and temperature regimes. Information on the composition of SOC would allow targeted investment of climate finance, as different OC forms have contrasting residence times and susceptibilities to losses (Trumbore, 2000). The composition of SOC is important in affecting degradation rates but is poorly studied in drylands, where understanding of the relationship between composition and susceptibility to decomposition gained from mesic soils does not apply (Austin, 2011). Some organic carbon molecules are rapidly decomposed and highly transient in soils with residence times of days to weeks (Mager and Thomas, 2011). Others, such as lignin, are extremely resistant to decomposition and can reside in soil for hundreds to thousands of years. Lignin is a major constituent of all woody material and an inhibitor of biotic decomposition in mesic soils, but in drylands has the opposite effect as it aids light absorption, stimulating photochemical reactions and organic mass loss (Austin and Ballare, 2010). Another example of carbon in a form with considerable longevity in soils is that in biochar, a highly porous charcoal. Biochar is increasingly used as a soil enhancer due to associated improvements in water holding capacity and nutrient retention, and because its carbon is resistant to mineralization, significantly increasing the stable fraction of the soil carbon store (Sohi et al., 2010; Lehmann et al., 2006). Some evidence suggests that biochar additions can also aid the retention of other forms of SOC (Liang et al., 2010), but more research into the short to medium term effects of biochar on soil properties is needed (see Powlson et al., 2011).

Insufficient data on the form, distribution and processes affecting SOC represents an important barrier to more holistic assessment of the impacts of shifts towards land uses, management strategies and the effects of community projects aiming to enhance carbon storage. Inclusion of fine-resolution SOC data collection within protocols used in major regional and global soil database development is essential.

2.1.2. *Limited empirical data to test soil respiration models*

Determining whether investments in soil carbon storage are sustainable and can contribute to climate change mitigation, adaptation and poverty alleviation over the long term, requires models to predict the effects of climate and land use changes on the processes controlling SOC losses. Data on respiration losses is needed to feed into flux models to allow prediction of annual losses under given land use, soil types and climates. The relatively few studies of soil respiration from dryland soils (e.g. Sponseller, 2007; Liu et al., 2009; Sheng et al., 2010) provide an incomplete understanding of dryland carbon cycling (Scholes et al., 2009). The latest most comprehensive global reviews of soil CO_2 efflux data by Bond-Lamberty and Thomson (2010a, b) underscore this lack of data. Only *c.* 5% of 1562 field-based studies of soil respiration on un-modified plots included in their review come from drylands. Measurements in different climatic regimes are vital as scientific consensus is lacking on the relationship between respiration losses and climate. The relationship between respiration and moisture/temperature is rarely linear (Davidson et al., 2006). Large pulses of CO_2 efflux typically occur following precipitation after prolonged dry periods (Liu et al., 2002). A high proportion of annual CO_2 losses from dryland soils occur during these re-wetting pulses (Borken and Matzner, 2009). Although the magnitude and duration of carbon-loss pulses are critical to the longer-term soil carbon balance, few field data exist upon which to base annual carbon loss estimates, including on the role of plant root turnover and exudates, soil microbial content, temperature and moisture in affecting these losses.

Although Bond-Lamberty and Thomson (2010a,b) show climatic warming is increasing the global flux of CO_2 from soils to the atmosphere, most meta-analyses do not distinguish between CO_2 from microbial decomposition of SOC and that from plant roots. It is thus impossible to determine if any increase in soil CO_2 efflux is due to accelerated SOC decomposition (and therefore represents a decline in SOC stores) or greater primary productivity (with no associated decline in SOC). It is challenging to separate the two sources in the field (for methods by which this can be achieved, see Kuzyakov, 2006). Consequently, there are few in situ data, particularly in drylands. Reliable assessment of processes affecting CO_2 efflux rates requires in situ chamber monitoring systems to collect gases from remote field locations (see Luo and Zhou, 2006) which not only quantify efflux rates but the source of C contributing to the efflux. Studies in the Kalahari (Thomas et al., 2011) show the potential for establishing reliable monitoring methods to assess soil CO_2 efflux, providing high temporal resolution CO_2 efflux data from remote sites, but with limited replications and constrained monitoring periods. Extension, both spatially and temporally, of in situ CO_2 efflux measurements is essential for improved data on soil respiration required for modelling of carbon budgets (Maestre and Cortina, 2003). Such new data could be used to test carbon flux estimates of models such as the Joint UK Land Environment Simulator (JULES), and the Soil Plant Atmosphere (SPA) model (Williams et al., 1996) which has been applied successfully in modelling C fluxes in Australian drylands (Zeppel et al., 2008; Whitley et al., 2011). New data could also enhance dynamic models such as GEFSOC, developed globally for national and sub-national assessments of SOC stocks and dynamics (Milne et al., 2007), results from which could usefully inform the development of policy options and scenarios.

2.1.3. *Limited awareness of the unique factors and processes affecting dryland SOC*

Processes affecting dryland SOC stores have some fundamental differences to those in mesic ecosystems. First, despite low precipitation and microbial activity, rates of above ground litter decomposition in drylands remain high. Austin and Vivanco's (2006) experiments showed that intercepted solar radiation was the only factor with a significant effect on decomposition of organic matter in a semi-arid Patagonian steppe. Estimates of carbon loss due to such photodegradation in drylands could be substantial when up-scaled, with annual estimates ranging from 1 to 4 g/m^2 to 16 g/m^2 (Brandt et al., 2009; Rutledge et al., 2010). This suggests there is a "short-circuit" in the dryland carbon cycle as carbon fixed in above ground biomass is lost directly to the atmosphere without the need for microbial decomposition. This is important because it disconnects SOC turnover from biotic factors such as water availability and microbial activity. Quantification of these processes may help resolve discrepancies in traditional models of biotic controls on decomposition. Thus, future changes in cloudiness, ozone depletion, fire incidence and vegetation type and cover are likely to have more significant effects on the dryland carbon balance than temperature or precipitation changes (Austin, 2011). This concept represents a significant shift from the long-standing paradigm of water-limitations and precipitation pulses controlling dryland biogeochemical cycles (Noy-Meir, 1973). Second, fire can remove significant components of above ground litter, and even smoulder through coarse roots deep into the soil, affecting both the amount and type of organic inputs to SOC. Third, deep rooting is common in dryland trees. It is poorly quantified but leads to C inputs at depth in soils. Deep rooting is also connected to the Birch effect (Jarvis et al., 2007), which can result in plant roots lifting deep soil water to surface soils during the night, leading to pulses of decomposition in surface soils as they moisten.

Information on the amount, distribution and species composition of microbes in soils is critical to respiration and the fate of SOC, yet empirical information is lacking on how enzymes are affected by disturbance and climatic changes. Fungi and bacteria largely control SOC respiration processes, influencing the residence time of SOC storage. Widespread occurrence of fungi in dryland soils may further explain the poor correlation between biotic factors and decomposition. Fungi have higher tolerance to desiccation than bacteria so are more likely to survive periods between rainfall events, facilitating microbial activity despite very low water availability (Austin, 2011). Linked to this is the persistence of carbon degrading enzymes, particularly phenol-oxidases, which provide an advantage for rapid organic turnover despite long periods of unfavourable conditions (Stursova and Sinsabaugh, 2008). High enzyme activity, coupled with warm and well aerated conditions generally favours rapid SOC turnover and limits SOC retention in drylands (Mills and Cowling, 2010). Furthermore, the ability of dryland soils to sequester more carbon is constrained by limitations in other nutrients, particularly N and P (van Groenigen et al., 2006). New microbiological methods such as next-generation 454 pyrosequencing can rapidly identify soil bacterial and fungal communities that underpin plant and soil productivity (Acosta-Martinez et al., 2008), though these data are yet to be collected for sub-Saharan African soils. New understanding gained through these methods will move towards locating spatial and temporal thresholds at which carbon storage capability declines, or significant respiration losses are instigated. For these advances that are essential for the quantification of the longevity of soil carbon storage to occur requires soil science to be linked with microbiological analyses.

2.2. Above ground biomass (AGB) stores and fluxes

Above ground biomass (AGB) stores are determined by the balance between carbon accumulation from primary production and carbon losses related to mortality, fire, human use and land use change. AGB influences settlement patterns across a landscape, as well as playing a vital role in rural livelihood activities such as livestock grazing, timber harvesting, fuelwood and charcoal production. It is therefore crucial to understand drivers of AGB, projected future trends, and their implications, in order to develop policy that protects resources and promotes livelihood options. Significant knowledge gaps nevertheless remain, including:

(1) Limited observational data on the spatial distribution and temporal variability of AGB.
(2) Poor understanding of the natural and human drivers of AGB and the links to changes in other ecosystem services.

Addressing these gaps requires integration of forestry expertise with ecological and remote sensing techniques alongside livelihoods and resource use assessments grounded in the social sciences.

2.2.1. Lack of observational data for present day AGB storage

AGB varies considerably across drylands at a range of scales, complicating mapping and monitoring. AGB can be estimated from plot studies, where biomass is related to standard forestry observations such as tree diameter. In this way, change in AGB storage can be monitored through resampling of permanent vegetation plots. Such efforts are necessary yet labour intensive, restricting the achievable spatial and temporal coverage. Recently, remote-sensing studies have been used to estimate biomass storage and these offer potential for developing regional AGB estimates.

Modelling suggests that dryland and sub-humid areas contain most of Africa's AGB, due to their extensive coverage (Ciais et al., 2009). However, it is difficult to determine the accuracy of these estimates due to limited observational data. Published plot monitoring reports of AGB and SOC in natural and human-modified landscapes of sub-Saharan Africa focus largely on miombo woodland systems in South Africa, Mozambique and Tanzania; many other ecosystem types remain largely unsampled. Even where studies have occurred, accessing data is difficult, or data are old. Ecosystem-specific equations relating AGB to standard tree measurements for many systems are also lacking. Permanent monitoring plots need to be established particularly in drier savanna woodland and grasslands as well as covering a broader range of miombo woodlands (Ryan et al., 2011). These would generate a standardised database of AGB, tree growth, plant–soil relations and effects of human impacts through annual resurveys.

Earth observation (EO) studies offer considerable scope for extending monitoring and understanding of AGB on national and regional scales (Mitchard et al., 2009) and first need to be calibrated against in situ observations. Baccini et al. (2008) used optical data from the moderate resolution imaging spectroradiometer (MODIS) on the Terra and Aqua satellites, trained and tested against plot-based biomass data (from locations 2°N to 6°N) to predict above ground biomass at 1 km resolution over tropical Africa. Validity of this calibration at other latitudes remains to be fully tested. New biomass products are being generated (Saatchi et al., 2011), and need to be inter-compared to determine areas of agreement and confusion.

Radar remote sensing offers new possibilities for monitoring forest biomass with significant advantages over optical methods (Le Toan et al., 2011) as radar backscatter from plant structure can be calibrated against field plots and distinguish effectively between forests across a range of biomass values (Ryan et al., 2012). A new EO approach to AGB monitoring in rangeland systems uses MODIS leaf area index and fraction of photosynthetically active radiation (f_{PAR}) products to make estimates of water use efficiency (WUE) and annual net primary productivity (Palmer et al., 2010). WUE defines how efficiently the individual plant or landscape uses precipitation to produce biomass and has been used to define rangeland functionality (Holm et al., 2003). Such approaches need to be carefully verified as if links can be proven, a new route to monitoring landscape scale changes in carbon storage will be available (Richmond et al., 2007).

2.2.2. Lack of quantitative assessments of natural and human drivers of AGB storage

Global change affects AGB storage largely through shifts in precipitation – a major uncertainty in climate projections – and through poorly understood responses to rising atmospheric CO_2 concentrations. Global modelling studies suggest Africa provides a carbon sink (excluding land-use change) of 0.28 PgC year^{-1} for the period 2000–2005 with the majority of the sink simulated to occur in savanna soils (Ciais et al., 2009). Verifying these estimates requires a comprehensive plot network measuring both AGB and SOC, resampling at regular intervals, as AGB and SOC stocks are not well correlated for savanna systems (Ryan et al., 2011).

Fire is a dominant feature of Africa's dryland and sub-humid landscapes burning 256 million hectares of land annually (1997–2008 mean, Giglio et al., 2010) and resulting in large losses of C (0.72–0.86 PgC a^{-1}, Lehsten et al., 2009; Roberts et al., 2009). In miombo woodlands fire controls AGB through complex feedbacks between production, tree-grass competition, fuel load, fire intensity and stem mortality (Ryan and Williams, 2011). Reduced fire prevalence may allow closed canopy woodlands to expand into savanna regions (Bond and Keeley, 2005), resulting in a substantially enhanced carbon sink. However, the drivers of African fire remain poorly attributed, especially their links to the surrounding socio-economic context and livelihood strategies. Consequently it is difficult to predict future changes in fire frequency and extent (Archibald et al., 2009). Lehsten et al. (2010) suggest that declining precipitation between 1980 and 2060 will result in a 20–25% reduction in the area burned by wildfire across Africa. However, the combined effect of reduced precipitation and reduced fire on ecosystem carbon balance is unknown. Whilst improved fire prevention could lead to a substantial carbon sink (Grace et al., 2006), regional analysis across southern Africa suggests that humans may already be suppressing the fire regime (Archibald et al., 2010).

Humans modify the natural occurrence of fire through the ignition and suppression of fires. Fire models need to incorporate both these aspects as well as climatic controls to be able to accurately simulate fire and to enable the models to predict the prevalence of fire under future climate and under different land management regimes. Separating human and climatic drivers requires sub-sampling of regions of similar climatic influence but different human impacts, for example across protected area or national/regional boundaries, and further requires links to specific model classes. Process models of biomass–disturbance interactions (fire, grazing), resolving age structure, are most suitable for identifying sensitivity to stochastic impacts. Integrated biogeochemical–biophysical models are better able to resolve climate controls on biomass distributions over larger areas. Links to participatory monitoring approaches to gain local and indigenous knowledge can also feed into dynamic systems models (e.g. Dougill et al., 2010). To advance such integrated models, sampling protocols for future studies must explicitly explore the role of human drivers including fire suppression, local policies and regulations, community based fire management, land tenure and management practices. Only with such extensions will it be possible to predict fire extent and intensity against annual fire monitoring programmes. These predictions are essential if fire management is to be included in carbon budget analysis and linked to monitoring and payment schemes. Such advances will be significant for carbon investors because determination of fire risk will affect decisions on where to invest and enable likely losses of biomass due to fire to be insured against.

Additional direct human impacts are important determinants of AGB stores; increased demand for charcoal leads to forest degradation (Ahrends et al., 2010), smallholder agriculture contributes to deforestation (Syampungani et al., 2011), whereas farm abandonment allows AGB accumulation (Williams et al., 2008). Clearance of miombo woodlands for agriculture reduces both AGB and SOC, resulting in a release of up to 30 tC ha^{-1} and after cessation of agriculture, AGB recovers at *c.* 0.7 tC ha^{-1} $year^{-1}$ reaching pre-disturbance levels after 20–30 years, whereas SOC shows no significant changes over these timescales (Williams et al., 2008). Further assessments across agro-ecological settings are essential to widen the significance of these plot-based case studies, enabling development of national and regional-scale analyses.

Across Africa, land-use change is estimated to emit 0.13–0.33 PgC $year^{-1}$ (Houghton and Hackler, 2006; Ciais et al., 2009; Bombelli et al., 2009; Canadell et al., 2009) equivalent of up to 23% of global land-use emissions. Uncertainties remain substantial and relate to deforestation and degradation rates, biomass storage and poor treatment of the impacts of logging, livestock grazing, fires and shifting cultivation which are difficult to identify and quantify by remote sensing. National surveys suggest forest area declined at rates of 1% per annum in East Africa and 0.5% per annum in southern Africa over the period 2000–2010 while savanna woodland declined by 0.5% per annum (FAO, 2011). However, national survey data is sparse and land-cover definitions are particularly problematic for savanna woodland systems.

AGB is vital in meeting domestic requirements for energy, with fuelwood collection in Africa exceeding 600 million m^3 $year^{-1}$ (FAO, 2011), equivalent to 0.18 PgC $year^{-1}$ (assuming a wood density of 0.58 Mg m^{-3} and carbon content of 0.5) or about 2% of NPP from the African savanna biome. The role of fuelwood collection in determining regional forest quality and AGB storage is uncertain, although unsustainable extraction in peri-urban locations has been documented (FAO, 2009). Haberl et al. (2007) estimated that human appropriation of NPP (excluding human-induced fires) for sub-Saharan Africa was 18%, though this is likely to grow as the population rises.

Accounting methodologies under the UNFCCC's Clean Development Mechanism (CDM) recognise the issues and uncertainties specific to forestry and carbon storage in trees and other AGB. Tradeoffs between timber harvesting and carbon storage and the temporary residence of carbon in AGB are acknowledged (Rueff and Schwartz, 2012) and incorporated into carbon accounting methodologies facilitated by open access models that simulate long-term (30-year) forestry mitigation projects (e.g. Schelhass et al., 2004; Tuomi et al., 2008). Further research addressing gaps in understanding relating to above- and below-ground carbon storage is vital to improve the accuracy and representation of key processes within models and accounting methodologies.

3. Linking scientific evidence gaps and ecosystem service valuation challenges

Carbon store and flux dynamics are physical changes to an ecosystem's structures and processes, resulting in changes in the bundle of services flowing from an ecosystem and the benefits that humans derive from interactions with that ecosystem (Daily, 1997). Ecosystem services associated with carbon are numerous (e.g. Mtambanengwe and Mapfumo, 2005) although the exact nature of relations are poorly quantified in drylands and need further testing. Such knowledge is vital if payments for carbon sequestration are to capture all potential impacts that changing land management practices can have on the bundle of ecosystem services drawn on by the rural poor in pursuit of their livelihoods. Ecosystem services are often interdependent, so optimization of a single service may have unforeseen impacts on other ecosystem services (Abson and Termansen, 2010). For example, optimisation of climate regulation through payments for carbon sequestration may affect food provision and water regulation, at worst, limiting successful adaptation to climate change. Broader ecosystem service impacts of carbon sequestration schemes therefore require careful consideration. We identify the key challenges as a:

(1) Need to better understand the relationships between carbon, ecosystem service provision, and drivers of future change;
(2) Lack of nuanced understanding of the links between poverty and land tenure and the implications this has for the design and implementation of carbon payment schemes.
(3) Shortage of appropriate decision support tools in informing land management decisions and adaptation strategies, alongside the thresholds at which land users will shift towards carbon mitigation scenarios, particularly in rangelands.

Without such understanding it is difficult to quantify the potential carbon sequestration that could be achieved under different land management strategies or the implications for poverty and ecosystem service provision.

3.1. *Understanding the relationships between carbon, ecosystem service provision and drivers of future change*

Identifying the complete bundle of ecosystem services associated with increased carbon storage and related synergies and trade-offs is vital when considering the multi-faceted nature of livelihoods and the pressures and changes to which the poor adapt (Stringer et al., 2009). Climate change affects community, state governance and service delivery across multiple sectors, so mitigation and adaptation strategies need to reflect this complexity and scope if ecosystem services are to be sustained whilst carbon is stored. This requires inter- and multi-disciplinary approaches to capture aspects that straddle traditional academic disciplinary boundaries. It is also imperative that indigenous knowledge and traditional land management approaches are integrated into the science—policy dialogue if solutions are to be both durable and acceptable in local social and cultural contexts (Stringer et al., 2007; Thomas et al., 2012). Similarly, efforts to build capacity to address climate change at larger scales require investments to be 'future-proof' (resilient in the face of multiple development challenges that extend into the future). These challenges include, for example, food security, population growth and rural–urban and transboundary migration. Modelling these processes and their dynamic interactions to assess impacts on carbon storage and ecosystem services introduces numerous uncertainties, adding to those associated with climate change model projections (IPCC, 2007).

Given the limitations of predictive models in such dynamic and complex social-ecological systems, scenario techniques offer a window into different possible futures (MA, 2005), allowing currently unseen conditions to be incorporated into planning processes operating across multiple dimensions and scales (Kok et al., 2007). This can guide investments in carbon storage projects and associated land use and land management practices towards being future-proof. Numerous global-, regional- and national-scale databases exist, considering different ecosystem services, vulnerability assessments and climate change scenarios, yet only preliminary integrated analysis of this information has been undertaken (e.g. Davies et al., 2010; Ericksen et al., 2011). Sites where time can be substituted for space can provide evidence for the opportunities and threats faced by the poor in future, as well as the changing capabilities of different land cover types to store carbon and provide other ecosystem services. Analogue approaches enable links to direct farmer-to-farmer programmes that raise awareness of likely adaptation strategies that are feasible in areas of warmer and drier climates (as analogues for predicted climate futures) and can feed into vulnerability assessments, identifying those with high potential to become poor or whose ecosystem services are likely to degrade in future.

Regional databases such as AfSIS (the African Soils Information System), together with IGBP regional programmes (e.g. SAFARI programme, Swap et al., 2004) and EO approaches, could significantly enhance case study understanding of the links between carbon stores and ecosystem services, helping to improve robustness in models of future change. Primary sampling sites of these programmes offer varied agro-ecosystem and climatic settings in which to develop understanding of carbon-ecosystem service relations. However, they need to be complemented with fine-scale scientific consolidation of the biophysical pathways and relationships linking carbon with other ecosystem services and processes (such as nutrient cycling, water holding capacity, soil erodibility and fire). Quantitative testing of these relationships is required because optimal amounts of soil organic carbon are site-specific and depend on local biophysical and socio-economic contexts (Giller et al., 2009). A fundamental awareness of local taboos and norms is also required, ensuring interventions are culturally acceptable and in the best interests of land users (Ifejika Speranza, 2006), because the livelihood priorities of potential carbon service providers (agro-pastoral and pastoral actors and communities) may not correspond with the logic of earning payments from carbon-sequestration, but instead, show a better match with goals to increase and maintain the land's productivity. This logic needs to be linked with the priorities of other actors (governments, project developers) if carbon sequestration projects are to take hold in sub-Saharan Africa (Henry et al., 2011).

Although small-scale payment schemes are becoming more widespread, low carbon prices (<US$10/tonne as of October 2011), weak legal regimes following UNFCCC COP17, and high transaction costs indicate that prospects for carbon payments being able to lift populations out of poverty are currently low (De Pinto et al., 2010). Nevertheless, the carbon market fluctuates greatly and prices and demand could rise if more stringent targets are set. Lipper et al. (2010) suggest that a minimum carbon price of >US$100/tonne is required in Burkina Faso's drylands in order to sufficiently compensate herders' opportunity costs of renouncing cropping. Generally, involvement of local communities in projects explicitly incorporating benefit sharing appears to support the success of carbon forestry projects, even when such activities increase costs (Reynolds, 2012). However, more research is needed on the costs and benefits of mitigation activities for households resulting from land use changes across sub-Saharan Africa. Such assessments provide an important underpinning to decisions on where to target development interventions using payment for ecosystem service approaches. Indeed, the associated costs of managing land to enhance carbon storage may not always be worth any gains in ecosystem services it provides. Costs in this context span a range of different capital assets and include changes to traditional working patterns or additional labour, weed control requirements or fertiliser/manure applications (Giller et al., 2009), in addition to costs associated with monitoring, recording and verification of carbon storage. Cost-benefit analyses that consider trade-offs and synergies across carbon and ecosystem service dimensions as well as across different cultural logics represent vital assessment tools in further advancing understanding of trade-offs.

While scientific and process-based evidence for carbon-ecosystem service relationships is lagging, changes to land management practices to deliver carbon sequestration and other ecosystem services benefits are already being implemented by other stakeholders. The World Bank has provided support for initiatives and projects encompassing climate-smart agriculture and efforts are in place to blend public, private, development and climate finance streams to support carbon sequestration linked to land management. This includes support to soil carbon projects such as the Kenyan Agricultural Carbon Project, funded through the World Bank BioCarbon fund together with a Swedish NGO (Tennigkeit, 2010). Several community projects have adopted agro-forestry approaches and 'evergreen agriculture', using low impact integration of trees and forest conservation with agricultural production (Garrity et al., 2010). Economic benefits of such initiatives are valued through both annual carbon payments and increased annual revenues from yield improvements (Tennigkeit et al., 2009). Through voluntary carbon standards such as those in the Plan Vivo Foundation system (amongst others), participatory processes are used to select suitable trees/shrubs, with decisions on locally-suitable land management systems being co-developed with the communities involved in the project, paying particular attention to gender and wealth differences. Further research assessing the impacts of these schemes as they spread across sub-Saharan Africa will be essential.

3.2. *Poverty, institutions and land tenure: implications for carbon payment schemes*

Delivering carbon payment and ecosystem service benefits to the poorest groups in society first requires identification of who is poor and where they are located. Large-scale datasets permit comparability across different areas and can target climate finance as a poverty alleviation mechanism using analyses of current and future climate risk and vulnerability mapping. However, for the poor to benefit, requires a context-specific understanding of what poverty is and how it is managed. Existing datasets use multiple indicators to determine what poverty is and who is poor (e.g. Thornton et al., 2002), reflecting the multi-dimensional nature of poverty, taking into account lack of choice or capability, as well as material living standards and an inability to meet basic needs. However, those living in poverty have their own ideas about what it means to be poor, based on what is socially and culturally important to them. Participatory well-being assessments can identify hotspots of poverty (e.g. White and Pettit, 2004), providing nuanced understanding of poverty-environment links, yet, developing generalisations from these specific studies remains challenging, particularly when dealing with common property regimes.

Control over land shapes land use and the willingness of land users to incur costs in implementing land management practices (Place, 2009). In much of Africa, the poor own very small plots while communal tenure arrangements may limit access, use and benefit-sharing (Mwangi and Dohrn, 2008; Larson, 2011). Diverse land tenure systems in Africa, characterised by customary and statutory land rights, legal pluralism, land claims through tree planting and the misconception of ''abandoned'' land (Unruh, 2008), means addressing these challenges is difficult. Roncoli et al. (2007, p. 101) highlight that in systems with a mixture of open access and common property regimes (e.g. north-central Malian rangelands) the multifunctional, fragmented and dynamic characteristics of land use by pastoralists and farmers requires a holistic approach that besides carbon, also integrates crop and livestock production. Bennett et al. (2010) highlight a general inability to define and enforce rights to particular grazing resources and inadequate local institutions responsible for management in open access regime of community rangelands in Eastern Cape Province, South Africa. Carbon sequestration projects can thus pose a collective action problem (cf. Ostrom, 1990). Strong but flexible local institutions embedded in multi-level governance structures are key to addressing legal ambiguity, social tensions, social inequalities and overlapping resource-use rights (Bennett et al., 2010; Skutsch and Ba, 2010). The prior existence of active local organisations and ensured participation may serve as criteria for establishing community-oriented carbon sequestration projects (Landell-Mills and Porras, 2002). However, where institutions do not exist or are weak, institutional capacity building might be necessary to address these problems (Roncoli et al., 2007; Stringer et al., 2012). Plan Vivo accredited projects offer important lessons here, recognising that there are usually local-level institutions in place that can appropriately manage the distribution of benefits (Palmer and Silber, 2012). The challenge for researchers is to understand these local institutions, while practitioners need to ensure carbon payment benefits are shared fairly, especially along gender, age, wealth and ethnic lines. Various payment system structures are demonstrated by current Plan Vivo projects. Direct cash payments may be delivered by contracts signed with individuals, based on land ownership and actions to increase carbon storage. Alternatively, 'community' carbon projects consider the community has rights over a delineated area from which it can derive carbon benefits, so payments go towards civic projects (e.g. for improved water sources or housing), livelihood projects (e.g. agro-forestry systems) and social benefit funds (Solly, 2010). Some payment mechanisms can thus deliver broader social co-benefits through improved community governance systems, capacity building, and the creation of local community development plans. These approaches permit the community to identify who is poor and vulnerable. In other approaches, for example in Tanzania, local farmers receive Tsh 20 (US$0.02) per tree per year for a period of 20 years for carbon sequestration (Scurrah-Ehrhart, 2006), yet fruit, timber, firewood and non-timber forest products provide significant and more immediate co-benefits and can help to encourage local involvement in such schemes. Reynolds (2012) suggests that projects emphasising multiple environmental and social goals (e.g. biodiversity conservation, reduced erosion, improved food security, employment opportunities, etc.) are much more likely to succeed than those specialising in carbon sequestration alone.

Such considerations play an important role in determining the economic viability of projects, as well as the willingness with which land users participate. Transaction costs can be reduced if smallholders organise themselves into larger groups, or, in pastoral areas, if avoided emissions that prevent land degradation are taken into account in payment systems (Lipper et al., 2010). Involving buyers directly in community-based mitigation projects can reduce intermediaries and increase ownership by both buyers and carbon credit suppliers, hence increasing revenues. Community-based projects that seek to enhance livelihoods more widely, beyond carbon payments, nevertheless reduce the risk of leakage, as well as reducing related management costs (De Pinto et al., 2010).

3.3. *Tools to support economically viable land management decisions*

Participatory monitoring protocols and standards linked to carbon and climate finance are still at a nascent stage (Dangerfield et al., 2010). Most progress has been made in forest areas, where methodologies and support tools for land management decisions have benefited from international policy focus on forests, largely through REDD+. However, further research is needed to reduce monitoring, reporting and verification costs and promote simplified monitoring technology, especially for monitoring soil carbon. Despite lessons from a growing body of forestry projects, a critical knowledge gap remains for rangelands.

Thornton and Herrero (2010) suggest that pasture management can mitigate 691 kg C ha^{-1} y^{-1} up to 2030 in sub-Saharan Africa (3.6 times more than in Central South America because of the higher level of rangeland degradation in sub-Saharan Africa) (Neely et al., 2009), demonstrating the importance of incorporating rangelands into carbon finance and assessment methodologies. Even with currently low income prospects due to weak carbon prices a slight addition to the income of herders living under poverty line could make a substantial difference (Thornton and Herrero, 2010). Transient use of rangelands by mobile pastoralists and communal property rights nevertheless mean rangeland is often used by large numbers of people (Failey and Dilling, 2010), further reducing direct or indirect rewards per user from particular strategies. Transaction costs for small-scale projects remain high, hampering large-scale involvement of the poorest groups in moving towards a carbon mitigation scenario (Locatelli and Pedroni, 2006). Communities need extension, financial and organizational support to minimize costs and maximize payments and other collateral ecosystem service benefits that can be gained through managing land for carbon. Crediting of mitigation projects and benefits often occurs over long periods, so decisions to adopt strategies that aid carbon sequestration are difficult to operationalise. Higher income from short-term management decisions (e.g. higher livestock stocking levels, cash cropping on steep slopes) can appear more attractive, even though long-term returns are lower, particularly if land degrades. Smallholders and pastoralists may nevertheless consider adopting a carbon management scheme if payments can adequately compensate for renouncing these short-term gains, or by being aware that collateral ecosystem service benefits delivered by carbon-friendly land management can diversify adaptation options and enhance other income streams. One example of a successful payment for ecosystem service model in African rangelands is found in the Subtropical Thicket of the Eastern Cape, South Africa (Mills et al., 2010). This straddles commercial livestock ranches (freehold tenure), state-owned conservation land (national parks) and communal land (leasehold tenure). The project is underpinned by evidence of the SOC and AGB gains achieved by growing *Portulacaria afra* (Mills and Cowling, 2010), a succulent shrub that does not burn, propagates easily, is palatable to domestic livestock and wildlife, and is drought tolerant. Some aspects of this project have received VCS and CCBA accreditation. When further analysed, the successes and challenges experienced in this case will greatly improve understanding and practices in payments for ecosystem services in rangelands and other complex tenure systems.

Decision-support tools (DSTs) represent an important route forward. DSTs can highlight trade-offs and synergies between carbon payments and other core livelihood strategies and may be of particular benefit in semi-arid rangeland areas. Such tools need to consider the links between carbon payments, carbon storage in soils and vegetation (taking into account the protocol limitations and scientific knowledge gaps we have identified) and the wider costs and benefits that can affect livelihoods within the timeframe of a typical (30-year) mitigation project. They may demonstrate that even if there is a slight immediate decline in income when adopting a carbon-friendly form of land use (e.g. destocking), long-term effects show higher gains across both financial and ecosystem service dimensions, while the land use is brought back to sustainable levels. One such tool could model annual changes in carbon stored in AGB and SOC and livestock related emissions as a result of a certain management decision at the paddock or village level (as per the CO2FIX decision tool for aforestation/reforestation (Schelhass et al., 2004; Masera et al., 2003)). Central to this is the need for visual representation to demonstrate the long-term benefits from given management strategies towards carbon sequestration. Further development of tools that currently focus largely on cattle condition and ecological indicators (e.g. Kruger and Katjivikua, 2010; Reed and Dougill, 2010), can help to identify thresholds for decision-change, by explicitly outlining pro-poor benefits and incentives associated with moves towards mitigation scenarios.

4. Conclusion: key steps towards climate-smart pro-poor investments in carbon sequestration

This paper has outlined key scientific and process-based knowledge gaps and methodological challenges in understanding carbon storage in soils and AGB across dryland sub-Saharan Africa. The data gaps and interdisciplinary opportunities we have identified are summarised in Table 1. The need for these evidence gaps to be filled using new and integrated methodological approaches has been situated within the context of the political and economic opportunities and challenges for carbon sequestration to deliver ecosystem service and poverty alleviation benefits (Fig. 1). For example, with improved data on SOC and AGB, model uncertainty can be reduced, leading to more accurate and reliable spatial predictions of stores and fluxes. With this information, maps can be developed to inform decision making and policy development, enhancing practice through the development of payment schemes for carbon storage that build on community-level institutions and multi-stakeholder partnerships, and which both help to mitigate climate change and provide adaptation options. Current research nevertheless fails to 'join the dots' between these different aspects.

Fig. 1 outlines an interdisciplinary multi-stakeholder pathway to integrate new scientific knowledge with policy and practice to deliver poverty reduction and ecosystem services benefits, while the research and practical experiences drawn upon in our analysis highlight the importance of collaborative multi-stakeholder working across scales. Improved data and knowledge on the spatial distribution of carbon storage and release, whilst important in its own right, will not directly create poverty alleviation, carbon storage, adaptation and ecosystem service benefits without new forms of collaborative working across academic disciplines and with

Table 1 – Data gaps and opportunities.

Data gaps	Methodological and development opportunities
Insufficient data on the amount, spatial distribution and form of SOC at appropriate scales, particularly in drylands	Incorporation of sampling strategies (e.g. crust sampling) that match dryland characteristics within protocols used in major regional and global soil databases. GEFSOC provides a protocol for linking existing GIS-based soil and terrain information to field-collected soil C data, but still requires an accepted sampling method
Lack of empirical data on CO_2 efflux from the soil surface: this is vital to advance models of flux variability and predict annual losses under given land use, soil and climate conditions	Use of new, in situ chamber monitoring over larger areas, with a view to feeding data into models such as JULES, GEFSOC and SPA. Such monitoring will enable separation of soil CO_2 efflux into autotrophic components and heterotrophic mineralisation of soil organic matter
Lack of data on the amount, distribution and species composition of dryland soil microbes, critical to the respiration and fate of SOC	Improved understanding of the microbial processes affecting the soil C store, including microbial content and enzyme activity analyses, moving us towards identifying tipping points at which SOC storage capability declines or respiration losses are instigated
Limited measuring and monitoring data on the spatial distribution of AGB	New permanent monitoring plots in drier savanna woodland, grassland environments and across a broader range of miombo woodlands. These can be used to calibrate and validate EO estimates of AGB and their associated errors, allowing more accurate regional assessment of carbon storage
Poor understanding of the natural and human drivers of AGB fluxes	Sub-sampling is needed in regions of similar climatic influence but different human impacts, linked to participatory monitoring approaches, disturbance histories and indigenous knowledge
Limited understanding of how ecosystem services relate to AGB and how changing management will drive changes to AGB and ecosystem services	Livelihood and AGB surveys within regions of similar climatic influence but of different land management to quantify those ecosystem services relied upon by local communities and learn how they vary under different land management regimes
Need to better understand the relationships between OC and ecosystem service provision, linked to a more holistic approach to human-environment relationships, especially in light of the drivers of future change	Inter- and multi-disciplinary approaches, working with multiple stakeholders at a range of scales. Scenario and analogue approaches offer an important window into future relationships between drivers of change, poverty, carbon storage and ecosystem services
Lack of understanding relating to poverty-environment relationships and the implications this has for the design and implementation of carbon payment schemes	Large-scale databases linked to local classifications of poverty and patterns of ecosystem service provision and access, with projects building on local institutions and priorities
Shortage of appropriate tools and methodologies in informing land management decisions and lack of ability to identify thresholds at which land users will shift their management strategies towards carbon mitigation scenarios	Decision-support tools to raise awareness of different land management strategies. Lessons need to be assessed, evaluated and where appropriate, transferred, from forest settings to rangeland contexts, in order to engage pastoralists in community carbon initiatives
Lack of understanding on how to reduce transaction costs for the rural poor when engaging in carbon trading	Simple and cost-effective carbon accounting methodologies need to be tested and validated. Ways to secure yearly payments, alongside adequate organisational structures that enable smallholders to coordinate themselves into larger units to reduce costs are required

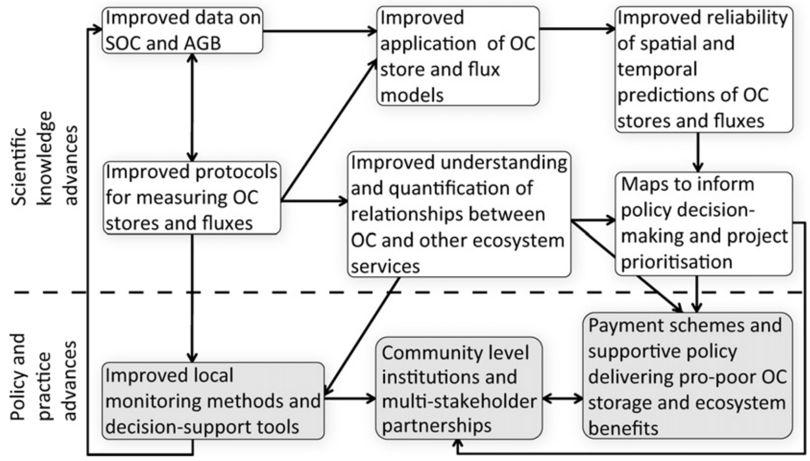

Fig. 1 – Possible route to delivering pro-poor carbon storage and ecosystem service benefits based on an improved scientific evidence base.

partners at the community-level, in the private sector and in national government. Reflections on the experiences of such multi-stakeholder, multi-level partnerships will be essential to the wider uptake of carbon-friendly land management projects with support from international bodies and the private sector in ensuring the full valuation of benefits and their trading in the emerging climate finance sector.

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