Grazing Lands, Livestock and Climate Resilient Mitigation in Sub-Saharan Africa:

The State of the Science

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Cover Photo: Grazing lands in Damot Gale, SNNPR, Ethiopia. July 2014. Credit: E. Milne.

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Introduction

Why are grazing lands important globally?

More than half of the world's land surface is grazed (Follet and Reed, 2010) and just under one third (31%) is grassland (including grasslands, shrublands and savannah) (Petri et al., 2010). Grazing lands therefore represent a huge stock of carbon (C), not because of their C content (which is relatively low per unit area) but because of the vast area they occupy. The majority of this C stock is held in the soil and can be influenced by environmental factors and management. There is, therefore, scope for existing stocks to be maintained and, in degraded grazing lands, for depleted stocks to be increased. This makes grazing lands of great interest in terms of sequestering carbon from the atmosphere as a means of mitigating climate change, with estimated sequestration rates of ~ 0.6 gigatons (Gt) CO_2 equivalents yr^{-1} (Gerber et al., 2013). It has been estimated that grazing lands account for a quarter of potential C sequestration in world soils (Follet and Reed, 2010). Despite this, they are neglected in terms of inclusion in mitigation strategies.

Increasing carbon stocks in grazing lands is a 'win-win' strategy. It improves water infiltration and cycling, increases productivity and is linked to increased biodiversity both below and above ground. In addition, grazing lands support some of the world's poorest people (Neely et al., 2010) and livestock is growing as a sector, accounting for up to 50% of GDP in countries with significant areas of rangeland (World Bank, 2007). We therefore have a window of opportunity to improve livelihoods, mitigate climate change and provide a range of other environmental benefits through C-friendly management of grazing lands.

Why focus on Sub-Saharan Africa?

Sub-Saharan Africa is home to 23% of the world's poor, the majority of who depend on livestock for some part of their livelihoods (Thornton et al., 2002). There are an estimated 25 million pastoralists and 240 million agro-pastoralists in Sub-Saharan Africa (IFPRI and ILRI, 2000). Increasing population levels, loss of grazing lands to cropland, urbanisation and enforced settlement are threatening grazing lands and many of the traditional systems which have been practiced for hundreds of years. There is a need to show the benefits of grazing lands in terms of the services they provide including the role they can play in climate change mitigation. Batjes (2004) estimates that grasslands in Africa have the potential to sequester between 7 – 42 Tg C $year^{-1}$. In addition, climate change is already having an impact (changing rainfall and temperature regimes) in arid and semi-arid regions of Sub-Saharan Africa where livelihoods are mainly supported by livestock, with future impacts predicted to be most negative for grasslands around Senegal and in Southern Africa, where Namibia, Botswana and South Africa meet (van de Steeg et al., 2013). Many management strategies which increase above and below ground carbon stocks, thereby benefitting climate change mitigation, can also have adaptation benefits, leading to improved use of scare resources such as water and manure.

What is the purpose of this report?

This report is a detailed review, synthesis, and analysis of the current "state of the science" concerning the potential for carbon sequestration in grazing lands through improved land management practices in Sub-Saharan Africa (SSA). It aims to provide an up-to-date assessment of the science of C sequestration from improved land management, including the current levels of uncertainty, major gaps in knowledge and data, areas for near term research and development, major determinants of sequestration potential, and current and potential scientific monitoring tools. The report firstly gives an overview of current grazing lands in SSA (Chapter 1) and explores the major determinants of C sequestration in grass/rangeland systems (Chapter 2). It then considers current research work on C impacts of grazing land management systems (Chapter 3). Available measurement techniques are summarized in Chapter 4 and a map based approach is then used in Chapter 5 to estimate present C stocks in grazing lands in SSA. This is followed by Chapter 6 which looks at available modeling techniques and Chapter 7 which presents a model based estimate of C sequestration potential in grass/rangelands in SSA. The final chapter (Chapter 8) provides a synthesis of the report's findings.

Who will the output be helpful to?
This work has been supported by USAID and the output aims to provide USAID with an up-to-date assessment. In addition, it is hoped it will be useful to other funding bodies (GEF, World Bank, bilateral donors, etc.), non-governmental organizations (NGOs) (both national and international), policy makers and scientists who support climate change mitigation and adaptation work in Africa.

Chapter 1. Current types of grazing lands in Sub-Saharan Africa and associated management practices

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1. 1 Introduction

Sub-Saharan Africa (SSA) covers an area of ca. 2.4x10^9 ha of which ca. 35% is permanent pasture (Table 1). Two-thirds of SSA is made up of arid and semi-arid lands. Over 60% of the population of SSA depends on agriculture (including pastoralism) for their livelihood and agriculture accounted for on average 29% of GDP between the years 1998 and 2000 (IAASTD, 2009). Approximately 65% of agricultural land, 35% of permanent pastures and 19% of forest and woodland in the region are estimated to be affected by some form of degradation (Oldeman, 1994; WRI, 2005). Approximately 25 million pastoralists and 240 million agro-pastoralists depend on livestock as their primary source of income (AU-IBAR, 2012). Soil erosion and land degradation represent a major threat to food security in SSA rangelands (Kiage, 2013). Climate change poses additional challenges and threats to livelihood and sustainable development objectives within the region.

Table 1. Land use in Sub-Saharan Africa ca. 2002 (WRI, 2005).

Land-use	Cover (%)
Permanent pasture	35
Arable and permanent cropland	8
Forested (FAO estimates for 2000 >10% cover)	20
All other land	37
Total	100

1.2 Major Livestock Production Systems in SSA

Grazing lands are defined as "land used for animal production e.g. natural or semi- natural grasslands, open woodlands, improved or planted pastures" (WOCAT, 2008). Therefore grazing lands can include non-grasslands. Grazing lands represent both a source of animal feed and a key element in biodiversity protection. Most rural people living in SSA rangelands are agro-pastoral, combining small-scale farming with livestock keeping, or specialize in herding (pastoralists) or farming (Homewood, 2004). In much of SSA, grazing lands are primarily governed by common-property regimes, which enable people to pool and reduce the risks associated with variable forage production. The ability of the land to sustain increasing numbers of livestock owners without damaging the environment will be determined in part by the way the users themselves can govern access to and use of this vital resource (McCarthy et al., 1999).

In SSA, grazing lands can include rangelands, croplands, and forestlands. Across these land cover types, different livestock production systems can be distinguished. These include pastoralism, agro-pastoralism and mixed crop-livestock systems. Livestock production systems in SSA are determined by rainfall amount and seasonality (uni-modal or bi-modal), population density and cultural predispositions. In very general terms it can be said that historically pastoralism dominates in the drylands of eastern Africa while limited crop-livestock integration and agro-pastoralism dominate in the dryland ecosystems of western Africa and that this difference can be attributed partially to bi-modal versus uni-modal rainfall patterns (Ellis and Galvin, 1994). Figure 1 provides the most recently updated map of global livestock production distribution in Sub-Saharan Africa.

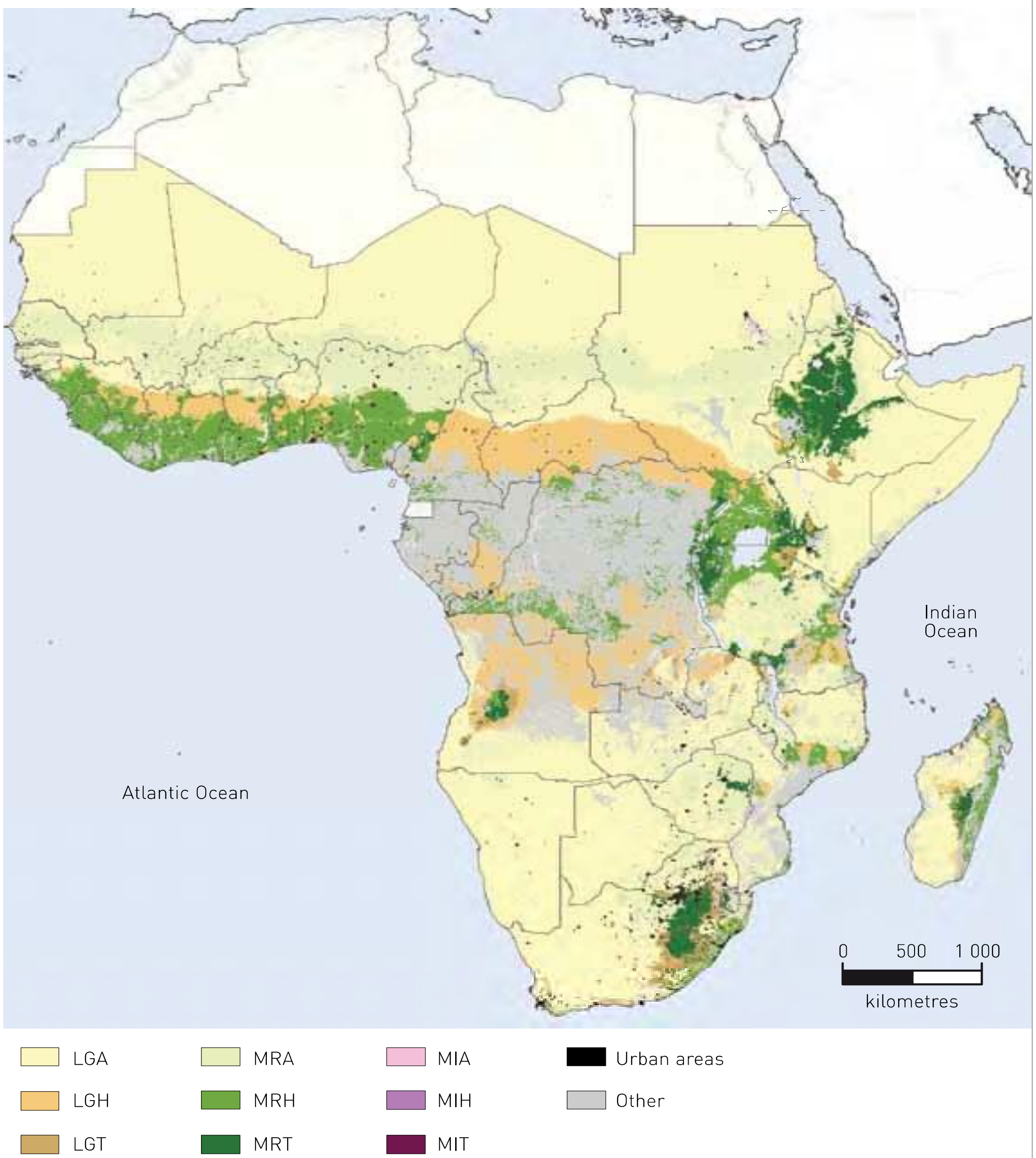

Figure 1 Livestock production systems in SSA (Robinson et al., 2011). *Legend: LGA Livestock only systems, arid and semi-arid, LGH Livestock only systems, humid and sub-humid, LGT Livestock only systems, highland/temperate, MRA Rainfed mixed crop/livestock systems, arid and semi-arid, MRH Rainfed mixed crop/livestock systems, humid and sub-humid, MRT Rainfed mixed crop/livestock systems, highland/temperate, MIA Irrigated mixed crop/livestock systems, arid and semi-arid, MIH Irrigated mixed crop/livestock systems, humid and sub-humid.*

Pastoralism

The term ‘rangeland’ is used more narrowly than grazing lands to describe “land on which the native vegetation (climax or natural potential) is predominantly grasses, grass-like plants, forbs, or shrubs”. Rangelands include natural grasslands, savannas, shrub lands, most deserts, tundra, alpine communities, coastal marshes, and wet meadows. Rangelands occupy 51% of the terrestrial land surface and contain ca. 36% of the world’s total carbon in the above and belowground biomass. Rangelands make up ca. 43% of Africa’s inhabited surface. These lands are very diverse and characterized by high variability in terms of rainfall, grazing intensity, vegetation dynamics, and the effects of biotic and abiotic variables. Recent studies suggest that most arid and semi-arid rangeland systems exhibit both equilibrium and non-equilibrium states at different scales, which implies a management approach that takes into account temporal variability and spatial heterogeneity (Okayasu et al., 2011; Vetter, 2005). In SSA, much of the rangeland is utilized by nomadic pastoralists who move their herds and flocks over extensive areas during the course of the year to find water and grazing for their livelihood (Hodgson, 1999). This is predominantly the case in the drylands of eastern Africa, due in part to the bi-modal rainfall regime with insufficient rainfall in either rainy season to reliably sustain crops (Ellis and Galvin, 1994).

Pastoral grazing strategies in SSA form sophisticated management systems to sustain human life in harsh environments. Pastoralists have been, and continue to be, the custodians of African rangelands. Pastoral systems are one of the dominant forms of livestock production system in SSA and are found mainly in arid and semi- arid areas, and in limited areas in the sub-humid zones in East Africa and West Africa (Sandford, 1983; Wilson et al., 1983; Ellis and Swift, 1988). In the drier areas of northern Kenya and Southern Ethiopia, crop agriculture is impossible or unreliable and the pastoral scene is dominated by extensive nomadic pastoralism as in Turkana and Borana (Ellis et al., 1987).

Pastoral institutions need to be understood and built upon when contemplating actions to improve carbon management. For example, there is an extensive literature about traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods (Ellis and Swift, 1988; McCabe, 1990, 1994, 2004; Behnke, 1999; McCabe et al., 1999; Niamir-Fuller, 1999; Turner, 1999; Fernandez-Gimenez, 2002; Homewood et al., 2004; Vetter, 2005; Fernandez-Gimenez and Le Febre, 2006; Reid et al., 2008; Butt et al., 2009; Nkedianye et al., 2011; Turner, 2011; Goldman and Riosmena, 2013; Tsegaye et al., 2013). These institutions (rules and norms) determine when and where people graze, for how long, with what types of livestock, who uses which water source, and how to judge grassland health. These pastoral institutions and norms have developed over the last 7000 years, when pastoralists became the first people to produce food in much of Sub-Saharan Africa (Marshall, 1998; Marshall and Hildebrand, 2002), and they are still negotiated today. This is especially true in the face of major development changes in pastoral systems, especially in areas near towns and market centers, but also in more remote rangelands where unprecedented changes in climate are starting to occur and options for movement are more restricted than in the past (Reid et al., 2008). Pastoral systems have been developed to sustain life in rangelands where vegetation and water resources are often ephemeral in time and patchy in space, and pastoral movement to these ephemeral patches is a critical strategy to improve milk and meat production (Boone and Hobbs, 2004).

Agro-pastoralism

Agro-pastoralism is found in the semiarid, sub-humid and humid tropics and in tropical highland areas (Sere and Steinfeld, 1996). Livestock are dependent on natural forage and cropping is important but there is low integration with livestock. Livestock productivity is higher than in the pastoral system but still insufficient to meet the needs of the growing population in SSA. This system is particularly well developed in densely inhabited areas of eastern and southern Africa where cold storage facilities allow for longer-term storage of meat. Agro-pastoralism is growing in eastern Africa in areas traditionally used for nomadic pastoralism which are becoming ever more fragmented by the encroachment of cropland (Greiner, 2013).

In West Africa there exist transhumant systems in which livestock are taken north into the Sahel during the wet season to graze and then return south to the higher rainfall zones where they graze on crop residues (Brottem et al., 2014). The owners of the fields may be from the same or different ethnic groups. This seems to be a fairly symbiotic relationship in which the pastoralists gain livestock fodder and the agriculturists gain manure and urine inputs to their fields (high nitrogen inputs in exchange for low nitrogen contributions). It is not clear what the net C sequestration balance for these fields is. This is a type of integrated agro-pastoral system in which the two components are spatially separated by quite long distances.

There are also agro-pastoral systems in eastern Africa (Coppock, 1990), but there the movements of livestock are not transhumant. There may be situations in which the agro-pastoralists are essentially sedentary, living near their fields and grazing their livestock within a restricted grazing radius of that location, returning to the central place every night. In this case, there are opportunities for manure to be collected and used in the fields. In other cases the fields and the grazing areas may be more distant from each other and grazing more extensive with less frequent return to the central place. These types of systems are becoming more common in East Africa wherever rainfall is sufficient to support production of a crop fairly consistently (annual precipitation > 700 mm) and where there is movement towards privatization of the lands (Bekure et al., 1991) as is found on the Maasai steppe and in the Kajiado district in Kenya.

Mixed crop-livestock system
In these systems, crops and livestock are integrated on the same farm with livestock commonly grazed on cropped areas. These systems are widespread in rain-fed areas (Thomas, 2006), and form the backbone of smallholder agriculture (Devendra et al., 2005). These mixed systems predominate in humid and sub-humid agro-ecological zones but they are also found in arid and semiarid tropics and the tropical highlands of East and West Africa. Ruminant animals graze native pastures and use crop residues as additional feed sources after harvest, whereas non-ruminants depend on crop by-products and household kitchen wastes.

1.3 Grazing land Management Practices in SSA

Across these different land use systems, farmers and livestock keepers use a wide range of management practices to primarily achieve profitable gains (food security, livelihoods, income, etc.) but also to improve the “condition/health” of the grazing lands. Most, if not all, of the management practices aim predominantly to a) reduce and combat land degradation, b) restore/rehabilitate the land, and c) improve land productivity for livestock production. Therefore all have a potential impact on carbon stocks in soils and biomass.

These management practices include:

- **Grazing management:** controlled grazing management practice is considered beneficial in conditions of poor vegetation cover, overgrazing and degraded soils, and is considered as the most promising sustainable land management practice to restore degraded rangelands as it enhances the vigor of mature perennial grasses (AU-IBAR, 2012). Rotational grazing is a management system based on the subdivision of the grazing area into a number of enclosures and the successive grazing of these paddocks by animals in a rotation so that not all the area is grazed simultaneously. The main principles of rotational grazing are: (1) control the frequency at which pasture is grazed, (2) control the intensity at which the pasture plants are grazed by controlling the number of animals which graze each paddock and their period of occupation, and (3) reduce the extent of selective grazing by confining a relatively large number of animals to a small portion of the rangeland.

- **Livestock management:** control of the stocking rate, animal breeds and species, improved feeding practices.
- **Restoration of degraded rangelands**: by vegetation management - use of fire, chemical control and re-vegetation of degraded and bare lands, introduction of grass species with higher productivity, or carbon allocation to deeper roots, have been shown to increase soil carbon - other land management (e.g. soil and water conservation practices, agro-forestry, fencing, nutrient management, etc.).

Table 2, an excerpt from Smith et al., (2007), lists proposed measures for mitigating greenhouse gas emissions from grazing lands and livestock production systems, their apparent effects on reducing emissions of individual gases where adopted (mitigative effect), and an estimate of scientific confidence that the proposed practice can reduce overall net emissions at the site of adoption. Chapter 3 of this report reviews the current science surrounding the carbon impacts of grazing land management systems and practices in Sub-Saharan Africa.

Table 2. Measures for mitigating GHG emissions from livestock production systems (from Smith et al., 2007)

Measure	Examples	Mitigative Effects[a]			Net Mitigation[b] (confidence)	
		CO2	CH4	N2O		
Grazing land management/ pasture improvement	Grazing intensity Increased productivity (fertilizers) Nutrient management Fire management Species introduction (including legumes)	+/- + + + +	+/- + 	+/- +/- +/- +/- +/-	* ** ** * *	* * ** * **
Restoration of degraded rangelands	Erosion control, organic amendments, nutrient amendments	+		+/-	***	**
Livestock management	Improved feeding practices Specific agents and dietary additives Structural changes, animal breeding		+ + +	+ +	*** ** **	*** *** *
Manure/ biosolid management	Improved storage and handling Anaerobic digestion More efficient use as nutrient source	 +	+ + 	+/- +/- +	*** *** ***	** * **

Notes

[a] + denotes reduced emissions or enhanced removal (positive mitigation effect)

- denotes enhanced emissions or reduced removals (negative mitigation effect)

+/- denotes uncertain or variable response

[b] A qualitative estimate of the confidence in describing the proposed practice as measures for reducing net GHG emissions. Agreement refers to the relative degree of consensus in the literature (the more asterisks the higher the agreement). Evidence refers to the relative amount of data in support of the proposed effect (the more asterisks the more evidence).

Chapter 2. Analysis of the key determinants of Carbon sequestration in grass/rangeland systems in different conditions

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All ecosystems – forested ecosystems, agro-ecosystems, grassland ecosystems, etc. – take up atmospheric CO_2 and mineral nutrients and transform them into organic products. Ecosystems are thus a major source and sink for the three main biogenic greenhouse gases – CO_2, N_2O, and CH_4. In undisturbed ecosystems, the carbon balance tends to be positive: carbon uptake through photosynthesis exceeds losses from respiration; even mature, old-growth forest ecosystems take up slightly more carbon than they release (Stephens et al., 2007; Gough et al., 2008; Luyssaert et al., 2008). Disturbance, such as fire, drought, or disease, can lead to substantial losses of carbon from both soils and vegetation (Page et al., 2002; Ciais et al., 2005; Adams et al., 2009). Disturbance is a defining element of all ecosystems that continues to influence the carbon uptake and losses that determine long-term ecosystem carbon balance (Randerson et al., 2002).

Human land use activities often function much like natural disturbances in their influence on ecosystem carbon balance. Carbon dioxide is produced when vegetation is burned, and soil carbon stocks begin to decline soon after disturbances to the soil (Smith et al., 1999). Like natural disturbances such as fire and drought, which prompt release of C and decrease C uptake, land use change affects vegetation and soil dynamics, often prompting longer-term increases in carbon release or decreases in carbon uptake. Deforestation, degradation of native grazing lands, and conversion to cropland have prompted losses of 450-800 Gt CO_2 from vegetation and soil carbon pools – equivalent to 30-40% of cumulative fossil fuel emissions (Houghton et al., 1983; DeFries et al., 1999; Marland et al., 2000; Olofsson and Hickler, 2008). Emissions from deforestation have dominated carbon losses from terrestrial ecosystems (DeFries et al., 1999), but substantial amounts of carbon have been lost from vegetation and soils of systems other than forests as well (Shevliakova et al., 2009).

Future carbon balance in ecosystems will be affected by the climate, though the net global response of ecosystems remains uncertain due in part to several feedbacks (Field et al., 2007). Carbon stored in vegetation (~630 Pg C) and soil (at least 1600 Pg C, but perhaps as much as twice that (Tarnocai et al., 2009)) is vulnerable to loss at warmer temperatures (Friedlingstein et al., 2006; Piao et al., 2009). Whether disturbances decline, mitigating future carbon losses, or the scope of human-induced disturbances continues to expand in the future is a very important determinant of future concentrations of atmospheric CO_2 (Page et al., 2002; Limpens et al., 2008).

Much of the world's grazing lands are under pressure to produce more livestock by grazing more intensively, particularly in Africa's rangelands, which are vulnerable to climate change and expected to still supply most of the beef and milk demanded in Africa (Reid et al., 2004). As a result of past practices, somewhere between five and ten percent of the world's grasslands have been degraded by overgrazing (Oldeman, 1994). Previous research has documented that improved grazing management could lead to greater forage production, more efficient use of land resources, and enhanced profitability and rehabilitation of degraded lands (Oldeman, 1994). The tightening linkage between ecosystem services and human well-being in the world's dryland systems acutely demonstrates the need for a new, integrated approach to diagnosing and addressing sustainable development priorities, including maintenance of the supply of critical ecosystem services.

One of the reasons for the intensive use of grasslands is the high natural soil fertility. Grasslands characteristically have high inherent soil organic matter content averaging 331 Mg ha^{-1} (Schlesinger, 1977). Soil organic matter influences the fate of organic residues and inorganic fertilizers, increases soil aggregation which can limit soil erosion, and also increases cation exchange and water holding capacities (Miller and Donahue, 1990). In addition, soil organic matter is an important source of plant nutrients and can enhance production (Kononova, 1966; Allison, 1973; Tate, 1987; Miller and Donahue, 1990), and is a key regulator of grassland ecosystem processes. Thus a key underlying goal of sustainable management of grassland ecosystems is to maintain high levels of soil organic matter and soil carbon stocks.

Portions of the grasslands on every continent have been degraded due to human activities, with about 7.5% of grasslands having been degraded because of overgrazing (Oldeman, 1994). More recently, the Land Degradation Assessment in Drylands (LADA) concluded that about 16% of rangelands are currently undergoing degradation and that rangelands comprise 20-25% of total land area currently being degraded, which affects the livelihoods of over 1.5 billion people worldwide (Bai et al., 2008). This current degradation is likely occurring in addition to historic degradation (Bai et al., 2008). Historically, intensive use of grasslands has resulted in the transfer of 993 Mt of soil carbon to the atmosphere in the form of CO_2 just from land use change in the United States (Kern, 1994). Soil organic matter losses due to conversion of native grasslands to cultivation are both extensive and well documented (Donigian et al., 1994; Kern, 1994; Follett et al., 2001a). Removal of a large amount of aboveground vegetation, continuous heavy stocking rates, and other poor grazing management practices are important human-controlled factors that influence grassland production and have also led to depletion of soil carbon stocks (Ojima et al., 1993; Conant and Paustian, 2002). However, good grassland management can potentially reverse historical soil carbon losses and sequester substantial amounts of carbon in soils. Disturbance through overgrazing, fire, invasive species, etc. can deplete grassland systems of carbon stocks (Smith et al., 2008). Harvesting a large proportion of plant vegetation enhances yields of useful material (e.g., for forage or fuel), but decreases carbon inputs to the soil (Wilts et al., 2004). Primary production in overgrazed grasslands can decrease if herbivory decreases plant growth capacity, vegetation density, community vegetation, or, if community composition changes (Chapman and Lemaire, 1993). If carbon inputs to the soil in these systems decrease because of decreased net primary production or direct carbon removal by livestock, soil carbon stocks would decline. Like carbon sequestration in forests or agricultural land, sequestration in grassland systems – primarily, but not entirely in the soils – is brought about by increasing carbon inputs. It is widely accepted that *continuous excessive* grazing is detrimental to plant communities (Milchunas and Lauenroth, 1993) and soil carbon stocks (Conant and Paustian, 2002). When management practices that deplete soil carbon stocks are reversed, grassland ecosystem carbon stocks can be rebuilt, sequestering atmospheric CO_2 (Follett et al., 2001a).

Many management techniques intended to increase livestock forage production have the potential to increase soil carbon stocks, thus sequestering atmospheric carbon in soils. Methods of improved management include fertilization, irrigation, intensive grazing management, and sowing of favorable forage grasses and legumes. Grassland management to enhance production (through sowing improved species, irrigation or fertilization), minimize negative impacts of grazing, or rehabilitate degraded lands can each lead to carbon sequestration (Conant et al., 2001; Follett et al., 2001b; Conant and Paustian, 2002; Ogle et al., 2004). Improved grazing management (management that increases production) leads to an increase of soil carbon stocks by an average of 0.35 t C ha^{-1} yr^{-1} (Conant et al., 2001; Conant et al., 2015). Agroforestry enhances carbon uptake by lengthening the growing season, expanding the niches from which water and soil nutrients are drawn, and in the case of nitrogen-fixing species, enhancing soil fertility (Nair et al., 2009). The result is that when agroforestry systems are introduced in suitable locations, carbon is sequestered in the tree biomass and tends to be sequestered in the soil as well (Jose, 2009) and sometimes leads to increased forage productivity. Improved management in existing agroforestry systems could sequester 0.012 Tg C yr^{-1} while conversion of 630 Mha of unproductive or degraded croplands and grasslands to agroforestry could sequester as much as 0.59 Tg C annually by 2040 (IPCC, 2000), which would be accompanied by modest increases in N_2O emissions as more nitrogen circulates in the system.

Using seeded grasses for cover cropping, catch crops (fast growing crops grown between main crops), and more complex crop rotations all increase carbon inputs to the soil by extending the time over which plants are fixing atmospheric CO_2 in cropland systems. Rotations with grass, hay, or pasture tend to have the largest impact on soil carbon stocks (West and Post, 2002). Adding manure to soil builds soil organic matter in grasslands (Conant et al., 2001). The synthesis by Smith et al. (2008) suggests that adding manure or biosolids to soil could sequester between 0.42 and 0.76 t C ha^{-1} yr^{-1} depending on the region (sequestration rates tend to be greater in moist regions than in dry). Rapid incorporation of manure into fields would reduce the time that manure decomposes in anaerobic piles and lagoons, reducing emissions of CH_4 and N_2O. Smith et al. (2007) estimate the technical potential for reduction of CH_4 emissions from manure to be 12.3 Tg C yr^{-1} by 2030; N_2O emissions could also be reduced. Adding manure in one place to build soil carbon stocks is offset by removal or what would be carbon inputs in another place (by forage or feed harvest); the balance between these has not been well characterized.

Globally an estimated 0.2-0.8 Gt CO_2 yr^{-1} could be sequestered in grassland soils by 2030 given prices for CO_2 of 20-50 $US per tonne (Smith et al., 2007). This estimate accounts for the fact that sometimes costs preclude adoption of practices even when they have multiple benefits. Although fertilization and fire management could both contribute to carbon sequestration, most of the potential sequestration in non-degraded grasslands is through changes in grazing management practices. Estimated rates of carbon sequestration per unit area are lower than those for sequestration on agricultural land, but the total sequestration potential is comparable to that of agricultural lands because grasslands cover such a large portion of the Earth's surface. Nearly 270 Mha of grassland worldwide have been degraded to some degree by mismanagement (Oldeman, 1994; Bridges and Oldeman, 1999). Much of this land can be rehabilitated by enhancing plant productivity, capturing water resources and using them more efficiently, or improving soil fertility; doing so could sequester about as much carbon as could be sequestered in agricultural lands (0.15-0.7 Gt CO_2 yr^{-1} depending on carbon prices) (Smith et al., 2007).

Grasslands contain a substantial amount of the world's soil organic carbon. Integrating data on grassland area (FAO, 2009) and grassland soil carbon stocks (Sombroek et al., 1993) results in a global estimate of about 343Bt C – nearly 50% more than is stored in forests worldwide (FAO, 2007). Just as is the case for forest vegetation carbon stocks, grassland soil carbon stocks are susceptible to loss upon conversion to other land uses (Paustian et al., 1997) or following activities that lead to grassland degradation (e.g., overgrazing). Current rates of carbon loss from grassland systems are not well-quantified. Over the last decade grassland area has been declining while arable land area has been growing, suggesting continued conversion of grassland to croplands (FAO, 2009). When grasslands are converted to agricultural land, soil carbon stocks tend to decline by an average of about 60% (Paustian et al., 1997; Guo and Gifford, 2002). Grassland degradation has also expanded (Bai et al., 2008), likely prompting the loss of grassland ecosystem carbon stocks. Arresting grassland conversion and degradation would preserve grassland soil carbon stocks. The magnitude of the impact on atmospheric CO_2 is much smaller than that due to deforestation, but preserving grassland soil carbon stocks serves to maintain the productive capacity of these ecosystems that make a substantial contribution to livelihoods.

Gaps in our understanding
In addition to the determinants of C sequestration discussed above there are a number of aspects of determinants which are unique to the situation in SSA and of which our understanding is still limited. Perhaps the largest of these is an understanding of shifts between shrublands and grasslands, the drivers of this change, and the impact this has on above and below ground C stocks. Anthropogenic management of woody vegetation and impacts on C stocks are considered in Chapter 3. Another area which is poorly understood is the role of termites. Termites play an important role in savannas and grasslands in SSA, bringing lighter textured soil to the surface and concentrating nutrients, something which has long been utilized by African farmers who use the mounds as fertilizer (Reid, 2012a). The role of termites in determining soil organic carbon (SOC) stocks need to be studied further in SSA. Indeed the role of below ground biodiversity in determining SOC in grazing lands in SSA in general needs further investigation.

There are gaps of our understanding of phosphorus and the role it plays in C sequestration in grasslands. The co-limitation of nitrogen and phosphorus in grasslands, particularly those dominated by C4 species (as is the case in SSA), is not well understood (Craine et al., 2008). The role of ultra violet (UV) radiation in the decomposition of grass litter and how this varies with aridity is of great relevance to SSA. Limited studies in the US have shown that photodegradation plays an important role in the decomposition process but the interaction with aridity is not well understood (Brandt et al., 2010). Similar investigations need to be conducted for different UV and aridity gradients in SSA.

Chapter 3. Carbon impacts of grazingland management systems and practices in Sub-Saharan Africa

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In this chapter the management of grasslands and subsequent impacts on carbon stocks are considered. Those aspects of grazing land management with the greatest relevance to SSA are explored, namely; grazing management, choice of livestock type, fire, management of woody biomass and management of water.

3.1 Grazing and browsing

Finding fodder for livestock is a crucial part of daily life for many in Sub-Saharan Africa (SSA). Chapter 1 outlines the variety of livestock producing systems in SSA and shows that systems which involve direct grazing or browsing are prevalent in the region. Livestock are an irreplaceable source of livelihood for the poor and are one of the fastest growing agricultural sectors globally (World Bank, 2008). Between 2000 and 2010, production of livestock increased by 3% in SSA (a relatively small increase given human population growth). However, this change was driven by increasing livestock numbers rather than gains in livestock productivity (Upton, 2012) putting pressure on SSA grazing lands and traditional grazing systems. Disagreement exists about the overall impact of grazing on both above and below ground carbon stocks with grazing being shown to have both positive (Reeder and Schuman, 2002) and negative (Hiernaux et al., 1999) effects in different situations. What is agreed is that grazing impacts on carbon stocks are site specific, being dependent on many factors including climate, soil type, land use history, vegetation, livestock type and grazing intensity. Here we explore some of the evidence relevant to SSA of grazing impacts on carbon (C) stocks under different aspects of land management.

3.1.1 Climate/grazing interactions and Soil Organic Carbon (SOC)

Soil organic carbon (SOC) refers to the organic carbon in the soils derived from decaying plant and animal material. This differs from soil inorganic carbon (SIC) which refers mainly to C found in carbonate minerals derived from rock parent material. Total carbon (TC) includes both organic and inorganic soil carbon (Nelson and Sommers, 1996). Most studies of climate change mitigation focus on SOC as this type of soil carbon can be manipulated by changes in land cover, use and management. Potential for C sequestration in SIC through changes in land management is deemed to be low with processes being extremely slow and poorly understood (Diaz, 2010). Therefore, here, in common with most other studies of climate change mitigation potential in land, we focus on SOC.

SSA has a wide range of climate regimes. The regions around the equator have bi-modal rainy seasons while regions further poleward have a single rainy season. Moving further away from the equator, the length of the rainy season becomes shorter. Extensive grazing land in Africa is concentrated in the semi-arid environments for several reasons: (i) crop production in these regions is limited by rainfall, thus reducing to some degree competing demands for land, (ii) forage production in the arid and semi-arid regions, although less in terms of biomass, is generally much better in terms of nutritional value compared to forage from the mesic savannas and humid zones, and (iii) most ruminants (e.g., cattle, sheep and goats) suffer increasing disease incidence in more humid regions reducing their utility for meat and milk production. However, in the semi-arid regions, seasonal and inter-annual variability in rainfall constrain not only the availability of forage but also the availability of surface water for animals to drink. Ellis and Swift (1988) put forward the theory that when high inter-annual rainfall variability is experienced, land degradation through grazing tends to be relatively low as periods of drought cause herbivore populations to collapse, allowing vegetation to recover. In these 'non-equilibrium' areas, they suggested that grazing is unlikely to cause land degradation. However, in more mesic (wetter) environments sustained grazing pressure is more likely to lead to degradation. Von Wehrden et al. (2012) reviewed 58 studies where rainfall variability and land degradation had been studied. They found Ellis and Swift's theory to hold true: degradation caused by grazing was found almost exclusively in higher rainfall grazing-lands with lower inter-annual variation.

This has major implications for our understanding of grazing impacts on soil carbon stocks (which tend to be low in degraded lands) in SSA. Studies from temperate regions have reported grazing can increase soil carbon in 0-30 cm soil layers in drier short-grass prairie ecosystems, while decreasing soil carbon stocks in more humid mixed- and tall-grass prairies (Derner et al., 2006). Derner and Schuman (2007), reviewing precipitation and grazing effects on SOC in temperate rangelands of the Great Plains, also concluded that soil carbon sequestration increased with grazing in drier rangeland sites. This identifies with a threshold in mean annual precipitation of 600 mm, above which grazing reduces soil carbon in rangelands, attributed to changes in the amount of root material in the upper soil profile. In SSA, rangeland systems with less than approximately 600 mm annual precipitation tend to be non-equilibrial (Ellis and Swift, 1988) and grazing effects are overridden by those of rainfall variability.

McSherry and Ritchie (2013) carried out a meta-analysis of studies considering grazing impacts on SOC stock rather than concentration (i.e., stock or SOC concentration plus bulk density were reported). For the 17 studies they considered they did not find a direct relationship between annual precipitation and grazing effect on SOC. They did, however, find different grazing effects on SOC for grasslands dominated by warm season (C4) or cool season (C3) grasses. They found that in C4 dominated grasslands and C3-C4 mixed grasslands increasing grazing intensity increased SOC by ~6-7%, whereas in C3 dominated grasslands increasing grazing intensity decreased SOC by ~ 18%. This has important implications for SSA where low altitude grasslands are dominated by C4 species and high altitude grasslands by C3 species (Tieszen et al., 1979). These findings suggest management of grazing to maintain/increase SOC needs to be different in the highlands and the lowlands of SSA.

Summary points

- In mesic environments sustained grazing pressure is more likely to lead to land degradation (and therefore may cause SOC to decrease)
- In dry areas with high rainfall variability land degradation from sustained grazing may not be seen as grazing effects are overridden by rainfall effects
- Findings on grazing/precipitation interactive effects on SOC are mainly from temperate areas with low rainfall variability
- Long-term studies on grazing/precipitation effects on SOC in rainfall variable areas in SSA are needed
- In C4 dominated grasslands, such as those found in dry lowland areas in SSA, increasing grazing intensity may increase SOC (this hypothesis needs to be tested)
- In C3 dominated grasslands, such as those found in high altitude areas in SSA, increasing grazing may decrease SOC (this hypothesis needs to be tested)
- Precipitation, soil type and soil nutrient status interact to determine the impacts of grazing regimes on SOC

3.1.2 Grazing management and SOC

Grazing intensities are often described in subjective terms, such as heavy, moderate or low intensity. Overgrazing, a function of both grazing and recovery time, results when livestock either overgraze the plants to a point of non-recovery or access the plants before they have had time to recover (Neely et al., 2010). Grazing intensity which removes vegetation beyond the point of recovery can impact SOC. Furthermore, in dryland landscapes, heavy grazing can also increase soil erosion by leaving bare patches of soil (prone to wind and water erosion) and by compacting the soil which increases the amount and intensity of runoff (Conant and Paustian, 2002). Compaction caused by trampling reduces infiltration of water which in turn can reduce plant productivity again limiting inputs to the soil (Pineiro et al., 2010). By contrast, others (McNaughton, 1979; Hulbert, 1988) point out that in drier climatic zones, the growth of grasses that are adapted to grazing, trampling and fire, can decline in the absence of disturbance due to accumulation of senescent material, leading to loss of vegetation cover and degradation.

Conant and Paustian (2002) estimated that 10.4% of grasslands in Africa are 'overgrazed' with most being located in eastern and southern Africa. They suggested that changing overgrazed to moderately grazed land in Africa could sequester carbon (C) at a rate of 0.21 Mg C ha^{-1} yr^{-1}, although this result was based on a limited number of studies and, as discussed in the previous section, changes in grazing management are highly context specific in terms of effects on SOC. Conant and Paustian (2002) conducted a global study and therefore did not consider impacts of individual grazing management systems typical of SSA on SOC. In this report (Chapter 8) Conant presents model based estimates of C sequestration potential for native rangelands and planted pastures in SSA, but does not cover the impacts of individual management practices. A recent meta-analysis by Yayneshet and Treydte (2015) included consideration of 34 studies from SSA which considered the effects of livestock grazing on SOC. They found no differences between rangelands, commercial ranches and game reserves. However a comprehensive regional scale analysis is still lacking.

Pastoralism

An estimated 50% of land in Africa (40% in SSA) is considered to be suited uniquely for pastoralism due to limited and highly variable rainfall (Eswaran et al., 1997; IRIN, 2007). Pastoralism is defined as the use of extensive grazing on rangelands for livestock production with an estimated 50 million pastoralists and agro-pastoralists in SSA (Rass, 2006). Many pastoralists are subsistence herders often living in multiple family groups with sufficient labour to ‘track’ changes in pastures closely by moving and splitting herds quickly (Western, 1982; Niamir-Fuller, 1999). Such systems, which have developed over thousands of years, allow herders to use resources as and when they become available, giving grazing lands time to recover before return (Reid, 2012b). When systems are disrupted by restrictions or loss of movement recovery periods may also be disrupted. Appendix 2 discusses pastoralism further.

Nomadic or transhumant pastoralism results in patterns of grazing intensity and frequency which differ from those experienced in continuously grazed lands. In terms of the impacts of short intense bursts of grazing on carbon stocks the scientific evidence is mixed. Beukes and Cowling (2003) looked at the impacts of short periods of high intensity grazing on SOC in the Nama Karoo district South Africa. They found this type of grazing reduced organic carbon in the topsoil which they attributed to increased microbial activity accelerating turnover of soil organic matter. Holocheck et al. (2000) reviewed studies considering short term, high intensity grazing and concluded the system led to soil degradation with implications for SOC. However Beukes and Cowling (2003) point out that impacts on soil organic matter (SOM) and SOC probably vary with soil type. Ardo and Olson (2003) modeling carbon dynamics in semi-arid Sudan found that whilst high grazing intensity decreased SOC on grasslands it increased SOC in savannas. Conversely Mills and Fey (2003) found SOC declined in a savanna in the Eastern Cape, South Africa after intense grazing by goats.

In SSA traditional pastoral systems rely on an ability to read the landscape and adjust herd sizes and grazing frequency appropriately. This can break down when movement and resources are restricted due to changes in land tenure or sedentarisation (Reid, 2012; Niamir-Fuller, 1999). The benefit of rotational grazing systems which exert short duration, intense grazing pressure followed by adequate recovery time (grazing and rest) has been the subject of controversy for decades. The original premise behind these grazing systems arose from the insight that, historically, African landscapes were grazed by large, bunched and moving herds of ungulates that were responding to predators (McNaughton, 1997; Savory and Butterfield, 1999). Briske et al. (2011) examined the debate surrounding rotational grazing and concluded that there were no significant benefits of rotational grazing over continuous grazing. Teague et al. (2011) found that multi-paddock, adaptive management strategies for improving vegetation and animal performance were superior to continuous grazing in terms of ecosystem services, conservation and restoration. Weber and Gokhale (2011) call for transformation of the debate over specific grazing systems into a greater understanding of adaptive management processes and attention to better management.

Enclosures/Exclosures

Fencing or protecting an area to exclude grazing animals is increasingly being used in parts of SSA (particularly Ethiopia) as an adaptive strategy to provide livestock fodder in times of stress (Catley et al., 2013). These enclosures/exclosures are generally used for cut and carry hay production so, although not grazing lands themselves, they do support livestock. Impacts on carbon stocks have yet to be confirmed. Yayneshet and Treydte (2015) in their meta analysis found that studies carried out in the highlands (>1500 masl) with more than 600 mm annual rainfall had higher SOC concentration in exclosures compared to surrounding rangeland. They also found woody biomass increases in those studies which reported exclosures. The World Bank /CARE Climate Smart Initiative, being executed by Cornell University, is currently estimating the impacts of long term (20 year) exclosures on carbon stocks in Ethiopia.

Summary points

- Impacts of high intensity infrequent grazing on SOC vary in different soil and climate conditions.
- Adaptive management systems may promote good land management in general with potential co-benefits for carbon.

3.1.3 Choice of Livestock Type

In SSA cattle, goats, sheep and camels are the most common types of grazing livestock (Winrock International, 1992). Different species are suited to different ecological niches and herd diversity is an important adaptation strategy of traditional pastoral systems (IUCN, 2010). For example, small ruminants, particularly goats, can flourish in hostile environments. Choice of livestock type can also impact the environment, with cattle only systems having a higher potential for bush encroachment than mixed cattle/small ruminant systems. Goats tend to browse more from trees and shrubs causing loss of woody biomass (Woomer et al., 2004) whereas cattle graze more on herbaceous biomass and can cause more compaction leading to different impacts on carbon stocks. In eastern and southern Africa many grazing lands are also subject to grazing by wildlife. Herbivore type can influence total system carbon in several ways. Bagchi and Richie (2010) considered the impacts of native herbivores versus livestock populations on SOC in India. They found SOC to be 49% lower in areas being grazed by livestock when compared to areas being grazed by native herbivores despite grazing intensities being comparable. They attributed the difference to the different diets of the animals and the knock on effect this had on productivity and returns to the soil. They therefore suggest that matching livestock densities to those of wildlife is unlikely to be enough to prevent negative impacts on carbon stocks.

3.1.4 Grazing Impacts on Vegetation

Grazing can impact plant species composition not only through the dietary preferences of livestock but also through the ability of a species to recover after grazing. This tends to favour annual over perennial grasses. Angassa (2012) found that light to moderate grazing increased herbaceous species diversity whereas heavy grazing, without 'overgrazing', reduced it in rangelands in Borana, Ethiopia. Tefera et al. (2007) also working in Ethiopia found perennial grasses were replaced by annual grasses in heavily grazed areas, a phenomenon which did not occur in lightly grazed areas. They hypothesised that this would lead to lower SOC in heavily grazed areas as annuals tend to have shallower rooting systems and lower returns to the soil. However no difference in SOC was found which was attributed to grazing disturbance having already exceeded a threshold of degradation. Ritchie (2014) also argued that warm season C4 species, which tend to dominate in tropical grasslands, respond differently to heavy grazing than temperate C3 species. He argued that warm season C4 species invest more in belowground growth in response to intermediate grazing pressure thereby increasing C inputs to the soil and therefore SOC. This hypothesis, supported by findings from grazing trials in Serengeti National Park, form the basis of the SNAP model, which is discussed further in Chapter 8 of this report. As discussed in section 4.1.1 the findings are also supported by the meta analysis of McSherry and Ritchie (2013), which found higher grazing intensity was associated with increased SOC in grasslands dominated by C4 grasses, but with lower SOC in grasslands dominated by C3 grasses.

Planting improved grass species and legumes as a potential management strategy for carbon sequestration is unrealistic for most grazing lands in SSA due to the practical constraints of cost and scale. Despite this, studies have shown a technical potential. Macharia et al. (2011) carried out a trial in a semi-arid area in Kenya to determine the impact of legume introduction on soil fertility in natural pastures. They found a significant increase in SOC after 2 years due to the increased volume of crop residues produced by the legumes. Herbaceous legumes are not found in many grazing land in Sub-Saharan Africa, especially those in the drylands. This has been attributed to susceptibility to drought, failure to cope with heavy grazing and high cost of seeds (Thairu and Tessema, 1987). However, leguminous trees are present in many native shrublands used for silvopasture throughout SSA and more work is needed to determine the role these play in improved soil nitrogen content and carbon sequestration above and below ground.

3.2 Fire

Burning is a common management practice in many grazing lands in SSA dating back to prehistoric times (Vagen et al., 2005). Some 75% of grasslands in Africa undergo burning every year (FAO/LEAD, 2006). Burning is a widespread management practice for a variety of reasons. Burning can often release nutrients from old plants, control bush encroachment, remove unwanted herbaceous species improving forage quality, lead to increased plant growth, reduce the risk of large wildfires and reduce the density of disease vectors (Reid, 2012). Burning can also increase annual dry matter production by encouraging early growth at the start of the new season (Ojima et al., 1994). Indeed, burning is a natural and necessary phenomenon in many grass and shrublands (Bond and Keely, 2005). Fires don't typically carry in the drier savannas (e.g., the Sahel), and tree-grass competition (Dohn et al., 2013) means that fires often don't carry in the mesic (wetter) savannas either, thus most fires are concentrated in the 400-

1000 mm Mean Annual Precipitation rainfall zones. Despite widespread use of fire as a management practice, the short term benefits of burning can sometimes be outweighed by long term problems and in terms of GHG emissions and C stocks, impacts are mixed and context specific making the picture complicated.

Fire and Green House Gas (GHG) Emissions

African ecosystems currently contribute an estimated 40% of global GHG emissions from fire, mostly from savanna burning (Cias et al., 2011). The burning of any biomass releases GHGs to the atmosphere mainly CO_2, CH_4, N_2O and NO_x. For the purposes of GHG accounting, CO_2 is assumed to be taken up again after burning by the growth of new plants and therefore has a 'neutral' impact on the atmosphere if land use remains the same (IPCC, 2006). Other GHGs released include CO, CH_4, N_2O and NO_x all of which have a bigger relative impact on atmospheric warming than CO_2. Despite recent studies, estimates of emissions from burning of grasslands and savannas in SSA are still uncertain. The CarboAfrica Project, which finished in 2010, estimated GHGs from fires in different biomes in SSA including grasslands and savannas (Bombelli et al., 2009). They measured the extent of burned area, the fire intensity and the resulting emissions. The project estimated C losses from fires in savannas in SSA as 0.79 Pg C y^{-1} (similar to earlier estimates of Williams et al., 2007) this includes all GHG emissions expressed as C equivalents. When they set this against their estimate of Net Ecosystem Productivity (NEP) for savannas they concluded that the savannas of SSA are currently a net C sink. However, they stress that there are many sources of uncertainty, one of which is the impact that fire has on carbon stocks in biomass and soils under varying conditions.

Fire – Impacts on carbon in soils and biomass

Globally, less research has been done on the impacts of burning on C in soils and biomass in grazing lands than on GHG fluxes from burning. Studies from Sub-Saharan Africa are fewer still. Burning has an immediate effect on above ground C by removing biomass. However the amount removed depends on many interacting factors including vegetation type, climate and fire regime (Andreae and Merlet, 2001). In SSA many grazing lands are dry savannas with a long history of frequent burning. Hanan et al. (2008) argued that in these systems fire tends to remove young seedlings and herbaceous ground vegetation, with little immediate impact on long-term carbon stock in biomass (i.e., savanna fires do not burn wood, but do burn grass and tree seedlings). In the longer term, however, fires restrict seedling survival and thus eventually reduce adult populations and thus biomass. Changes in fire regime may therefore be associated with long-term change in carbon stocks in biomass (Bond et al., 2004). Lehmann et al. (2014) in a global analysis found a general relationship where fire reduced tree basal area, having implications for above ground C stocks. However the relationship varies depending on moisture suggesting more than one model is needed.

The impact on below ground C can be even more complex. Following a fire, C can be lost from the soil due to an alteration in soil respiration and CO_2 can be emitted from the soil due to decomposition of material killed but not fully burned in the fire (Scholes and Andreae, 2000). Conversely, fire can lead to an increase of C inputs to the soil from plants killed during the fire especially from roots (Oluwole, 2008). Fynn et al. (2003), working in South Africa on a long term (50 yr) field trial in a native grassland, showed that annual burning caused a decrease in aboveground litter inputs (0-2cm),but also led to increased turnover of root material below the surface (4 – 10cm). However, overall they only found significant change in organic C (a decrease) in the 0-2cm soil layer. Bird et al. (2000), working on another long term (52 yr) trial in the Zimbabwean savanna, also found significant SOC losses in the upper soil layer (0-5cm) after annual burning with a decrease of 20-25% compared with pre-fire levels. Oluwole et al. (2008) reported an increase in SOC in a 25 yr trial in burned versus unburned grassland plots in the 0-15 cm layer and attributed this to turnover of root material after burning. These studies highlight the importance of depth when considering fire effects on soil C and suggest a decrease in C in the surface (0-5 cm) and a potential increase below the surface (4-15 cm).

Fire can also impact SOC indirectly by changing soil chemistry, which in turn affects soil microbial activity. Following burning, inputs of ash can raise soil pH, stimulating microbial activity and increasing the amount of labile carbon which is more easily lost (Mills and Fey, 2003). The study by Oluwole et al. (2008) also found that annual burning significantly increased soil macro-fauna populations (0-30 cm) compared to plots burned every 6 years. They attributed this to an increase in dead root mass caused by burning.

Soil texture in grazing lands is an import factor when considering impacts of burning on soil carbon. Bird et al. (2000) found annual burning reduced SOC by almost twice as much (10 Mg ha^{-1}) in sandy soils compared to clay soils (5.5 Mg ha^{-1}) in savanna in Zimbabwe. At both sites, annual burning resulted in greater SOC loss than 5 yearly burning. This type of management change may, therefore, be most effective in areas with sandy soils where burning occurs annually.

Fire Regime

Bond and Keely (2005) define fire 'regime' as patterns of frequency, season, type, severity and extent of fires. There is a long history of people manipulating fire regimes to manage grazing lands. However, impacts on C, especially in SSA, are still under researched. SOC has been shown to decrease with more frequent burning (Ardo and Olsen, 2003). Du Preez (2011) reviewed the effects of land management, including fire frequency, on SOC in South Africa. They cite a 34 year study in Kruger National Park, which showed a greater decline of SOC in the 0-15 cm depth in plots burned every 1, 2 or 3 years (Jones et al., 1990). Materechera et al. (1990) found soils on which vegetation had been burned annually for a 17 year period had lower SOC at 0-40cm (0.71%) than soils which were burned every 3 years (0.83%) but that levels in soils burned every 6 years were similar to those burned annually (0.77%). This shows that the relationship between burn frequency and SOC is not a straightforward one and further investigation is needed to determine optimum burn frequency for SOC in different vegetation, soil type combinations.

In addition to frequency, timing of burning can also determine impacts on SOC and again most evidence comes from studies in South Africa. Long-term (50 yr) experiments with burning of grasslands in South Africa showed dry season burning to result in significantly higher SOC losses from the 0-2cm layer than a no burn control. Wet season burning, which results in cooler and less destructive fires, had no effect on SOC (Fynn et al., 2003). Conversely, Everson (1989) found wet season burning caused more runoff and erosion than burning during the dry season, which would presumably lead to increased loss of SOC.

Interaction of fire and grazing

Fire has an important role in the management of rangelands; herbivores (livestock and/or wild) can reduce fuel load and thus 'help' reduce seedling mortality during fire, and may even prevent fires, and thus allow increases in woody biomass. Shrub encroachment can, in some cases, be an extreme case where most grass biomass is removed and fires suppressed, and thus shrub and/or tree cover increases. But this interaction is sensitive to the mix of grazers and browsers; mixes of sheep and goats in many subsistence systems impact both grass biomass and seedling establishment/survival, such that this can be less of an issue. There are several ways in which fire and grazing can interact to potentially impact SOC. Fire is generally followed by a flush of new plant growth which is attractive to grazers, more grazers in turn provide inputs to the soil from urine and faeces but also cause compaction reducing infiltration and thereby potentially reducing biomass production. One of the rare studies to consider fire-grazing interactions and measure SOC was carried out in Burkina Faso (Savadogo et al., 2007). They found increased grazing pressure led to lower SOC at 0-10 cm on both burned and unburned plots after 12 years but found no direct interactive effect. The impact of grazing on species composition was different in burned and unburned plots which could potentially lead to changes in SOC in the longer term.

Summary points

- Use of fire in management of grazing lands involves the trade-off between leaving forage in place through the dry season to be available for livestock, and burning to promote new growth (e.g. in early dry season while there is still enough soil moisture) or to clear last years' litter ready for next rainy season, and/or to reduce seedling establishment.
- More research is needed on how fire impacts vegetation structure in different African climates and how this translates to an impact on above ground C stocks.
- Burning can reduce litter inputs to the soil and speed up decomposition after burning in the surface layers (0-5 cm), however following burning increased turnover of root material may increase SOC in layers below (4- 10cm).
- In general SOC can be said to decrease with increased frequency of burning, however the relationship is site specific and an optimum burn frequency should be sought.
- Timing of burning can determine impacts on SOC in the 0-2 cm layer
- More research is needed on the impacts of fire and grazing on above and below ground carbon and the role played by changes in species composition

3.3 Management of Woody Biomass

The majority of grazing lands in Sub-Saharan Africa have a woody component with varying impact on the net value of the land for livestock and other goods and services. Trees tend to increase herbaceous production in drier systems but decrease it in wetter regions (Dohn et al., 2013), while trees can provide sources of fuel, forage and other products of benefit to local communities (Tredennick and Hanan, 2015). Management strategies range from clearing (with or without burning) to prevent the formation of thickets, to planting of shrubs and trees to provide shelter and browsing options in low rainfall times. Estimated impacts on carbon stocks in biomass and soils vary. In Senegal, Woomer et al. (2004) estimated 0.77 t C ha^{-1} yr^{-1} could be sequestered over 20 years by restoring degraded grasslands to woody grasslands. They also propose that each percent increase in woody canopy cover results in 800 kg C ha^{-1} in woody biomass. Gray and Bond (2009) looked at thicket invasion and effect on C stocks in AGB and soil in South Africa. They found total carbon was highest in forests (485 Mg C ha^{-1}), second highest in thickets (397Mg C ha^{-1}) and lowest in grasslands (283Mg C ha^{-1}). Clearing shrubs and trees to produce grasslands is a management strategy common in the US and Europe. Whilst this may be suited to high input systems in high rainfall areas it is impractical in many of the low input low rainfall grazing lands of Sub-Saharan Africa.

Summary point

- From a carbon sequestration point of a view, research is needed to determine optimum livestock productivity and optimum woody coverage in conjunction with each other.

3.4 Water Management

Most of Sub-Saharan Africa's grasslands are located in arid and semi-arid areas, where water scarcity makes livestock production the only viable livelihood option. In areas dependent on the seasonal availability of water, livestock movement is often driven by water. Water management options include those that increase availability of drinking water for livestock and increase water availability for growth of grazing species. Both can influence carbon stocks and GHG emissions. In Africa alone, more than 300 million people live under severe water restrictions (Bennett, 2000), the majority of whom rely on grazing lands to support them. Providing a reliable source of drinking water for people and their livestock by accessing ground water where appropriate or providing rainwater harvesting tanks can provide relief. However, areas immediately surrounding water sources can suffer degradation due to trampling with potential impacts for SOC. In addition water recharge capacity needs to be considered in water scarce areas.

Approximately 4% of the cultivated area in Sub-Saharan Africa is irrigated, and this is generally confined to high production prime agricultural land (NEPAD, 2002). Constraints of cost and lack of water resources make it impractical to consider the use of irrigation as a means of increasing C sequestration in grazing lands in Sub-Saharan Africa. However, water harvesting techniques such as bunds (semicircular, L shaped or other) or micro-catchments have been shown to increase forage production and therefore have potential to increase both above and below ground C in areas with erratic rainfall (IEM, 2009). However, water harvesting in grazing lands is not widespread in Sub-Saharan Africa and its potential is therefore under researched (Kay, 2001). Previous attempts to promote water harvesting in croplands in Sub-Saharan Africa have had limited uptake due to high labor requirements and lack of consideration for the cultural context in which they were being placed. In addition most of the current models/methods for carbon accounting do not cater for this kind of intervention which is likely to have multiple highly localized impacts.

Water management is also important in high rainfall, high altitude areas such as the highlands of Ethiopia where grazing lands can lose carbon through soil erosion. Preventing erosion by controlling runoff can increase biomass, thus increasing inputs to the soil and therefore has the potential to maintain soil carbon.

Summary points

- Low tech water harvesting techniques such as bunds have potential to increase productivity and potential C stocks in grazing lands. Techniques to assess their impact on C stocks and constraints to their adoption need to be developed.
- The role of payments for multiple ecosystems services such as water and carbon need to be considered in grazing lands in SSA.

Chapter 4. Measuring rangeland health and soil carbon in Africa

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4.1 Background

Monitoring of natural resources has become increasingly important over the last five decades and this trend is likely to continue (de Gruijter et al., 2006). This is largely due to an increasing awareness of the negative impact that human pressure on natural resources has on ecosystem health and human wellbeing (de Gruijter et al., 2006; Herrick et al., 2013). The current trends observed in Africa of increasing land degradation and deterioration in rangeland health negatively impact livestock, wildlife and livelihoods. Good rangeland health information will inform sound decision making on livestock futures (size of herds, alternate types of livestock) at different strategic and institutional levels. Due to a lack of consistency in protocols to measure and monitor rangeland health, it is difficult to conduct systematic cross-regional analyses of the factors affecting rangeland productivity. Moreover, existing information is often qualitative, seasonal and/or site specific. The general lack of standardized methods for collection and analysis of information on rangeland condition and associated risk factors has hindered critical decision making in the drylands of Africa on the design and implementation of sound rehabilitation programmes that benefit livelihoods and the environment.

More specifically, there is increasing need for improved monitoring data on soil carbon. With the increasing threat of climate change to ecosystem functioning, soil carbon sequestration is recognized as an important strategy to mitigate the adverse impacts of climate change. African soils are important carbon reservoirs in which the top 100 cm constitutes about 68% of the terrestrial carbon pool of Africa (Henry et al., 2009). One major outcome of the Bali convention was the development of a framework for measuring, reporting and verification (MRV) of climate mitigation actions through Nationally Appropriate Mitigation Actions (NAMAs). MRV provides opportunities for developing countries to claim financial, technical and capacity building support from developed countries to implement their NAMAs. A number of carbon measurement schemes are emerging for specific applications. However, the lack of robust, rapid and cost-effective methods of soil carbon measurement and monitoring is a constraint for implementing MRV (Ellis and Larsen, 2008; Bellon-Maurel and McBatney, 2011; Conant et al., 2013). The aim of this chapter is to review existing cost effective methods of monitoring rangeland health and soil carbon stocks.

4.2 Measuring soil organic carbon

The UNFCCC has adopted a three-tier approach for estimating carbon benefits to allow for increasing level of effort and precision as appropriate or economically viable (IPCC, 2003). Tier 1 uses default parameters or globally available data sources to estimate carbon stock changes while Tier 2 includes available regional or national data. Tier 3 requires detailed field measurements and modeling efforts to quantify carbon stocks. In Tier 3, measuring soil carbon often demands more sampling efforts than Tier 1 and Tier 2 due to high inherent variability of soil properties. This could lead to higher measuring costs, which may exceed the benefits expected from the increase in carbon stocks (IPCC, 2003). Therefore, the development of locally calibrated models that can use easily collected data, minimizing the cost of demonstrating a change in soil organic carbon stocks (IPCC, 2003), is a research priority.

The IPCC Good Practice Guidance for Land Use, Land-Use Change and Forestry (GPG-LULUCF) (IPCC, 2003), the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) and The Voluntary Carbon Standard (VCS) are the widely referred carbon accounting methods (Bird et al., 2010).

Decision makers are interested in monitoring land and soil health more generally, not only soil carbon, to guide intervention planning and management. Soil carbon stock changes usually need to be interpreted using additional land and soil information. Furthermore other ecosystem service benefits of soil may have a higher value than soil carbon sequestration alone. The World Agroforestry Centre (ICRAF) and partners have proposed a land health surveillance and response framework, which is modelled on scientific principles in public health surveillance, to increase rigour in land and soil health measurement and management. The key objectives are to: (i) identify land health problems, (ii) establish quantitative objectives for land health promotion, (iii) provide information for the design and planning of land management intervention programmes and resource allocation priorities, (iv) determine the impact of specific interventions, and (v) identify research, service and training needs for different stakeholder groups (UNEP, 2012; Shepherd et al., 2015). Measurement protocols to implement the framework have been developed over a number of years through various projects in Africa and are currently being refined in the context of the Africa Soils Information Service (AfSIS: www.africasoils.net) and the Carbon Benefits Project: Modeling, Measurement and Monitoring, funded by the Global Environment Facility (GEF) of the United Nations Environment Program (UNEP) (www.unep.org/climatechange/carbon-benefits).

There are a number of components to consider in a soil carbon or soil health measurement and monitoring protocol. These include sampling framework, field sampling and observations, laboratory analytical methods, change detection and costs.

4.2.1 Sampling framework

Sampling frameworks are designed to reduce sampling error, avoid biased selection of sampling sites and guide where and how many samples to take. It is vital to be clear on the decision or sets of decisions that the measurement will support before designing the sampling framework, bearing in mind that the main purpose of measurement is to reduce decision uncertainty. Hence, a very different sampling framework will be required if the decision is whether to certify a carbon stock change for carbon trading purposes with a 10% accuracy with a 90% confidence level, or to validate whether current soil carbon levels for Tier 2 estimates are within an expected range.

Randomizing sites within the target area or sampling strata is important to provide unbiased estimates of carbon stocks and other land health indicators within a stratum and allow inference to be made to the whole area. Most existing soil survey data does not meet this criterion. Providing unbiased data on the statistical distribution of variables is not only useful for reporting the prevalence of land health problems (e.g., low carbon stocks) but also provides a means of setting local reference values (defining what is low, moderate or high), which can in turn be conditioned on various factors (e.g., soil texture). A small probability sample generally provides much more useful information than a large biased sample.

Stratifying the area in terms of factors that influence carbon stocks will normally reduce errors in project-scale estimates of carbon stocks. At a continental level, climate trends explain more variation in existing soil organic carbon stocks than any other single factor (Wynn et al., 2006) but locally, historic land use often has a dominant influence, and this may not be well reflected by current land use. Where soil carbon changes are of primary interest, stratification may be best done on the basis of expected land use change (e.g., woodland margins may be more likely to show rapid changes than woodland interiors or permanently cultivated land, and sampling intensity may be increased in a woodland margins stratum). Stratifying on too many variables can rapidly become un-manageable in terms of the number of strata produced and so it is advisable to restrict them to include only a few dominant factors. In some cases there may be a target sub-population in an area, such as cultivated land, in which case the sub-population needs to be clearly defined and located first.

4.2.2 Field measurements

The importance of having a consistent field measurement protocol that can be applied under all expected conditions cannot be over-emphasized. This protocol can be supplemented by local and community based measures, but local measures do not provide a substitute for a consistent protocol for high level decision making. The Land Degradation Surveillance Framework is one example that was developed for project to continental land health surveillance in Africa (UNEP, 2012; Vågen et al., 2013), and has been applied at continental scale in the Africa Soil Information Service (AfSIS 2014), at national scale by the Ethiopia Soil Information System (EthioSIS, 2014), and at landscape scale (Waswa et al., 2013). It is also being deployed by the CGIAR in a network of long-term monitoring sites across the tropics (sentinel landscapes). The approach has been applied in assessing impacts of land management interventions such as the effects of livestock exclusions and of fire on rangeland carbon stocks in Ethiopia and Burkina Faso (Aynekulu et al., 2014). Examples of other protocols include rangeland monitoring (Herrick et al., 2013), LandPKS (Herrick et al., 2013) and Vital Signs (2014).

Such protocols can also be integrated into study designs (Shepherd et al., 2015) to provide evidence of the impact of sustainable land management interventions on soil health and soil carbon stocks, and in long-term household/farm monitoring studies, such as the World Bank Living Standards Measurement Study (Aynekulu and Shepherd 2013).

Measurement of soil organic carbon (SOC) stocks requires that not only the concentration of soil organic carbon in soil fines (< 2 mm fraction) is measured but also the bulk density (or cumulative soil mass) and quantity of course fragments. Tracking changes in soil carbon stocks over time requires that the same equivalent mass of soil be measured from one monitoring event to another. Estimates of soil carbon stocks to a fixed depth using a single depth bulk density are often biased (Lee et al., 2009). Murty et al. (2002), for instance, found that the impact of conversion of forests into cultivated lands on the changes in soil carbon stock was often inflated due to the confounding influence of changes in bulk density. Therefore, it is necessary to consider corrections for spatial and temporal variation in bulk density in quantifying SOC stocks along a soil profile (Lee et al., 2009). Here we propose the use of equivalent soil mass instead of using a fixed soil depth in measuring temporal variation of soil carbon stocks. The Land Degradation Surveillance Framework (LDSF), which was used in the Africa Soil Information Service (AfSIS, 2014), also used the equivalent soil mass method instead of bulk density at specific depth to calculate temporal variation of soil carbon stocks (Aynekulu et al., 2011). Other considerations concerning field measurements are accuracy of georeferencing, consistent use of sampling equipment, and soil preparation procedures used to provide soil fines, including sub-sampling methods.

4.2.3. Analytical/laboratory methods

Soil organic carbon concentration is measured on soil fines on a tiny fraction of soil relative to the mass of soil in the field samples. Therefore laboratory sub-sampling methods to obtain subs-samples for analysis that are representative of the bulk soil (e.g., use of cone and quartering procedures or riffle boxes) are important. Two widely used analytical methods to measure soil carbon concentration are the Walkley-Black procedure (Walkley and Black, 1934) and thermal oxidation (Skjemstad and Baldock, 2008). The Walkley-Black method is often criticised for under estimation of the soil organic carbon due to incomplete oxidation of soils with higher (> 2%) organic carbon (Donovan, 2013). The dry combustion or thermal oxidation is the most commonly recommended reference test for soil carbon, and is often cheaper than other Walkley and Black methods (Donovan, 2013). To remove the influence of inorganic carbon (carbonate) (McCarty et al., 2002), soil carbon is determined on acidified samples, i.e., fumigated or treated with hydrochloric acid to remove inorganic carbon (Harris et al., 2001). Standard operating procedures used by the Africa Soil Information Service are available (AfSIS, 2014).

It is difficult to get reproducible results among different labs. Even within the same lab using the same methods, reproducibility can be a problem providing a key source of error when monitoring soil carbon over time, especially when considering low-carbon soils. Furthermore, comparisons of data between labs are intensely problematic. It is possible that the cheapest way to measure changes in soil carbon with any of these methods is to collect, store, and then analyze a time series of samples from the same soil (to determine any limitations resulting from soil sample preparation and processing), in the same lab, at the same time (Palmer et al., 2002).

Soil infrared spectroscopy (IR) is another emerging technology that makes large area sampling and analysis of soil health feasible (AfSIS, 2014; Shepherd and Walsh, 2007). It overcomes the current impediments of high spatial variability of soil properties and high analytical costs, which are key challenges in monitoring soil health at the landscape scale (Conant et al., 2011). A review by Bellon-Maurel and McBatney (2011) showed an exponential increase in the use of near infrared (NIR) and mid infrared (MIR) reflectance spectroscopy for soil analysis. MIR is more reproducible and robust in measuring soil carbon than NIR (Rossel et al., 2006; Bellon-Maurel and McBatney, 2011; Nocita et al., 2014). IR is now routinely used for analyses of a wide range of materials in laboratory and process control applications in agriculture, food and feed technology, geology and biomedicine (Shepherd and Walsh, 2007).

IR enables soil-sampling density (samples per unit area) to be greatly increased with little increase in analytical costs. This reduces errors in quantifying soil carbon due to soil spatial heterogeneity. Some researchers have shown the potential application of IR to measure soil carbon onsite (Reeves III, 2010) as well as bulk density (Moreira et al., 2009), however accuracy is low. IR data can be integrated with geostatistical data (Cobo et al., 2010), remote sensing data and topographic information (Croft et al., 2012; Huang et al., 2007; Vågen and Winowiecki, 2013) for digital soil mapping at the landscape level (UNEP, 2012). For instance, Rossel et al. (2014) used IR data to develop a soil carbon map of Australia. Infrared spectroscopy data was also used in mapping soil carbon in Kenyan rangelands (Vågen et al., 2012). The Soil-Plant Spectral Diagnostics Laboratory of ICRAF supports a network of IR instruments in 10 counties in Africa, which can be used to analyse soil carbon. AfSIS is deploying centralised calibration development and an online spectral prediction application to reduce the need for each laboratory to develop its own calibrations. A number of other fundamental soil properties, in addition to soil organic carbon, can be simultaneously predicted (AfSIS 2015a).

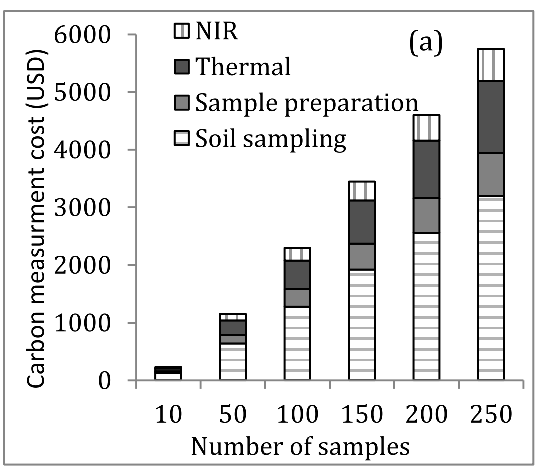
4.2.4 Monitoring soil organic carbon stocks

To understand the impact of projects in terms of capturing carbon, changes in SOC stock need to be measured over time. Soil monitoring assesses the changes in soil carbon status with reference to the soil carbon stock at the beginning of the project. The Marrakesh Accords specify that all emissions from sources and removal by sinks should be reported annually (IPCC, 2003). However, the inter-annual variability in SOC stock is often very low. Moreover, the cost of detecting a change in SOC stock using field and laboratory measurements is expensive. Hence, costs associated with detecting SOC changes might amount to more than the actual value of carbon sequestered, although soil-monitoring schemes may serve a number of other purposes.

Even though the change in SOC stock varies with factors that influence the rate of production and decomposition of carbon, UNFCCC (2006) recommend a monitoring interval of between 10 and 20 years. Short-term projects should not be expected to show a measurable increase in SOC. Therefore, projects should use well established and valid models to predict SOC stock changes over a longer time period based on land use and management practices. SOC monitoring at fine temporal resolution can be also achieved using a digital soil mapping approach where SOC is predicted using covariates such as terrain attributes e.g. topographic wetness index, spectral reflectance bands from satellite imagery, land cover and natural vegetation, and parent material (McBratney et al., 2003). Optimal sampling designs for repeated sampling also need to be established.

4.3 Can we measure soil carbon cost effectively?

The cost of measuring SOC largely depends on the number of samples, field accessibility, and laboratory analysis. The number of samples to be taken in a project depends on the level of variability in soil organic carbon in the target area, the required levels of precision and resource availability. In some cases the cost of demonstrating the change in carbon stocks in soils to the required accuracy and precision may exceed the benefits that accrue from the increase in stocks (IPCC, 2003). Although infrared spectroscopy significantly reduces the analytical cost of measuring soil carbon, costs incurred in soil sampling and preparation still form the largest component cost (Figure 2) (Aynekulu et al., 2011). Another major advantage of IR for soil analysis is the capacity to predict many soil properties from a single spectrum simultaneously (Rossel et al., 2006). This can further reduce the total analytical cost of soil analyses. A third big advantage of IR technology is the high throughput which is critical for carbon inventories at project level or larger geographical extents. The daily throughput of a thermal analyser is quite low (ca 50 samples) whereas it is possible to analyse about 1000 samples per day with robotic MIR systems.



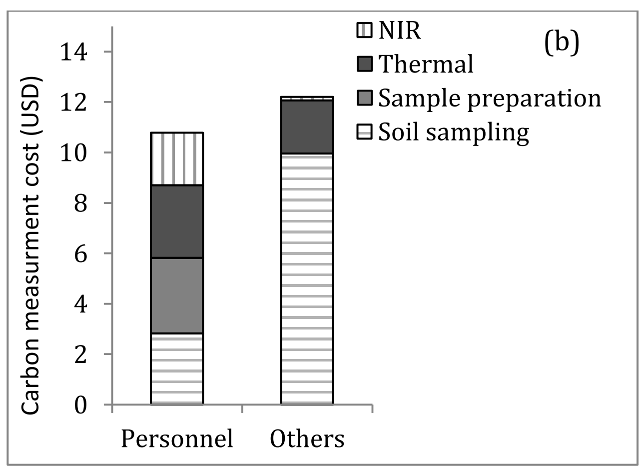


Figure 2. The cost of measuring SOC stocks. Costs (a) and cost structure (b) of measuring one soil sample. The personnel costs are more or less uniform for the four major activities, while soil sampling constitutes a large proportion of the other costs

4.4 Conclusions

A number of carbon measurement schemes are emerging for specific applications. However, it is important to first define the decisions that carbon measurements are being designed to inform, to define the critical decision uncertainties, and to establish measurement priorities. Although soil carbon is also a key soil quality indicator, it alone does not provide sufficient information to guide wise use of land resources, and therefore standalone carbon measurement systems will have limited value, especially given the large resources required for field sampling and soil preparation. In most cases it will be most efficient to embed carbon measurement within broader land health surveillance schemes.

Soil carbon modeling initiatives should take some soil carbon measurements to provide validation of soil carbon initial levels and changes. This is being made easier with the development of soil infrared spectroscopy as an analytical tool, which is reducing the cost, and increasing the speed and reliability of quantifying soil carbon and other soil properties simultaneously. The growing network of soil spectroscopy laboratories and capacity will be able to support soil carbon measurement in sustainable land management projects and national soil health surveillance systems in Africa. However, more work needs to be done to express the propagation of uncertainties and errors throughout the measurement or modeling processes in relation to the decisions that need to be taken – this will help focus further research on areas where uncertainty reduction will most improve decision outcomes.

Chapter 5. Map-based estimates of present carbon stocks of grazing lands in Sub-Sahara Africa

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Continental scale data sets for livestock production zones, climate, and soil types were analyzed using a GIS-based mapping approach (Appendix 3) to assess regional differences in soil organic carbon (SOC) reserves for Africa. Mean SOC stocks to 0.3 m, 0.5 m and 1 m depth for the continent are estimated at 83, 113 and 162 Pg C (Pg = 10^{15} g). This is somewhat less than earlier estimates of 86-96 Pg C (0-0.3 m) and 133-184 Pg C (0-1 m) that were based on coarser resolution soil-geographic databases and a more limited set of soil profiles (Batjes, 2004; Henry et al., 2009). As indicated by Henry et al. (2009), use of different soil profile databases, spatial datasets and/or mapping approaches can easily lead to differences of up to 30% in the estimates for SOC stocks for Africa.

For Sub-Sahara Africa (SSA), mean SOC stocks are estimated at 72 (0-0.3 m), 98 (0-0.5 m) and 140 (0-1 m) Pg C. This corresponds to a mean, area-weighted SOC content of 3.3 kg C m^{-2} to 0.3 m depth, 4.4 kg C m^{-2} to 0.5 m and of 6.3 kg C m^{-2} to 1 m depth; coefficients of variation are often in excess of 50%. The area-weighted, mean SOC content in the topsoil (0-0.3 m) ranges from almost nil, for a mapping unit that is comprised of coarse textured, mineral soils in a warm arid climate, up to some 61 kg C m^{-2} for a (small) mapping unit of organic soils (Histosols) mapped in the Ethiopian highlands (Figure 3). This broad range reflects the effect of regional differences in natural vegetation or land use, climate and soil type in determining the mean SOC content in a given region.

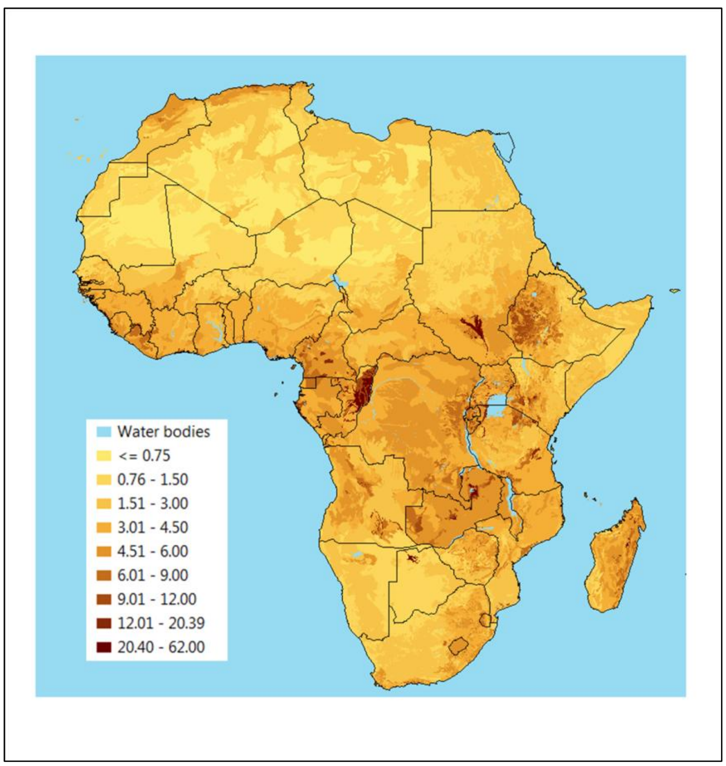

Figure 3. Distribution of soil organic carbon (kg C m^{-2} to 0.3 m depth) for Africa.

Estimates for mean SOC content, as observed in different livestock production systems (GLPS, see Robinson et al., 2011) in SSA, are listed in Table 3. In short, the system considers Rangeland (LG) and Mixed (M) land cover systems. The latter are comprised of croplands and rangelands; mixed irrigated systems (MI) are defined as being >10% irrigated versus <10% for rainfed systems (MR).

Table 3. Mean soil organic carbon content for main livestock production systems mapped for Sub-Sahara Africa, subdivided according to broad climate classes

Livestock production system[a]	Climate	SOC content ($kg\ C\ m^{-2}$)			Area	
		0-0.3m	0-1m	0-2m	$10^6\ km^2$	% [c]
LG - Rangeland systems	Hyperarid	1.0	1.7	2.7	1.0	5.4
	Arid	2.2	4.4	6.2	9.2	49.7
	Humid	4.1	7.8	10.6	2.2	12.0
	Temperate	4.5	8.0	10.6	0.2	1.2
	Undiff.	*2.5*	*4.8*	*6.8*	*12.6*	*68.3*
MR - Mixed rainfed systems.	Hyperarid	-	-	-	- [b]	-
	Arid	3.0	5.8	8.2	3.1	16.9
	Humid	4.5	8.6	11.9	2.0	11.0
	Temperate	5.7	10.6	14.3	0.7	3.5
	Undiff.	*3.8*	*7.3*	*10.2*	*5.8*	*31.4*
MI -Mixed irrigated systems	Hyperarid	-	-	-	-	-
	Arid	3.5	7.6	11.3	0.04	0.2
	Humid	5.4	11.1	16.7	0.01	0.1
	Temperate	6.0	11.8	16.4	0.01	0.1
	Undiff.	*4.2*	*8.7*	*12.8*	*0.06*	*0.4*

[a] Livestock production systems are defined according to Robinson et al. (2011) ; [b] SOC data not shown where the total area for a given livestock production zone is estimated at < $0.01 \times 10^6\ km^2$. [c] Total may differ from 100% due to rounding.

Overall, as a reflection of regional differences in the type and intensity of the soil forming factors of climate, parent material, relief, organisms (notably humans) and time (Jenny, 1941), and main drivers of recent soil change (e.g., land use and climate change), for a given livestock production system, the area-weighted SOC content increases as annual rainfall increases, and mean air temperature decreases. For example, for the topsoil (0-0.3 m), within rangelands (LG), the area-weighted mean SOC content is 1.0 kg C m^{-2} in the Hyperarid zone, 2.2 kg C m^{-2} in the Arid zone, and 4.1 kg C m^{-2} in the Humid zone (defined here as having a length of the growing period > 180 days and average air temperature (T_{av}) over 20 ^oC during the growing period), and 4.5 kg C m^{-2} in the Temperate zone (T_{av} between 5 and 20 ^oC or at least 1 month with T_{av} < 5 ^oC). This broad climatic pattern is also observed for the Mixed (M) systems; the area-weighted mean SOC content for irrigated (MI) systems being somewhat greater than that estimated for rainfed (MR) systems.

In accord with their definition, each of the above livestock production zones can cover a fairly broad range of landscapes, land cover types, climate zones and soil types. With respect to soil type, regional differences in soil drainage, soil texture, soil mineralogy (e.g., low activity clay (LAC) versus high activity clay (HAC) soils), and soil nutrient status are particularly important in determining mean regional SOC levels in a given livestock production system; this aspect is further discussed for the Rangelands (LG) class in Appendix 3.

Overall, for all livestock production systems, the mean proportion of the total SOC content in the upper 0.3 m to that held up to 1 m depth is 51%, with a range of 33% to 78% (min-max). When expressed as a ratio of SOC content to 2 m depth, the mean is 36% (17% - 70%). These figures are illustrative of the fact that changes in land use and management practices may result in significant losses of SOC, with concomitant emissions of CO_2 to the atmosphere, as well as a deterioration of soil health and human livelihood associated with a loss of soil organic matter content unless soils are judiciously managed (Bationo et al., 2007; Banwart et al., 2015; Milne et al., 2015).

Broad effects of soil type (e.g., drainage, clay mineralogy, texture, and soil nutrient status) on SOC levels can be discerned from the present exploratory analyses and such effects should be taken into consideration in any analyses of SOC dynamics. As indicated, these are a reflection of the regionally varying impacts of the soil forming factors of time, relief, climate, organisms (notably humans) and parent material (Jenny, 1941; Gray et al., 2011). Alternatively, effects of socio-economically and biophysically driven changes in land use and management on SOC content, including the occurrence of natural and man-made fires (Govender et al., 2006; Du Preez et al., 2011), within the various livestock production systems, cannot be distilled from the present continental-scale data sets. This is partly a reflection of the fact that effects of land use and management interventions on SOC dynamics largely occur at the local scale (Izac, 1997; Banwart et al., 2014). High resolution soil property maps, derived from digital soil mapping (e.g., Vågen and Leigh, 2013; Hengl et al., 2014), have very recently become available for Sub-Sahara Africa (see AfSIS, 2015a). In principle, these could be used to consider such landscape scale effects.

Possible changes in SOC content, and net CO_{2eq} emissions, over time, subject to defined changes in land use and management practices, as well as socio-economic conditions/incentives, can best be assessed using appropriately-scaled and regionally validated modeling tools (Conant et al., 2010; Bernoux et al., 2011; Milne et al., 2012), supported by long-term monitoring systems (de Brogniez et al., 2011; van Wesemael et al., 2011). Several of these aspects are discussed elsewhere in this report (Chapters 4 and 6).

Chapter 6 Modeling soil carbon in African grazing lands

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6.1 The Need for Soil Carbon Models

Africa accounts for 20% of the total land surface of the earth, making its carbon dynamics particularly relevant globally, however, these dynamics are poorly understood (Williams et al., 2007; Ciais et al., 2009). Landscapes in Africa are changing in ways that could change the region's capacity to store carbon, for example deforestation and reforestation (Higgins and Scheiter, 2012; West, 2012), the loss of wetlands (Milne and Brown, 1997), and agricultural expansion often at the expense of grazing lands (Olsen et al., 2004). More generally, Africa may be poised to experience more vegetation changes than other regions of the world due to the commonality of C4 plants (West et al., 2012). C4 grasses, and the fires they support, have played a major role in preventing trees from establishing and have led to the creation of vast areas of savanna (Sankaran et al., 2008). Due to the fact that species with a C4 photosynthetic pathway have a competitive advantage in lower CO_2 environments, increases in atmospheric CO_2 could provide advantages to C3 woody plants. That may cause large shifts in forest cover, which in turn could change soil carbon balances, although increasing temperatures and reduced precipitation make outcomes uncertain (Nie et al., 1992; Still et al., 2003). C4 species also invest more in rhizomatous storage than C3 species, which can allow C4 plants to compensate for stem loss and maintain leaf area when grazed (Ritchie, 2014). Compensatory responses to grazing have therefore been seen in Africa more than elsewhere. Other things that affect soil carbon storage in Africa include distinct soil parent material, phosphorous that may limit plant growth, the commonness of termites and dung beetles, and pervasive burning of vegetation biomass by residents. Some of these factors are not well represented in soil carbon models that may be used in Africa. This is due to a lack of observational data from the region and because the majority of models have been developed and parametrized for temperate conditions (Paustian et al., 1997a; Williams et al., 2007).

6.2 Current Modeling Approaches and Scope

Decades of development have yielded soil carbon models that span a range of spatial and temporal scales. Levels of complexity are equally diverse. Manzoni and Porporato (2009) reviewed ~250 biogeochemical models, and classified their theoretical and mathematical frameworks. Here, models that simulate discrete time are a focus and we emphasize models that are relevant at landscape and broader scales. It is at these scales that decisions about land management and policies relevant to grazing lands are made. The number of such models that have been produced is striking. Some models are point-based, such as CENTURY, where results from many point-based simulations are combined to yield an outcome for a region (GEFSOC, Easter et al., 2007), other models are relevant to landscapes (e.g., SAVANNA, Coughenour 1992, expanded upon below; SNAP, Ritchie 2014) and broader scales (e.g., G-Range, Boone et al., 2011a, 2013). Model structures are equally varied. For example, the RothC, SOMA, and SOCRATES models use five carbon pools, whereas CENTURY uses three pools plus aboveground and belowground plant residual pools (Lardy et al., 2011). Most models are modular, with components that may be turned on and off depending upon the needs for an application.

6.3 Examples of Application of Carbon Models to Grazing Lands in Africa

The CENTURY model is a dynamic ecosystem model that simulates the turnover of soil organic matter and plant nutrients (Parton et al., 1987). It was originally developed to simulate conditions in the North American Great Plains grasslands but has since been adapted and applied to many ecosystems worldwide, including savannas. Examples of application to grazing lands in Africa do exist but are few. Woomer et al. (2004) parameterized and applied CENTURY to grasslands with varying amounts of woody cover in Senegal. They used the analysis to estimate that restoring degraded grasslands to woody grasslands in Senegal over a 20 year period could sequester 0.77 t C ha^{1} yr^{1}. Ardö and Olsson (2003) linked CENTURY to a spatial database to make estimates of carbon sequestration in a region in semi-arid Sudan. This region included grasslands and savannas for which they simulated grazing intensity and fire regimes. Working at the national scale, Kamoni et al. (2007) also used the GEFSOC System (Easter et al., 2007; Milne et al., 2007) which links the CENTURY and the RothC models to a GIS to estimate C stock changes for Kenya. They simulated grasslands, savannas, and shrub/grasslands as a tropical grassland dominated by C4 grasses. For large scale applications of any model to African grazing lands, performance of the model is constrained by the availability of data to both parameterize and run the model. Likewise, long term data sets are ideally used for model validation, which is usually another constraint.

Michael Coughenour created the SAVANNA ecosystem model working in the Turkana District, Kenya (Coughenour, 1992). There have been many subsequent improvements and applications of the model, and it has been applied to systems world-wide (e.g., Boone, 2007; Boone et al., 2002, 2011b; Galvin et al., 2004; Thornton et al., 2004, 2007; reviewed in Ellis and Coughenour, 1998). SAVANNA models primary ecosystem interactions in arid and semi-arid landscapes, simulating functional groups of plants and animals over periods from 10 to 100 or more years using a weekly time-step (Ellis and Coughenour, 1998). SAVANNA has most often been used to assess forage availability for wildlife and livestock, but changes in soil carbon are well-tracked in SAVANNA, using routines adapted from CENTURY, and changes in plant biomass have been reported. For example, Boone (2005) used SAVANNA to assess the means in which vegetation changed under increasing levels of landscape fragmentation in South Africa, Tanzania, and Kenya. Other treatments have linked SAVANNA to population-based (Thornton et al., 2003) and agent-based (Boone et al., 2011b) household models. This coupled-systems approach would allow changes in soil carbon to be tracked in parallel with changes in ecosystem services provided to livestock owners, and the ways in which decision making by livestock owners may influence soil carbon stocks.

The Rothamsted Carbon Model (i.e., RothC; Coleman et al., 1997) is a simple and popular choice for analyses of carbon dynamics. It is unusual in this set in that it does not include procedures to model plant production or the annual contributions of carbon to soil. Instead, it tracks the fates of organic carbon in soil, or can infer what organic inputs must have been to yield observed soil organic carbon. Examples of the use of RothC in Africa include Smith et al. (2014), who compared the carbon balance of biogas digestion, composting, and biochar in Sub-Saharan Africa, its use in weighing the benefits of fertilization of West African savanna crops (Diels et al., 2001), as one of three methods used by Kamoni et al. (2007) to predict future soil organic stocks in Kenya, and to judge carbon turnover for coppicing fallows in eastern Zambia (Kaonga and Coleman, 2008).

Boone et al. (2011a, 2013) created the G-Range model, which simulates biogeochemical and plant functional group changes in global rangelands. At the core of G-Range is a streamlined representation of the CENTURY model (Parton et al., 1987). They created an application that was spatially explicit, and represents the rangelands of the world in a single simulation. Herbs, shrubs, and trees are simulated in a dynamic way. Parameters are provided for landscape units to model vegetation competition for water, light, nutrients, and space, as well as livestock offtake, the probability of fire, etc. The time-step for G-Range is one month, and simulations may be 10s to 100s of years. More than 100 spatial surfaces are produced each time step. Products using G-Range are in development, but simulation results for Africa have been used in planning documents (e.g., Denish et al., 2015).

Mark Ritchie developed the SNAP model (Ritchie, 2014), which explicitly incorporates plant compensation to grazing at an annual time-step, especially in regard to changes in leaf area index – in field studies, leaf area index showed only minor declines to increased grazing, until grazing intensity reached 70%. Ritchie developed the model using the ample observed data and literature from Serengeti National Park. Once fitted to observed values, SNAP was used to track the means in which soil carbon varied with different plant lignin and cellulose contents, soil sand fraction, and rainfall. The model has been adapted to the Northern Rangelands Trust conservancies of northern Kenya (Ritchie 2014a).

From the African Carbon Exchange project (Christopher Williams, Niall Hanan, and others) came the ACE model. That model focused on carbon dynamics and the ways in which disturbances may alter land cover. The ACE model used an annual time step to track long-term changes in landscape dynamics. From that project followed the model called TGVM (Tree-Grass Vegetation Model). TGVM simulates the growth and demographics of grasses and trees based on climate, disturbance, and management. Researchers are using TGVM to address questions of hydrology, lake persistence, and pastoral dynamics in Mali, and the model may be used in carbon assessments as well.

The DNDC (DeNitrification DeComposition) model has been used all over the world to simulate N_20, CO_2 and N_2 emissions from agricultural soils (Li et al., 1992). DNDC content was made available to researchers early in its development, and they modified the model to apply to many questions, conditions, and regions (Gilhespy et al., 2014 provides a type of model 'family tree'). For example, the model now may be used to predict methane and ammonia emissions, emissions from specific crops such as rice, and modeling of greenhouse gas emissions customized for Europe. The use of the model in Africa appears more limited than the longevity of the model would suggest. Grote et al. (2009) provides an example, where they used DNDC and joined that to a land-surface model to yield a tool that simulated climate chemistry and biogeochemical processes in a coupled way, applied to a reserve in Burkina Faso.

6.4 Predictive Capacity and Uncertainties in Soil Carbon Models

The primary processes affecting the cycling of carbon in ecosystems appear to be well understood at least at the plot to watershed scale. Existing soil carbon models can yield surfaces that align relatively closely with observed values, and comparisons among modeling results are promising (e.g., Smith et al., 1997) but with sometimes high uncertainties (Ciais et al., 2011). Examples include the results from CENTURY (Conant et al., 2001; Conant et al., 2002; Woomer et al., 2004) and G-Range (Figure 4). Future projections can have large uncertainties, but many applications include long so-called spin-up times, where ecosystem dynamics are simulated as stable over thousands of years prior to the period of assessment, which builds confidence in the validity of future projections.

Sources of uncertainty in modeling can include errors due to the inability of the model to fully describe a process (usually due to lack of information about that process) and errors in input data caused by measurement errors, erroneous definitions and unrepresentative sampling. The importance of these errors varies depending on the scale at which the model has been designed to work, the system it has been designed to represent and the situation to which it is being applied.

Model comparisons have been carried out. These include ensemble results (e.g., the TRENDY collection of eight dynamical global vegetation models; Sitch et al., 2013), where mean responses from multiple models incorporate more relationships than any one model. More explicit comparisons between models have been done, and help to identify both the range of variation expected in results and the structural differences in models that may lead to differences in outcomes (Huntzinger et al., 2013). For example, Weber et al. (2009) used remotely sensed data to assess carbon balance results from the large scale (global to continental) models ORCHIDEE, LPJ-DGVM, LPJ-Guess, and JULES models, with simulations done using standardized protocols and forcing data (e.g., total precipitation, temperature, humidity, wind). They found the modeled results differ in magnitude, but that the overarching patterns were similar. Fisher et al. (2013) used a similar approach with nine models (CLM4-CN, HYLAND, LPJwsl, LPJ-Guess, OCN, ORCHIDEE, SDGVM, TRIFFID, and VEGAS), and focused on uncertainty in carbon fluxes over African rainforests. Two effects were noted which increased uncertainty between models, droughts and increased observed CO_2 – reduced precipitation and hotter temperatures decrease CO_2 uptake, and may change the humid tropics from a net sink to a net source of atmospheric CO_2 (Fisher et al., 2013). Results from these large scale analyses and others suggest that when working at the continental scale, uncertainties in ecosystem responses to elevated CO_2 under a changing climate yield considerable uncertainties in modeled results.

Although savannas are highly variable in climate and soils, and are influenced in complex ways by fire and grazing, they are not as well studied as croplands and forests (Ciais et al., 2011). The paucity of field measurements has implications for parameterizing and tuning ecosystem models. In general, highly variable soils in Africa can complicate modeling and remote sensing efforts. Termites are a source of methane, but lack of information prevents their inclusion in most models. Traore et al. (2014) cites an “attrition of meteorological networks” indicating that the collection of some kinds of data which are necessary for modeling is in decline. A lack of data on the carbon budget for African landscapes makes dynamic modeling results more uncertain (Bombelli et al., 2009a; Ciais et al., 2011).

Ciais et al. (2011) proposed an African integrated carbon observing system, where an array of eddy covariance flux towers in grazing lands, ongoing satellite mapping of CO, CO_2, CH_4, fire, and land use change, and modeling fuse to provide information to guide planning. Efforts such as CarboAfrica (www.carboafrica.net) are improving our knowledge of Sub-Saharan African carbon dynamics. New satellite sensors are becoming available, such as microwave sensors that allow for estimation of tree biomass, which will reduce modeling uncertainties (Traore et al., 2014). These and other efforts will improve data availability.

6.5 Portability and Assessment of Soil Carbon Models

Whether or not a given model may be used in a variety of locations with confidence (i.e., its portability) is an important question. In terms of mechanics, many soil carbon models are highly portable. The structure of parameter sets allows values to be set to describe local ecosystem dynamics. For example, the popular biogeochemical model CENTURY has been used in hundreds of applications from all over the world. That model is distributed with draft parameter sets for a variety of land cover types. Some models were originally specific to sites, but those too may be modified to be applicable elsewhere.

Other models are globally applicable, suggesting their suitability for a variety of locations. Most models include some kind of landscape unit, an area treated as homogeneous, for which parameters are provided. For example, those parameters may be adjusted to best represent C4 rather than C3 grasses, or vegetation growing on nutrient rich volcanic soils. Even though termites are not explicitly captured in most models, decomposition rates may be adjusted to approximate their effects. And models may be edited to dampen effects of grazing on plant production, approaching a compensatory response. For example, CENTURY and G-Range include seven distinct plant functional responses to grazing, any one of which an analyst may use for a given landscape unit. Ongoing research will inform an upcoming revision of G-Range that will explicitly support compensatory vegetation production under grazing.

Related to portability and model choice, an analyst selecting a model for use must also consider whether the model is appropriate for the questions that are to be addressed. All models include constraints and simplifications that may limit their applicability to a particular problem. Some examples are briefly cited here; a more thorough review of models applicable to Africa and their limitations would be useful, as cited below. CENTURY is point-based, and so modeling large areas requires the integration of results from many simulations. This may be inappropriate for some uses. A simplification made when G-Range was adapted from CENTURY was the exclusion of phosphorous modeling; analyses where phosphorous limitations are a focus would exclude the use of G-Range. The RothC model is explicitly for non-flooded soils only (Coleman et al., 1997), although the ECOSSE model (based on RothC) has been developed to deal with flooded soils (Smith et al., 2010). The SNAP model does not include nitrogen dynamics nor is temperature considered – soil temperature varies so little in the Serengeti that its inclusion was unnecessary (Ritchie, 2014), but SNAP does include pathways for decomposition by invertebrate detritivores such as termites. More generally, some models include few parameters (e.g., SNAP has five). Analysts may balance the ease by which simpler models may be applied versus the usefulness of more complex models to address many kinds of scenarios. Models with many parameters (e.g., SAVANNA) take much longer to apply to a new area, but in essence, many of the parameters present opportunities for scenarios to be addressed.

A more difficult question regarding portability is the degree to which models must be assessed for each new application. Using an application in discovery and decision making without some calibration using local observations is risky (Stockman et al., 2013). Assessing each subprocess is not necessary, given that many have been assessed in many settings. Therefore broad ecosystem responses which are common to all systems may be expected to be simulated well in novel applications, given our understanding of ecosystem processes and success in incorporating these in models. However, as much assessment as may be practically carried out, in line with the intended use of the application, is recommended. For some uses, this may require a detailed comparison to spatially and temporally collected observations. However, for other uses, a comparison of simulated results to measures derived from satellite images (e.g., annual net primary productivity, leaf area index, greenness) may be sufficient.

6.5 Gaps in Methods, Recommendations and Prospects

In soil carbon modeling, areas for improvement include better representation of microbial systems and more detailed representations of food webs. Today models often represent those ecosystem components indirectly as elements that consume and break down materials, rather than as entities with dynamics of their own that can link decomposition and metabolism more realistically. More frequent inclusion of nutrient modeling responses would improve the predictability of models (Fisher et al., 2013). More frequent and variable weather events associated with climate change make intra- and inter-annual variation relevant for upcoming research. More generally, responses to disturbances such as droughts and fires are not simulated well with current methods (Liu et al., 2011). Models may benefit by more often representing soil organic matter differently through the soil profile, and incorporating the influence of pH on carbon dynamics (Stockman et al., 2013). Many models do not incorporate land use change when judging carbon flux, which would improve estimates (Ciais et al., 2009, 2011; Fisher et al., 2013). Gaps in spatial inputs remain, such as information on shrub cover, which is poorly known across the globe. Ground-truthed observations are now sparse, but that resource is improving. Lastly, we expect an important advancement will be inclusion of social aspects into modeling carbon stocks. This shift from ecosystem modeling to a coupled natural and human systems approach will help inform policy makers about the ecological and social consequences of decisions. For example, in a unified modeling platform, land use in a region may be represented in a spatially and temporally explicit way, and implications for carbon storage quantified. Changes to land use may then be put in place to meet some storage goal, and implications for peoples' livelihoods may be quantified.

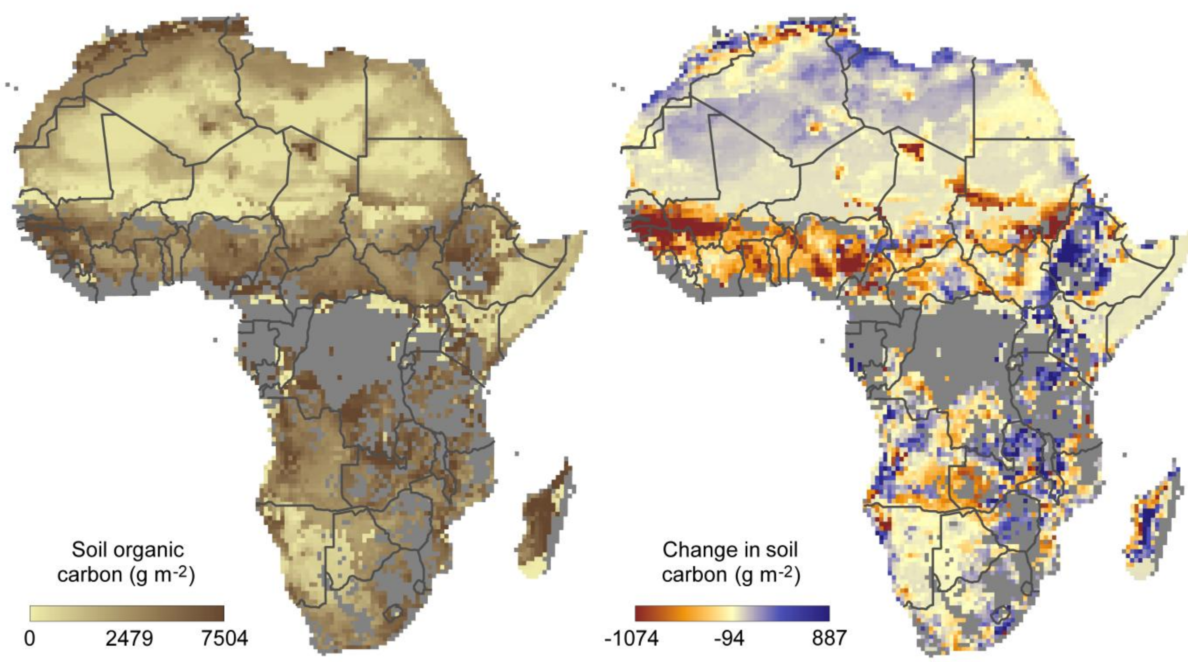
A practical means to speed analysts' identification of appropriate models would be helpful. Qualitative model comparisons usually entail defining a classification scheme, and then placing models within the defined classes according to their properties (e.g., VEMAP members, 1995; McGill 1996; Manzoni and Porporato, 2009). The classification may include attributes of the models, such as time step, relative complexity, spatial and temporal scale, cost, and data requirements. It may also include aspects of use of the model, such as how well the model has been field tested and its performance, how well it is documented, the agreement between the data needed by the model and available datasets, its flexibility for use in addressing questions, and its relative ease of use (Barnwell et al., 1992). The results may be presented on-line as matrices with the attributes of the models and the tasks to which they are most suited as matrix dimensions. An example of this for GHG accounting models in general has been developed by FAO and partners and consists of an Excel tool which guides the user through a number of decision trees to arrive at the most appropriate tool (Colomb et al., 2013).

In contrast to these qualitative model comparisons, quantitative comparisons of ecosystem models are difficult because any disagreements in output may be due to different processes incorporated in the models or different spatial or temporal scales (Melillo et al., 1995). Meta-analyses are helpful, but the diversity of approaches and applications prevent direct comparisons. Model comparisons can minimize these differences with efforts with formalized methods and use large research teams (e.g., VEMAP members 1995; Huntzinger et al., 2013). Because of the difficulty of these comparisons, such programs typically include a small number of models. Parton (1996) suggests that sufficient modularity in simulation tools would allow procedures from different models to be swapped, and the implications for results explored.

Standards in modeling indicate a maturing of approaches, and that is true for carbon storage. Programs, such as the Verified Carbon Standard (v-c-s.org), provide a suite of tools that standardize the means in which carbon dynamics are derived, or require the use of specific models or models with specific attributes in assessments. Standardization of methods is steadily improving and made more straightforward through web-based tools (e.g., Eve et al., 2014).

Manzoni and Porporato (2009), in their meta-analysis of models, estimated that the number of soil biology models is increasing at a rate of 6 percent per year. New models often build on the methods used in existing models. For example, the CENTURY model was drawn on to create parts of other models (i.e., SAVANNA, G-Range). This has sped the development of new models, but suggests that the cited growth rate should not be considered the rate of innovations. Related to this, carbon stock estimates for Africa are from a few teams using related tools (Stockman et al., 2013). Regardless, this review suggests that ample models exist for estimating soil carbon in Africa, that they capture processes to describe carbon dynamics sufficiently well for most policy questions, and that they are portable. Creating new soil carbon models is not a priority at this time. Instead, we encourage: 1) development of means to identify an appropriate model for a given task, from among the many candidates; 2) support more African applications at regional, national, and continental scale; 3) continued efforts to reduce uncertainties in ecosystem models and to incorporate new spatial data, which is rapidly developing ; 4) improve our modeling of disturbed, transient, and highly variable ecosystems; 5) create and improve linkages to human systems and their use of the land, allowing for coupled natural and human systems queries to be addressed; and perhaps most importantly, 6) continued advancement of our understanding of the magnitudes and durations of changes in ecosystems in response to climate change – reduced uncertainties in ecosystem science will lead to reduced uncertainties in ecological modeling outcomes.

Figure 4. Ensemble Spatial estimates of soil organic carbon (left) in the year 1972 from the G-Range model, and estimated change in soil carbon in 2050 from simulations using climate projections from five global circulation models based on Representative Concentration Pathway 8.5, with data downscaled to half-degree resolution. Climate projections were provided by P.K. Thornton using data from the Agricultural Model Intercomparison and Improvement Project (Rosenzweig et al., 2014). Map shading and legend labels reflect the mean plus and minus twice the standard deviation, except where the simulated carbon density cannot be less than zero.



Chapter 7. Model based estimates of potential carbon sequestration in grass/rangelands in Sub-Saharan Africa

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Given the widespread interest in the mitigation potential of grazing lands among policy makers and practitioners, Henderson et al. (2015) sought to estimate the effectiveness of mitigation practices in grazing lands using process-based models and detailed spatial information. Here we extract information relevant to Sub-Saharan Africa from that work, which was supported by the UN Food and Agricultural Organization and the Climate Change, Agriculture, and Food Security research program. All assessments of GHG mitigation potential in grazing lands were based on the concept that a change in management practices can lead to a change in C stocks and/or N_2O emissions (Conant, 2011). Thus all estimates of mitigation potential were constructed using (1) information about changes in soil C storage and N_2O emission rates given a change in land management and (2) information about where land management changes were feasible (Paustian et al., 1997). It was clear that not all management changes were appropriate or possible for all grazing lands, as their applicability and effectiveness depend on a range of factors such as accessibility, soil conditions, climate, and current and past management. To-date, limited data on grazing land management have constrained the ability of researchers to delineate areas amenable to improved management from those that are not amenable (Conant and Paustian, 2004). Another limitation of most broad-scale assessments is that they have relied on emission factors generated from the synthesis or meta-analysis of published studies (Conant et al., 2001; Ogle et al., 2004; Smith, 2007). While these are often the most sophisticated approaches possible (analogous to Tier 1 and Tier 2 approaches under IPCC), they are inherently reliant on a small set of observations under a restricted set of biophysical/management conditions that is widely extrapolated.

The work of Henderson et al. (2015) contributes to the current body of evidence about mitigation practices in the world's grazing lands in three important ways. First, they applied ecosystem models – the CENTURY and Daycent models (Parton et al., 1987, Parton et al., 1998) – that represent the effects of a variety of management practices on C and N cycling in agro-ecosystems. These models are capable of representing the multiple interactions between biophysical processes and management at a landscape scale, and they offer an alternative to the usual approach of extrapolation from field studies. Second, by using observations of past and current land use, they confined their assessment to areas where livestock production is present and where practice changes were likely to be effective, rather than assuming the blanket application of management practices across all or most of the Sub-Saharan Africa's grazing lands. Finally, by modeling the linkage between forage, animal production, and animal GHG emissions they aimed to shed new light on the net mitigation potential of C sequestration practices in the world's grazing lands.

Henderson et al. (2015) estimated that optimization grazing pressure to maximize forage production could lead to sequestration 33.3 Tg CO_2 yr^{-1} in grazing lands across Sub-Saharan Africa (Table 4). The soil C sequestration potential of 5.3 Tg CO_2 yr^{-1} for legume sowing was small due to its limited feasibility across a much smaller total area, though rates of sequestration per unit area were greater than for improved grazing management. Increased N_2O emissions from sowing legumes were estimated to offset 26% of the soil C sequestration benefits, in CO_2 equivalent terms. Increased N_2O emissions from N fertilization exceeded soil C sequestration in most Sub-Saharan Africa cells modeled. Estimates for C stocks though improved management in grazing lands from Henderson et al. (2015) are lower than other estimates (Lal, 2004; Smith et al., 2007), mainly due to the much smaller grazing land area over which mitigation practices were estimated to be effective. Henderson et al. (2015) found that each of the practices evaluated can lead to a large increase in dry matter production.

Conclusions

- The opportunities to increase forage production in Sub-Saharan grazing lands are substantial – on the order of 500 Tg DM yr^{-1}.

- Realizing these increased forage benefits could also lead to carbon sequestration that offsets potential increases in greenhouse gas emissions.

- Model results indicate net increases in greenhouse gas emissions with implementation of some practices that increase forage production, thus regional impacts on forage production should be carefully evaluated for associated net greenhouse gas emission costs.

- Improved grazing management and sowing legumes offer substantial opportunities for enhancing forage production while minimizing or offsetting greenhouse gas emissions.

Table 4. Carbon sequestration and net mitigation potential in Sub-Saharan Africa for a variety grazingland land management changes. Data extracted from results presented in Henderson et al. (2015)

Management practice	Amenable area	Amenable area	C sequestration potential	ΔN_2O flux	Net GHG mitigation potential		Δ Forage consumption
	(Mha)	(%)	(Tg CO_2)	($TgCO_2$-eq)	($TgCO_2$-eq)	(Mg CO_2.eq ha^{-1})	(Tg DM)
Native rangelands – optimum grazing	79.1	16.6	24.3	-	24.3	0.31	76.0
Planted pastures – optimum grazing	65.5	43.2	9.0	-	9.0	0.14	275.9
Sowing legumes	5.9	3.9	5.3	1.4	3.9	0.70	2.6
Fertilization	32.3	-	-1.26	9.37	-0.5	-8.11	151.8
SSA total	**182.8**	**-**	**37.3**	**10.8**	**36.7**	**-**	**506.3**

Chapter 8: Synthesis

This final section presents some of the salient points from the previous chapters to answer key questions concerning the potential for C sequestration in grazing lands through improved land management practices in Sub-Saharan Africa.

1) Batjes (2004) estimates that grasslands in Africa have the potential to sequester between 7 – 42 Tg C year^{-1}. Although fertilization and fire management could both contribute to C sequestration, most of the potential sequestration in non-degraded grasslands is through changes in grazing management practices. Many management strategies which increase above and below ground C stocks, thereby benefitting climate change mitigation, can also have adaptation benefits, leading to improved use of scarce resources such as water and manure.

2) The key determinants of C sequestration potential in grazing lands in SSA are climate (rainfall amount and regime) and soil type. On top of these key factors, sequestration potential is then determined by fire, drought and disease (both natural and human induced) and human activities such as land use conversion, grazing management practices and inputs from external sources (organic amendments, fertilizers, etc.). Removal of a large amount of aboveground vegetation, continuous heavy stocking rates, frequent human-ignited burning and other poor grazing management practices are important human-controlled factors that adversely influence grassland production and have also led to depletion of soil C stocks. However, good grassland management that contributes to plant recovery, diversity of species, soil cover, enhanced organic matter, and functioning ecosystem processes (solar energy capture, biological communities) and cycles (nutrients, water) can potentially reverse historical soil C losses and sequester substantial amounts of C in soils.

3) We understand the basic soil forming factors and how these govern C sequestration, however, our understanding of the following (which have high relevance to the situation in grasslands in SSA) are still poor:

- Phosphorus and the role it plays in C sequestration in grasslands, particularly the co-limitation of N and P in grasslands dominated by C4 species (as is the case in SSA)
- The effect of UV radiation on decomposition
- The role of termites
- An understanding of shifts between shrublands and grasslands, the drivers of this change and the impact this has on above and below ground C stocks

These gaps in our understanding could be narrowed in the next 5 years with targeted research but efforts should be focused on those geographic areas with the highest C sequestration potential.

4) Ecosystem models are capable of representing the multiple interactions between biophysical processes and management at a landscape scale. Examples of broad-scale application to situations in Africa, including grazing lands, are limited but increasing in number (Conant and Paustian, 2002; Ardo and Olsen, 2003; Kamoni et al., 2010). However the use of models is limited by a lack of information on biophysical aspects of grazing lands in SSA, data for model parameterization and a lack of information on management practices at the appropriate scale.

5) Key data that would improve our understanding of C sequestration in grazing lands in SSA include information on the land management practices themselves (i.e., what they are and where they occur) and the effects of socio-economically and biophysically driven changes in land use and management on

soil organic carbon (SOC) content, such as the occurrence of natural and human-made fires, within the various livestock production systems. Such landscape scale effects may be more easily considered now as high (250 m) resolution soil property maps, derived from digital soil mapping, have very recently become available for SSA (Hengl et al., 2015).

6) In some cases the cost of demonstrating the change in C stocks in soils to the required accuracy and precision may exceed the benefits that accrue from the increase in stocks (IPCC, 2003). Although infrared spectroscopy significantly reduces the analytical cost and speed of measuring soil C, costs incurred in soil sampling and preparation still form the largest component of the cost (Aynekulu et al., 2011). Reward for known 'good C practice' in leiu of frequent monitoring of C stocks may be a more realistic way forward. Although soil C is a key soil quality indicator, it alone does not provide sufficient information to guide wise use of land resources, and therefore standalone C measurement systems will have limited value, especially given the large resources required for field sampling and soil preparation. In most cases it will be most efficient to embed C measurement within broader land health surveillance schemes.

7) Chapter 5 presents a map-based estimate of SOC content in grazing lands in SSA, discusses differences in estimates observed in different livestock production systems, and points to the need for higher resolution biophysical and socio-economic information to underpin more detailed studies at a landscape scale.

8) Chapter 7 presents estimated C sequestration potential of grazing lands in SSA made using a model-based approach. This estimate is extracted from a global study and is therefore based on a limited number of data points and management practices. In order to improve this estimate a modeling exercise is needed which considers management practices specific to SSA and this needs to be underpinned by field studies.

In addition to the points given above the authors as a group wish to put forward the following general recommendations for future projects/activities concerning C sequestration in grazing lands in SSA:

I. C sequestration for climate change mitigation should be treated as a co-benefit rather than the target of a project/activity. There may be instances in which there are large social costs to C sequestration and by making sequestration the primary goal, a situation could arise where the social burdens of the project are greater than the C benefits. The likelihood of this is increased by the facts that the costs and benefits of C sequestration are not associated with the same people and the ones who must bear the costs (or burdens) are normally the poorest (Milne et al., 2015). That is to say, if the plan to sequester C involves a reduction in stocking rate or grazing intensity, it may have significant costs to the local livestock operators, but the benefits of C sequestration accrue to the world as a whole.

II. Paying people to maintain C stocks by persisting with existing good practice may be easier than introducing new practices to increase stocks. In addition, significant potential for peer-to-peer learning among herders exists by highlighting good practices and promoting cross-site visits and other learning opportunities for successful herder-led sequestration initiatives.

III. Projects should be developed with strong input from the communities living in the areas (bottom up) where the project is to occur so that they have a sense of ownership of the project and are then more likely to cooperate in its implementation

IV. The United Nations Convention to Combat Desertification (UNCCD) aims for zero net land degradation by 2030 (UNCCD, 2012). However, in terms of targeting resources, it appears that there is some acceptable level of degradation (measured as loss in SOC), and that if efforts are made to increase SOC in areas in which SOC loss is less than this acceptable level, there may be costs or burdens to the local people. When resources are limited, the focus should therefore be on areas where the co-benefits of C sequestration and livelihood improvement are highest and this may be in moderately rather than severely degraded areas. At present an acceptable level of degradation, or a way of defining it has yet to be determined, but interventions should consider net livelihoods of local populations as a key metric of success of programs that aim to improve rangeland health and C sequestration.

V. There is less evidence of the impact of rangeland management approaches in enhancing the carbon sequestration in non-equilibrial (erratic and changing) systems. This is because system responses are likely to be a function of rainfall and its variability as well as a function of manipulating livestock stocking rates and times of recovery. This does not mean that we should not intervene in non-equilibrial systems; well managed grasslands have the potential to enhance the capture of rainfall and subsequently opportunities to enhance people's lives and landscapes. In these areas we should concentrate on improving rangeland health and livelihoods of the local people with C sequestration for climate change mitigation coming as a co-benefit.

VI. In terms of where to focus investment, from a purely C sequestration point of view, investment should be in areas where mineralization rates are low (cold, moist) or input rates are high (warm, wet). In areas where C sequestration rates are likely to be lower (arid, semi-arid) but cover extensive land areas, projects should include C sequestration as part of a package of multiple benefits in which practices that sequester C in grasslands enhance productivity, improve livelihoods, increase biodiversity and benefit multiple ecosystem services.

9. References

Adams, H. D., Guardiola-Claramonte,M., Barron-Gafford, G. A., Villegas, J. C., Breshears, D. D., Zou, C. B., Troch, P. A., Huxman, T. E., 2009. Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America 106:7063-7066.

AfSIS, 2014. The Africa soil information service. http://www.africasoils.net (last accessed 29 Nov 2014).

AfSIS, 2015a. Africa Soil Information Service laboratory standard operating procedures. Soil-Plant Spectral Diagnostics Laboratory, World Agroforestry Centre (ICRAF), Nairobi: http://worldagroforestry.org/research/land-health-management/spectral-diagnostics-laboratory/sops (last accessed 22 July 2015).

Allen, V. G., Batello, C., Berretta, E. J., Hodgson, J., Kothmann, M., Li, X., McIvor, J., Milne, J., Morris, C., Peeters A., Sanderson, M., 2011. An international terminology for grazing lands and grazing animals. Grass and Forage Science 66: 2–28.

Allison, F. E., 1973. Soil organic matter and its role in crop production. Elsevier, Amsterdam, The Netherlands.

Andreae, M.O., Merlet, P., 2001. Emission of trace gases and aerosols from biomass burning. Global Biogeochemical Cycles 15 (4): 955-966.

Angassa, A., 2012. Effects of grazing intensity and bush encroachment on herbaceous species and rangeland condition in southern Ethiopia. Land Degradation and Development 25: 438 – 451.

Ardo, J., Olson, L., 2003. Assessment of soil organic carbon in semi-arid Sudan using GIS and the CENTURY model. Journal of Arid Environments 54: 633–651.

Ash, A. J., Corfield, J. P., McIvor, J. G., Ksiksi, T. S., 2011. Grazing Management in Tropical Savannas: Utilization and Rest Strategies to Manipulate Rangeland Condition. Rangeland Ecology & Management, 64 (3): 223-239.

AU-IBAR, 2012. Rational Use of Rangelands and Fodder Crop Development in Africa. AU-IBAR Monographic Series No. 1.

Aynekulu E, Koala E, Feyisa J, Sawadogo L, Shepherd K. 2014. Livelihood Diversifying Potential of Livestock Based Carbon Sequestration Option in Pastoral and Agro Pastoral Systems in Africa Project. Final Report. World Agroforestry Centre, Nairobi, Kenya.

Aynekulu E, Shepherd K. 2013. Improving Measurements of Agricultural Productivity through Methodological validation and Research (LSMS-ISA). World Agroforestry Centre (ICRAF).

Bagchi, S., Richie, M.E., 2010. Introduced grazers can restrict potential soil carbon sequestration through impacts on plant community composition. Ecology Letters 13: 959–968.

Bai, Z. G., Dent, D. L., Olsson, L., Shcaepman, M. E., 2008. Global assessment of land degradation and improvement. 1. Identification by remote sensing. Report 2008/01, ISRIC - World Soil Information, Waginengen.

Banwart, S., Black, H., Cai, Z., Gicheru, P., Joosten, H., Victoria, R., Milne, E., Noellemeyer, E., Pascual, U., Nziguheba, G., Vargas, R., Bationo, A., Buschiazzo, D., de-Brogniez, D., Melillo, J., Richter, D., Termansen, M., van Noordwijk, M., Goverse, T., Ballabio, C., Bhattacharyya, T., Goldhaber, M., Nikolaidis, N., Zhao, Y., Funk, R., Duffy, C., Pan, G., la Scala, N., Gottschalk, P., Batjes, N., Six, J., van Wesemael, B., Stocking, M., Bampa, F., Bernoux, M., Feller, C., Lemanceau, P., Montanarella, L., 2014. Benefits of soil carbon: report on the outcomes of an international scientific committee on problems of the environment rapid assessment workshop. Carbon Management 5: 185-192.

Banwart, S.A., Noelmeyer, E., Milne E (Eds.), 2015. Soil carbon: Science, Management and policy for multiple benefits. CABI, Wallingford (UK).

Barnwell, T.O., R.B. Jackson, IV, E.T. Elliott, I.C. Burke, C.V. Cole, K. Paustian, E.A. Paul, A.S. Donigian, A.S. Patwardhan, A. Rowell, and K. Weinrich. 1992. An approach to assessment of management impacts on agricultural soil carbon. Water, Air, & Soil Pollution 64: 423-435.

Bationo, A., Kihara, J., Vanlauwe, B., Waswa, B., Kimetu, J., 2007. Soil organic carbon dynamics, functions and management in West African agro-ecosystems. Agricultural Systems 94: 13-25.

Batjes, N.H., 2004. Estimation of soil carbon gains upon improved management within croplands and grasslands of Africa. Environment, Development and Sustainability 6: 133-143.

Behnke, R. H., 1999. Stock-movement and range-management in a Himba community in north-western Namibia. in Niamir-Fuller, M. (Ed). Managing Mobility in African Rangelands. FAO and Beijer International Institute of Ecological Economics, London. Pages 184-216.

Bekure, S., de Leeuw, P. N., Grandin, B. E., Neate, P. J. H. (Eds.), 1991. An Analysis of Maasai Pastoralists in Eastern Kajiado District, Kenya. ILCA Systems Study 4, International Livestock Center for Africa, Addis Ababa, Ethiopia.

Bellon-Maurel V, McBratney A. 2011. Near-infrared (NIR) and mid-infrared (MIR) spectroscopic techniques for assessing the amount of carbon stock in soils e Critical review and research perspectives. Soil Biology & Biochemistry 43: 1398-1410.

Bennett, A. J., 2000. Environmental consequences of increasing production: some current perspectives. Agriculture, Ecosystems and Environment 82: 89–95.

Bernoux, M., Tinlot, M., Bockel, L., Branca, G., Gentien, A., 2011. Ex-ante carbon balance tool (EX-ACT): Technical Guidelines for Version 3.0. FAO, Rome, p. 84.

Beukes, P. C., Cowling, R. M., 2003. Non-Selective Grazing Impacts on Soil-Properties of the Nama Karoo. Journal of Range Management 56 (5): 547-552.

Bird, M.I., Veenendaal, E.M., Moyo, C., Lloyd, J., Frost, P., 2000. Effect of fire and soil texture on soil carbon in a sub-humid savanna (Matopos, Zimbabwe). Geoderma 94: 71–90.

Bird DN, Pena N, Schwaiger H, Zanchi G. 2010 Review of existing methods for carbon accounting. Occasional paper 54. CIFOR, Bogor, Indonesia

Bombelli, A., Henry, M., Castaldi, S., Adu-Bredu, S., Arneth, A., de Grancourt, A., Grieco, E., Kutsch, W.L., Lehsten, V., Rasile, A., Reichstein, M., Tansey, K., Weber, U., Valentini, R., 2009. An outlook on the Sub-Saharan Africa carbon balance, Biogeosciences, 6: 2193-2205.

Bombelli, A., M. Henry, S. Castaldi, S. Adu-Bredu, A. Arneth, A. de Grandcourt, E. Grieco, W.K. Kutsch, V. Lehsten, A. Rasile, M. Reichstein, K. Tansey, U. Weber, and R. Valentini. 2009a. The Sub-Saharan Africa carbon balance, an overview. Biogeosciences Discussion 6: 2085-2123.

Bond, W.J., Woodward, F.I., Midgeley, G.F., 2004. The global distribution of ecosystems in a world without fire. New Phytologist 165 (2): 525-538.

Bond, W.J., Keely, J.E., 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. Trends in Ecology and Evolution 20: 387-394.

Boone, R.B., M.B. Coughenour, K.A. Galvin, and J.E. Ellis. 2002. Addressing management questions for Ngorongoro Conservation Area using the Savanna Modeling System. African Journal of Ecology 40: 138-150.

Boone, R. B., Hobbs, N. T., 2004. Lines around fragments: Effects of fencing on large herbivores. African Journal of Range & Forage Science 21: 79-90.

Boone, R.B. 2005. Quantifying changes in vegetation in shrinking grazing areas. Conservation & Society 3: 150-173.

Boone R.B. 2007. Effects of fragmentation on cattle in African savannas under variable precipitation. Landscape Ecology 22: 1355-1369

Boone, R.B., R.T. Conant, and T.E. Hilinski. 2011a. G-Range: Development and use of a beta global rangeland model. Final report to the International Livestock Research Institute, Nairobi, Kenya

Boone, R.B., K.A. Galvin, S.B. BurnSilver, P.K. Thornton, D.S. Ojima, and J.R. Jawson. 2011b. Using coupled simulation models to link pastoral decision making and ecosystem services. Ecology and Society 16(2): 6 [online] URL: http://www.ecologyandsociety.org/vol16/iss2/art6/

Boone, R.B., R.T. Conant, and J. Sircely. 2013. Adjustment and sensitivity analyses of a beta global rangeland model. Final report to the International Livestock Research Institute, Nairobi, Kenya

Brandt, L.A., King, J.Y., Hobbie, S.E., Milchunas, D.G., Sinsabaugh, R.L. 2010. The role of photodegradation in surface litter decomposition across a grassland ecosystem precipitation gradient. Ecosystems 13: 765-781.

Bridges, E. M., Oldeman, L. R., 1999. Global assessment of human-induced soil degradation. Arid Soil Research and Rehabilitation 13: 319-325.

Brottem, L., Turner, M. D., Butt, B., Singh, A., 2014. Biophysical Variabilty and Pastoral Rights to Reources: West African Transhumance Revisited. Human Ecology 42: 351 – 365.

Briske, D.D., Sayre, N.F., Huntsinger, L., Fernandez-Gimenez, M., Budd, B., Derner, J.D., 2011. Origin, Persistence, and Resolution of the Rotational Grazing Debate: Integrating Human Dimensions Into Rangeland Research. Rangeland Ecol. Manage. 64: 325–334.

Butt, B., Shortridge, A., WinklerPrins, A., 2009. Pastoral herd management, drought coping strategies, and cattle mobility in southern Kenya. Annals of the Association of American Geographers 99: 309-334.

Catley, A., Lind J., Scoones, I. (Eds.), 2013. Pastoralism and Development in Africa: Dynamic Change at the Margins. London: Routledge and Earthscan. Pp. 295. ISBN-13: 978-0-415-54071-1.

Chapman, D. F., Lemaire, G., 1993. Morphogenic and structural determinants of plant regrowth after defoliation. In Baker, M. J. (Ed.). Grasslands for Our World. SIR, Wellington, NZ. Pages 55-64

Ciais, P., Reichstein, M., Viovy, N., Grainer, A., Ogee, J., Allard, V., Aubinet,M., Buchmann, N., Bernhofer, C., Carrara, A., Chevallier, F., De Noblet, F., Friend, A. D., Friedlingstein, P., Grunwald, T., Heinesch, B., Keronen, P., Knohl, A., Krinner, G., Loustau, D., Manca, G., Matteucci, G., Miglietta, F., Ourcival, J. M., Papale, D., Pilegaard, K., Rambal, S., Seufert, G., Soussana, J. F., Sanz, M. J., Schulze, E. D., Vesala, T., Valentini, R., 2005. Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature 437: 529-533.

Ciais, P., S.-L. Piao, P. Cadule, P. Friedingstein, and A. Chedin. 2009. Variability and recent trends in the African carbon balance. Biogeosciences 5: 3497-3532.

Ciais P., Bombelli, A.,Williams, M.,Piao, S.L.,Chave, J.,Ryan, C.M.,Henry, M.,Brender, P.,Valentini, R., 2011. The carbon balance of Africa: synthesis of recent research studies. Phil. Trans. R. Soc. A.369: 1–20.

Cobo JG, Dercon G, Yekeye T, Chapungu L, Kadzere C, Murwira A, Delve R, Cadisch J. 2010. Integration of mid-infrared spectroscopy and geostatistics in the assessment of soil spatial variability at landscape level. Geoderma 158: 398–411.

Coleman, K., D.S. Jenkinson, G.J. Crocker, P.R. Grace, J. Klír, M. Körschens, P.R. Poulton, and D.D. Richter. 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma 81: 29-44.

Colomb V., Touchemoulin O., Bockel L., Chotte J., Martin S., Tinlot M., Bernoux, M. 2013, Selection of appropriate calculators for landscape-scale greenhouse gas assessment for agriculture and forestry Environ. Res. Lett. 8: 015029.

Conant, R. T., Paustian, K., Elliott, E. T., 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11: 343-355.

Conant, R. T., Paustian, K., 2002. Potential soil carbon sequestration in overgrazed grassland ecosystems. Global Biogeochemical Cycles 16: 1143.

Conant, R.T., Paustian, K., Elliott, E.T., 2002. Grassland management and conversion into grassland: effects on soil Carbon. Ecol. Appl. 11: 343–355.

Conant, R.T., Ogle, S.M., Paul, E.A., Paustian, K., 2010. Measuring and monitoring soil organic carbon stocks in agricultural lands for climate mitigation. Frontiers in Ecology and the Environment 9: 169-173.

Conant RT, Berdanier AA, Grace PR. 2013. Patterns and trends in nitrogen use and nitrogen recovery efficiency in world agriculture. Global Biologeochemical Cycles 27: 558-566.

Conant, R. T., Cerri, C. E. P., . Osborn, B. B. P., Paustian, K., 2015. Grassland management impacts on soil carbon stocks; A new synthesis. Ecological Applications (submitted).

Coppock, D. L., 1990. Ethiopia's arid and semi-arid lowlands: environments, pastoral economies and issues in natural re-source sustainability. Pages 116-122 in H. Van Houten, ed. Proceedings of the First Conference on Natural Resources Improvement in Ethiopia. February 1989, Addis Ababa, Ethiopia. Institute of Agricultural Research, Addis Ababa, Ethiopia.

Coughenour, M. B. 1992. Spatial modeling and landscape characterization of an African pastoral ecosystem: a prototype model and its potential use for monitoring drought. in: D.H. McKenzie, D.E. Hyatt and V.J. McDonald (eds.). Ecological Indicators, Vol. I. Elsevier Applied Science, London and New York. pp. 787-810

Craine, J.M., Morrow, C.,Stock, W.D. 2008. Nutrient concentration ratios and co-limitation in South African grasslands. New Phytologist 179: 829-836.

Croft H, Kuhn NJ, Anderson, K. On the use of remote sensing techniques for monitoring spatio-temporal soil organic carbon dynamics in agricultural systems. Catena 94: 64–74

de Brogniez, D., Mayaux, P., Montanarella, L., 2011. Monitoring, Reporting and Verification systems for Carbon in Soils and Vegetation in African, Caribbean and Pacific countries. Publications Office of the European Union, Luxembourg, p. 99.

DeFries, R. S., Field, C. B., Fung, I., Collatz, G. J., Bounoua, L., 1999. Combining satellite data and biogeochemical models to estimate global effects of human-induced land cover change on carbon emissions and primary productivity. Global Biogeochemical Cycles 13: 803-815.

de Gruijter J, Burs D, Bierkens M, Knotters M. 2006. Sampling for Natural Resource Monitoring. Springer-Verlag Berlin Heidelberg. Pp.332.

Devendra., C., Morton, J., Rischkowsky, B., Thomas, D., 2005. Livestock systems. In: Owen, E., Kitalyi, A., Jayasuriya, N., Smith, T. (Eds.) Livestock and wealth creation: Improving the husbandry of animals kept by resources—poor people in developing countries. Nottingham Univ. Press, UK.

Derner, J.D., Boutton, T.W., Briske, D.D., 2006. Grazing and ecosystem carbon storage in the North American Great Plains. Plant and Soil 280: 77-90.

Derner, J.D., Schuman, G.E., 2007. Carbon sequestration and rangelands: a synthesis of land management and precipitation effects. Journal of Soil and Water Conservation, 62: 77–85.

Dohn, J., Dembélé, F., Karembé, M., Moustakas, A., Amévor, K.A., Hanan, N.P., 2013. Tree effects on grass growth in savannas: competition, facilitation and the stress-gradient hypothesis. Journal of Ecology 101: 202–209.

Diaz, D., Charnley, S. and Gosnell, H., 2010. Engaging western land owners in climate change mitigation: A A Guide to Carbon-Oriented Forest and Range Management and Carbon Market Opportunities. USDA General Technical Report PNW-GTR-801 Dec 2009.

Diels, J., K. Aihou, E.N.O. Iwuafor, R. Merckx, O. Lyasse, N. Sanginga, B. Vanlauwe, and J. Deckers. 2001. Options for soil organic carbon maintenance under intensive cropping in the West African savannah. In: Vanlauwe, B., J. Diels, N. Sanginga, and R. Merckx (eds.), Integrated plant nutrient management in sub-Saharan Africa: from concept to practice. CABI Publishing Series, Oxfordshire, UK. Pages 299-312.

Dinesh D., B. Bett, R. Boone, D. Grace, J. Kinyangi, J. Lindahl, C.V. Mohan, J. Ramirez-Villegas, R. Robinson, T. Rosenstock, J. Smith and P. Thornton. 2015. Impact of climate change on African agriculture: focus on pests and diseases. Copenhagen, Denmark: CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS). www.ccafs.cgiar.org

Donigian, A. S., Jr., Barnwell, T. O., Jackson, R. B., Patwardhan, A. S., Weinreich, K. B., Rowell, A. L., Chinnaswamy, R. V., Cole, C. V., 1994. Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States. EPA/600/R-94/067, US EPA, Athens, GA.

Donovan P. 2013. Measuring soil carbon change: A flexible, practical, local method. Available at http://soilcarboncoalition.org/files/MeasuringSoilCarbonChange.pdf (Last accessed 29 Nov 2014)

Du Preez, C.C., Van Huyssteen, C.W., Mnkeni, P.N.S., 2011. Land use and soil organic matter in South Africa 1: A review on spatial variability and the influence of rangeland stock production. S. Afr. J. Sci., 107: 27-34.

Easter, M., K. Paustian, K. Killian, S. Williams, T. Feng, R. Al Adamat, N.H. Batjes, M. Bernoux, T. Bhattacharyya, C.C. Cerri, C.E.P. Cerri, K. Coleman, K., P. Falloon, C. Feller, P. Gicheru, P., Kamoni, E. Milne, D.K. Pal, D.K., D.S. Powlson, Z. Rawajfih, M. Sessay, and S. Wokabi. 2007. The GEFSOC soil carbon modeling system: a tool for conducting regional-scale soil carbon inventories and assessing the impacts of land use change on soil carbon. In: Milne, E., D.S. Powlson, C.E.P. Cerri, (eds.), Soil Carbon Stocks at Regional Scales. Agric. Ecosyst. Environ. 122: 13–25.

Ellis, J.E., Galvin, K., 1994. Climate Patterns and Land-use Practices in the Dry Zones of Africa Comparative regional analysis provides insight into the effects of climate variations. Bioscience 44 (5): 340 – 349.

Ellis, J. E., Galvin, K., McCabe, J. T., Swift, D. M., 1987. Pastoralism and drought in Turkana District Kenya. Report to the Norwegian Agency for International Development, Nairobi, Kenya.

Ellis, J., Coughenour, M.B. 1998 The Savanna integrated modelling system. In: Drylands: sustainable use of rangelands in the twenty-first century (eds Squires V, Sidahmed A) pp 97-106. IFAD technical reports, Rome.

Ellis J, Larsen K. 2008. Measurement, Reporting and Verification of Mitigation Actions and Commitments. OECD/IEA, Paris.

Ellis, J. E., Swift, D. M., 1988. Stability Of African Pastoral Ecosystems - Alternate Paradigms And Implications For Development. Journal of Range Management 41: 450-459.

Eswaran, H., R., Reich, A.P., Zdruli, P., 1997. Soil quality and soil productivity in Africa. J. Sustainable Agric. 10(4): 75–94.

EthioSIS 2014. Ethiopia Soil Information System. Ethiopian Agricultural Transformation Agency. Available at www.ata.gov.et/projects/ethiopian-soil-information-system-ethiosis. (Last accessed 29 Oct 2014).

Eve, M., Pape, D., Flugge, M., Steele, R., Man, D., Riley-Gilbert, M., Biggar, S. (Eds.). 2014. Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory. Technical Bulletin Number 1939. Office of the Chief Economist, U.S. Department of Agriculture, Washington, DC. 606 pages. July 2014.

Everson, C.S., George, W.J., Schulze, R.E., 1989. Fire Regime Effects on Canopy Cover and Sediment Yield in the Montane Grasslands of Natal South. African. Journal of Science 85: 113-116.

FAO, 2007. State of the World's forests 2007. Food and Agricultural Organization of the United Nations, Rome.

FAO, 2009. FAO STAT Statistical Database 2007. FAO, Rome.

FAO/LEAD, 2006. Livestock's long shadow. Environmental issues and options. FAO Rome. Available from ftp://ftp.fao.org/docrep/fao/010/a0701e/a0701e.pdf. (Last accessed 13th March 2015).

Fernandez-Gimenez, M. E., 2002. Spatial and social boundaries and the paradox of pastoral land tenure: A case study from postsocialist Mongolia. Human Ecology 30: 49-78.

Fernandez-Gimenez, M. E., Le Febre, S., 2006. Mobility in pastoral systems: Dynamic flux or downward trend? International Journal of Sustainable Development and World Ecology 13: 341-362.

Field, C. B., Lobell, D. B., Peters, H. A. Chiariello, N. R., 2007. Feedbacks of Terrestrial Ecosystems to Climate Change. Annual Review of Environment and Resources 32: 1-29.

Fisher, J.B., M. Sikka, S. Sitch, P. Ciais, B. Poulter, D. Galbraith, J.-E. Lee, C. Huntingford, N. Viovy, N. Zeng, A. Ahistrom, M. R. Lomas, P.E. Levy, C. Frakenberg, S. Saatchi, and Y. Malhi. 2013. African tropical rainforest net carbon dioxide fluxes in the twentieth century. Philosophical Transactions of the Royal Society B 368: 20120376.

Follett, R. F., Kimble, J. M., Lal, R., 2001a. The potential of US grazing lands to sequester carbon and mitigate the greenhouse effect. CRC Press, Chelsea, MI.

Follett, R. F., Kimble, J. M., Lal, R., 2001b. The potential of US grazing lands to sequester soil carbon. Pages 401-430 *in* Follett, R. F., Kimble, J. M, Lal, R. (Eds.). The potential of US grazing lands to sequester soil carbon. CRC Press, Chelsea, MI.

Friedlingstein, P., Cox, P., Betts, R., Bopp, L., Von Bloh, W., Brovkin, V., Cadule, P., Doney, S., Eby, M., Fung, I., Bala, G., John, J., Jones, C., Joos, F., Kato, T., Kawamiya, M., Knorr, W., Lindsay, K., Matthews, H. D., Raddatz, T., Rayner, P., Reick, C., Roeckner, E., Schnitzler, K. G., Schnur, R., Strassmann, K., Weaver, A. J., Yoshikawa, C., Zeng, N., 2006. Climate-carbon cycle feedback analysis: Results from the (CMIP)-M-4 model intercomparison. Journal of Climate 19: 3337-3353.

Fynn, R.W.S, Haynes, R.J. and O'Connor, T.G., 2003. Burning causes long-term changes in soil organic matter content of a South African grassland. Soil Biology and Biochemistry 35: 677 – 687.

Galvin, K.A., P.K. Thornton, R.B. Boone, and J. Sunderland. 2004. Climate variability and impacts on East African livestock herders. African Journal of Range and Forage Sciences 21: 183-189.

Gilhespy, S.L., S. Anthony, L. Cardenas, D. Chadwick, A. del Prado, C. Li, T. Misselbrook, R.M. Rees, W. Salas, A. Sanz-Cobena, P. Smith, E.L. Tilston, C.F.E. Topp, S. Vetter, and J.B. Yeluripati. 2014. First 20 years of DNDC (DeNitrification DeComposition): model evolution. Ecological Modelling 292: 51-62.

Goldman, M. J., Riosmena, F., 2013. Adaptive capacity in Tanzanian Maasailand: Changing strategies to cope with drought in fragmented landscapes. Global Environmental Change-Human and Policy Dimensions 23: 588-597.

Gough, C. M., Vogel, C. S., Schmid, H. P., Curtis, P. S., 2008. Controls on annual forest carbon storage: Lessons from the past and predictions for the future. Bioscience 58: 609-622.

Guo, L. B., Gifford, R. M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8: 345-360.

Govender, N., Trollope, W.S.W., Van Wilgen, B.W., 2006. The effect of fire season, fire frequency, rainfall and management on fire intensity in savanna vegetation in South Africa. Journal of Applied Ecology 43, 748-758.

Gray, E.F. Bond, W.J., 2009. Carbon sequestration following a biome switch from mesic savanna to forest. SAAB Annual Meeting Abstracts. doi:10.1016/j.sajb.2009.02.056.

Gray, J.M., Humphreys, G.S., Deckers, J.A., 2011. Distribution patterns of World Reference Base soil groups relative to soil forming factors. Geoderma 160: 373-383.

Greiner, C., Alvarez, M. Beker, M., 2013. From Cattle to Corn: Attributes of Emerging Farming Systems of Former Pastoral Nomads in East Pokot, Kenya. Society and Natural Resources 26: 1478-1490.

Grote, R. E. Lehmann, C. Brümmer, N. Brüggemann, J. Szarzynski, and H. Kunstmann. 2009. Modelling and observation of biosphere-atmosphere interactions in natural savannah in Burkina Faso, West Africa. Physics and Chemistry of the Earth 34: 251-260.

Haile, M., 2005. Weather patterns, food security and humanitarian response in Sub-Saharan Africa, Philosophical Transactions: Biological Sciences.360, 1463: 2169-82.

Hanan, N.P., Sea,W.B., Dangelmayr, G. and Govender, N., 2008. Do fires in savannas consume woody biomass? A comment on approaches to modeling savanna dynamics. The American Naturalist, 171: 851-856.

Harris D, Horwath WR, van Kessel C. 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon13 isotopic analysis. Soil Science Society of America Journal 63: 1853-1856

Hengl, T., Mendes de Jesus, J., MacMillan, R.A., Batjes, N.H., Heuvelink, G.B.M., Ribeiro, E.C., Samuel-Rosa, A., Kempen, B., Leenaars, J.G.B., Walsh, M.G., Gonzalez, M.R., 2014. SoilGrids1km— global soil information based on automated mapping. PLoS ONE 9, e105992.

Hengl, T., Heuvelink, G.B.M., Kempen, B., Leenaars, J.G.B., Walsh, M.G., Shepherd, K.D., Sila, A., MacMillan, R.A., Mendes de Jesus, J., Tamene, L., Tondoh, J.E. 2015. Mapping Soil Properties of Africa at 250 m Resolution: Random Forests Significantly Improve Current Predictions. PLoS ONE 10, e0125814. doi: 10.1371/journal.pone.0125814.

Henry, M., Valentini, R., Bernoux, M., 2009. Soil carbon stocks in ecoregions of Africa. Biogeosciences Discuss. 6: 797-823.

Herrick J.E., Urama K.C., Karl J.W., Boos J., Johnson M.V.V., Shepherd K.D., Hempel J., Bestelmeyer B.T., Davies J., Guerra J.L., Kosnik C., Kimiti D.W., Losinyen A.H., Muller E.K., Norfleet L., Ozor N., Reinsch T., Sarukhan J., West L.T., 2013. The global Land-Potential Knowledge System (LandPKS): Supporting evidence-based, site-specific land use and management through cloud computing, mobile applications, and crowdsourcing. Journal of Soil and Water Conservation 68(1): 5A-12A.

Hiernaux, P., Bielders, C. L., Valentin, C., 1999. Effects of livestock grazing on physical and chemical properties of sandy soils in Sahelian rangelands. Journal of Arid Environments, 41: 231–245.

Higgins, S.I. and S. Scheiter. 2012. Atmospheric CO2 forces abrupt vegetation shifts locally, but not globally. Nature 488(7410): 209-212.

Hodgson, D. L., 1999. Images and Interventions: The Problems of Pastoralist Development. In: Anderson, D. M., Broch-Due, V., (Eds.), The Poor are not us: Poverty and Pastoralism in East Africa. Bloomington: Indiana University Press; London: James Currey Plbs. p221-239.

Holechek, J.L., Gomes, H., Molinar, F., Galt, D., Valdez, R., 2000. Short duration grazing, the facts in 1999. Rangelands 22: 18-22.

Homewood, K. M., 2004. Policy, environment and development in African rangelands. Environmental Science & Policy 7: 125-143.

Houghton, R. A., Hobbie, J. E., Melillo, J. M., Moore, B., Peterson, B. J., Shaver, G. R., Woodwell, G. M., 1983. Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: A net release of CO2 to the atmosphere. Ecological Monographs 53:235-262.

Huang X, Senthilkumar S, Kravchenko A, Thelen K, Qi J. 2007. Total carbon mapping in glacial till soils using near-infrared spectroscopy, Landsat imagery and topographical information. Geoderma 141: 34–42

Hulbert, L.C., 1988. Causes of fire effects in tall grass prairie. Ecology 69: 46-58.
IEM, 2009. Integrated Ecosystem Management in the transboundary areas between Nigeria and Niger. Project GEF/GFL-2328-2770-4889. 3rd Annual Progress Report, July 2008 – June 2009.

Huntzinger, D.N., Schwalm,C., Michalak, A.M, Schaefer,K., King, A.W., Wei, Y., Jacobson, A., Liu, S., Cook, R.B., Post, W.M., Berthier, G., Hayes, D., Huang, M., Ito, A., Lei, H., Lu,C., Mao, J., Peng, C.H., Peng, S., Poulter, B.,Riccuito, D., Shi, X., Tian, H., Wang,W., Zeng, N, Zhao. F., and Zhu, Q., 2013. The North American Carbon Multi-Scale Synthesis and Terrestrial Model Intercomparison Project – Part 1: Overview and experimental design. Geoscience Modeling and Development 6: 2121-2133.

IAASTD, 2009. International Assessment of Agriculture Knowledge, Science and Technology for Development. Volume V: Sub-Saharan Africa (SSA) Report. McIntyre, B. D., Herren, H. R., Wakhungu, J., Watson, R. T. (Eds.).

IFPRI and ILRI 2000. Property rights, risk and livestock development in Africa. N. MacCarthy, B Swallow, M. Kirk and P. Hazel (Eds.) Washington DC, International Food Policy Research Institute (IFPRI) and International Livestock Research Institute (ILRI). 433 pp.

IPCC, 2000. Land use, land use change, and forestry: A special report of the IPCC. Cambridge University Press, Cambridge, UK.

IPCC. 2003. Good Practice Guidance for Land Use, Land-Use Change and Forestry, In Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F., (Eds.), Institute for Global Environmental Strategies (IGES), Japan.

IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories. Prepared by the National Greenhouse Gas Inventories Programme. Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.). Japan: IGES.

IRIN, 2007. Africa: Can pastoralism survive in the 21st century? UN Integrated Regional Information Networks. World Press. Available at www.worldpress.org/Africa/2861.cfm (last accessed 22 July 2015).

IUCN, 2010. Building climate change resilience for African livestock in Sub-Saharan Africa World Initiative for Sustainable Pastoralism (WISP): a program of IUCN - The International Union for Conservation of Nature, Eastern and Southern Africa Regional Office, Nairobi, March 2010, viii + 48p. Available from http://cmsdata.iucn.org/downloads/resilence2.pdf. (Last accessed March 16th 2015).

Izac, A.M.N., 1997. Developing policies for soil carbon management in tropical regions. Geoderma 79, 261-276.

Jenny, H., 1941. Factors of Soil Formation: A System of Quantified Pedology. McGraw-Hill, New York (Reprinted in 1994 by Dover Publications, Mineaola, NY).

Jones, C.L., Smithers, N.L., Scholes, M.C., Scholes, R.J., 1990. The effect of fire frequency on the organic components of a basaltic soil in the Kruger National Park. S. Afr. J. Plant Soil 7: 236–237.

Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agroforestry Systems 76: 1-10.

Kay M., 2001 Smallholder irrigation technology: Prospects for Sub-Saharan Africa. International Program for Technology and Research in Irrigation and Drainage Knowledge Synthesis Report No. 3, March 2001. FAO, Rome. Available from http://www.fao.org/3/a-y0969e.pdf. (Last accessed March 13th 2015).

Kamoni, P.T., P.T. Gicheru, S.M. Wokabi, M. Easter, E. Milne, K. Coleman, P. Falloon, and K. Paustian. 2007. Predicted soil organic carbon stocks and changes in Kenya between 1990 and 2030. Agriculture, Ecosystems and Environment 122: 105-113.

Kaonga, M.L. and K. Coleman. 2008. Modelling soil organic carbon turnover in improved fallows in eastern Zambia using the RothC-26.3 model. Forest Ecology and Management 256: 1160-1166.

Kern, J. S., 1994. Spatial patterns of soil organic matter in the contiguous United States. Soil Science Society of America Journal 58: 439-455.

Kiage, L. M., 2013. Perspectives on the assumed causes of land degradation in the rangelands of Sub-Saharan Africa. Progress in Physical Geography 37(5): 664–684.

Kononova, M. M., 1966. Soil Organic Matter. Its Nature, Its Role in Soil formation and in Soil Fertility. 2nd edition. Pergamon Press, Oxford.

Lardy, R., G. Bellocchi, and J.-F. Soussana. 2011. A new method to determine soil organic carbon equilibrium. Environmental Modelling Software 26: 1759-1763.

Lee J, Wopmans JW, Rolston DE, Baer SG, Six J. 2009. Determining soil carbon stock changes: Simple bulk density corrections fail. Agriculture. Ecosystems and Environment 134: 251-256

Lehmann, C.E.R., Anderson, T.M., Sankaran, M., Higgins, S.I., Archibald, S., Hoffmann, W.A., Hanan, N.P., Williams, R.J., Fensham, R.J., Felfili, J., Hutley, L.B., Ratnam, J., San Jose, J., Montes, R., Franklin, D.,Russell-Smith, J., Ryan, C.M., Durigan, G.,Hiernaux, P., Haidar, R., Bowman, D.M.J.S., Bond, W.J., 2014. Savanna Vegetation-Fire-Climate Relationships Differ Among Continents. Science 343: 548-552.

Li, C., S. Frolking, and T.A. Frolking. 1992. A model of nitrous oxide evaluation from soil driven by rainfall events: 1. Model structure and sensitivity. Journal of Geophysical Research 97: 9759-9776.

Limpens, J., Berendse, F., Blodau, C., Canadell, J. G., Freeman, C., Holden, J., Roulet, N., Rydin, H., Schaepman-Strub, G., 2008. Peatlands and the carbon cycle: from local processes to global implications - a synthesis. Biogeosciences 5: 1475-1491.

Liu, S., B. Bond-Lamberty, J.A. Hicke, R. Vargas, S. Zhao, J. Chen, S.L. Edburg, Y. Hu, J. Liu, A.D. McGuire, J. Xiao, R. Keane, W. Yuan, J. Tang, Y. Luo, C. Potter, and J. Oeding. 2011. Simulating impacts of disturbances on forest carbon cycling in North America: processes, data, models, and challenges. Journal of Geophysical Research 116:GOOK08, doi:10.1029/2010JG001585.

Luyssaert, S., Schulze, E. D., Borner, A., Knohl, A., Hessenmoller, D., Law, B. E., Ciais, P., Grace, J., 2008. Old-growth forests as global carbon sinks. Nature 455: 213-215.

Macharia, P.N., Gachene, C.K.K. and Mureithi J.G., 2011. Using Forage Legumes to Improve Soil Fertility for Enhanced Grassland Productivity of Semi-arid Rangelands of Kajiado District, Kenya. In Bationo, A., Waswa, B., Okeyo, J.M., Maina, F., Kihara, J.M. (Eds.), Innovations as Key to the Green Revolution in Africa, 309, Springer Science + Business Media B.V.

Manzoni, S. and A. Porporato. 2009. Soil carbon and nitrogen mineralization: theory and models across scales. Soil Biology & Biochemistry 41:1355-1379.

Marland, G., Boden, T. A., Andres, R. J., 2000. Global, regional, and national CO2 emissions. Trends: A compendium of data on global change. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, Oak Ridge, TN.

Marshall, F., 1998. Early food production in Africa. The Review of Archaeology 19: 47-58.

Marshall, F., Hildebrand, E., 2002. Cattle before crops: The beginnings of food production in Africa. Journal of World Prehistory 16: 99-143.

Materechera, S.A., Mandiringana, O.T., Mbokodi, P.M., Nyamamphene, K., 1998. Organic matter, pH and nutrient distribution in soil layers of a savanna Thornveld subjected to different burning frequencies at Alice in the Eastern Cape. S Afr J Plant Soil. 15: 109–115.

McBratney AB, Mendonça-Santos ML, Minasny B. 2003. On digital soil mapping. Geoderma 117: 3–52

McCabe, J. T., 1990. Turkana pastoralism: A case against the tragedy of the commons. Human Ecology 18: 81-103.

McCabe, J. T., 1994. Mobility and land use among African pastoralists: old conceptual problems and new interpretations. Pages 69-90 *in* E. Fratkin, K. A. Galvin, and E. A. Roth, editors. African Pastoralist Systems: An Integrated Approach. Lynne Rienner Publishers, Boulder, Colorado, USA.

McCabe, J. T., 2004. Cattle Bring Us to Our Enemies. University of Michigan Press, Ann Arbor, USA.

McCabe, J. T., Dyson-Hudson, R., Wienpahl, J., 1999. Nomadic movements. Pages 108-121 in Little M. A. and Leslie P. W. (Eds.). Turkana Herders of the Dry Savanna. Oxford University Press, Oxford, England.

McCarthy, N., Swallow, B., Kirk, M., Hazell, P., 2000. Property Rights, Risk, and Livestock Development in Africa. International Food Policy Research Institute, Washington, D. C., USA.

McCarty GW, Reeves III JB, Follett RF, Kimble JM. 2002. Mid-Infrared and Near-Infrared Diffuse Reflectance Spectroscopy for soil carbon measurment. Soil.Sci. Am. J., 66: 640-646.

McGill, W.B. 1996. Review and classification of ten soil organic matter (SOM) models. In: Powlson, D.S., P. Smith, and J.U. Smith (eds.) Evaluation of soil organic matter models. Springer-Verlag, Berlin, Germany. pp111-132.

McNaughton, S.J., 1979. Grazing as an optimization process: grass-ungulate relationships in the Serengeti. Am. Nat.113: 691-703.

McSherry, M.E., Ritchie M.E., 2013. Effects of grazing on grassland soil carbon: a global review. Global Change Biology 19: 1347–1357.

Melillo, J.M., Borchers, J., Chaney, J., Fisher, H., Fox, S., Haxeltine, A., Janetos, A., Kicklighter, D.W., Kittel, T.G.F., McGuire, A.D., McKeown, R., Neilson, R., Nemani, R., Ojima, D.S., Painter, T., Pan, Y., Parton, W.J., Pierce, L., Pitelka, L., Prentice, C., Rizzo, B., Rosenbloom, N.A., Running, S., Schimel, D.S., Sitch, S., Smith, T., Woodward, I., 1995. Vegetation ecosystem modeling and analysis project - comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate-change and CO_2 doubling. Global Biogeochemical Cycles 9: 407–437.

Milchunas, D. G., Lauenroth, W. K., 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecological Monographs 63: 327-366.

Milne, E., Banwart, S.A., Noellemeyer, E., Abson, D.J., Ballabio, C., Bampa, F., Bationo, A., Batjes, N.H., Bernoux, M., Bhattacharyya, T., Black, H., Buschiazzo, D.E., Cai, Z., Cerri, C.E., Cheng, K., Compagnone, C., Conant, R., Coutinho, H.L.C., de Brogniez, D., Balieiro, F.d.C., Duffy, C., Feller, C., Fidalgo, E.C.C., da Silva, C.F., Funk, R., Gaudig, G., Gicheru, P.T., Goldhaber, M., Gottschalk, P., Goulet, F., Goverse, T., Grathwohl, P., Joosten, H., Kamoni, P.T., Kihara, J., Krawczynski, R., La Scala Jr, N., Lemanceau, P., Li, L., Li, Z., Lugato, E., Maron, P.-A., Martius, C., Melillo, J., Montanarella, L., Nikolaidis, N., Nziguheba, G., Pan, G., Pascual, U., Paustian, K., Piñeiro, G., Powlson, D., Quiroga, A., Richter, D., Sigwalt, A., Six, J., Smith, J., Smith, P., Stocking, M., Tanneberger, F., Termansen, M., van Noordwijk, M., van Wesemael, B., Vargas, R., Victoria, R.L., Waswa, B., Werner, D., Wichmann, S., Wichtmann, W., Zhang, X., Zhao, Y., Zheng, J., Zheng, J., 2015. Soil carbon, multiple benefits. Environmental Development 13: 33-38.

Milne, E., Neufeldt, H., Smalligan, M., Rosenstock, T., Bernoux, M., Bird, N., Casarim, F., Denef, K., Easter, M., Malin, D., Ogle, S., Ostwald, M., Paustian, K., Pearson, T., Steglich, E., 2012. Methods for the quantification of emissions at the landscape level for developing countries in smallholder contexts. CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS), Copenhagen (DK), p. 59.

Milne, E., R. Al-Adamat, N.H. Batjes, M. Bernoux, T. Bhattacharyya, C.C. Cerri, C.E.P. Cerri, K. Coleman, M. Easter, P. Falloon, C. Feller, P. Gicheru, P. Kamoni, K. Killian, D.K. Pal, K. Paustian, D. Powlson, Z. Rawajfih, M. Sessay, S. Williams, and S. Wokabi. 2007. National and sub-national assessments of soil organic carbon stocks and changes: the GEFSOC modelling system. In: Milne, E., Powlson, D.S. and Cerri. C.E.P. (eds.), Soil Carbon Stocks at Regional Scales. Agric. Ecosyst. Environ. 122: 3–12.

Milne, R. and T.A. Brown. 1997. Caron in the vegetation and soils of Great Britain. Journal of Environmental Management 49: 413-433.

Miller, R. W., Donahue, R. L., 1990. Soils. Sixth edition. Prentice Hall, Englewood Cliffs, New Jersey, USA.

Mills, A.J., Fey, M.G, 2003. Declining soil quality in South Africa: Effects of land use on soil organic carbon and surface crusting. South African Journal of Science 99: 429-436.

Moreira CS, Brunet D, Verneyre L, Sa SMO, Galdos MV, Cerri CC, Bernoux M, 2009. Near infrared spectroscopy for soil bulk density assessment. European Journal of Soil Science 60: 785-791

Murty D, Kirschbaum UF, Mcmurtrie RE, Mcgilvray H. 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literature. Global Change Biology 8: 105-123

Nair, P. K. R., Kumar, B. M., Nair, V. D., 2009. Agroforestry as a strategy for carbon sequestration. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 172: 10-23.

Neely, C., Bunning, S., Wilkes, A., 2010. Managing dryland pastoral systems: implications for mitigation and adaptation to climate change.In: Grasslands Carbon Sequestration: management, policy and economics. Proceedings of the workshop on the role of grassland carbon sequestration in the mitigation of climate change. FAO Integrated Crop Management, 11: 235-266 and Conclusion Chapter.

Nie, D., M.M. Krikham, L.K. Ballou, D.J. Lawlor, and E.T. Kanemasu. 1992. Changes in prairie vegetation under elevated carbon dioxide levels and two soil moisture regimes. Journal of Vegetation Science 3: 673-678.

Nelson, D.W. and Sommers, L.E., 1996. Total carbon, organic carbon and organic matter. In Bigham, J.M., Methods of Soil Analysis. Part 3. Chemical Methods-SSSA Book Series no. 5. Chp 34, p961. Pld SSSA, Madison, USA.

NEPAD, 2002. New Partnership for Africa's Development. Comprehensive Africa Agriculture Development Programme. FAO. Available from http://www.fao.org/docrep/005/y6831e/y6831e00.htm#TopOfPage. (Last accessed 11 March 2015).

Niamir-Fuller, M., 1999. Managing mobility in African rangelands. Pages 102-131 in N. McCarthy, B. Swallow, M. Kirk, and P. Hazell, (Eds). Property Rights, Risk and Livestock Development in Africa. International Food Policy Research Institute, International Livestock Research Institute, Washington, D.C., Nairobi.

Nkedianye, D., de Leeuw, J., Ogutu, J. O., Said, M. Y., Saidimu, T. L., Kifugo, S. C., Kaelo, D. S., Reid, R. S., 2011. Mobility and Livestock mortality in communally used pastoral areas: the impact of the 2005-2006 drought on livestock mortality in Maasailand. Pastoralism, Research, Policy, and Practice 1: 1-17.

Nocita M, Stevens A, Wesemael BV, Brown DJ, Shepher KD, Towett, E, Vargas R, Montanarella L. 2015. Soil spectroscopy: an opportunity to be seized. Global Change Biology 21:10-11.

Ogle, S. M., Conant, R. T., Paustian, K., 2004. Deriving grassland management factors for a carbon accounting method developed by the intergovernmental panel on climate change. Environmental Management 33: 474-484.

Ojima, D. S., Parton, W. J., Schimel, D. S., Scurlock, J. M. O., Kittel, T. G. F., 1993. Modeling the effects of climatic and CO2 changes on grassland storage of soil C. Water, Air, and Soil Pollution 70: 643-657.

Ojima, D., Schimel, D.S., Parton, W.J., Owensby, C.E., 1994. Long and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24, 67–84.

Okayasu, T., Okuro, T., Jamsran, U., Takeuchi, K., 2011. Threshold Distinctions Between Equilibrium and Non-equilibrium Pastoral Systems along a Continuous Climatic Gradient. Rangeland Ecology and Management, 64(1): 10-17.

Oldeman, L. R., 1994. The global extent of soil degradation. Pages 99-118 in D. J. Greenland, D. J., and I. Szabolcs, I. (Eds.). Soil Resilience and Sustainable Land Use. CAB International, Wallingford, UK.

Olofsson, J., Hickler. T., 2008. Effects of human land-use on the global carbon cycle during the last 6,000 years. Vegetation History and Archaeobotany 17: 605-615.

Olsen, J., Misana, S., Campbell, D.J., Mbonile, M., Mugisha, S., 2004. The Spatial Patterns and Root Causes of Land Use Change in East Africa. Land use change impacts and dynamics (LUCID), working paper No. 47. Nairobi, Kenya, International Livestock Research Institute.

Oluwole, F.A., Mackson_Sambo, J. and Sikhalazo, D. 2008. Long-term effects of different burning frequencies on the dry savannah grassland in South Africa. African Journal of Agricultural Research Vol. 3 (2): 147-153.

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: 61-65.

Palmer CJ, Smith WD, Conkling BL. 2002. Development of a protocol for monitoring status and trends in forest soil carbon at a national level. Environmental Pollution 116: S209–S219

Parton, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Science Society of America Journal 51: 1173-1179.

Paustian, K., Collins, H. P., Paul, E. A., 1997. Management controls on soil carbon. Pages 15-49 in Paul, E. A., Paustian, K., Elliot, E. T., Cole, C. V. (Eds.). Soil Organic Matter in Temperate Agroecosystems. CRC Press, Boca Raton.

Paustian, K., E. Levine, W.M. Post, and I.M. Ryzhova. 1997a. The use of models to integrate information and understanding of soil C at the regional scale. Geoderma 79: 227-260.

Piao, S. L., Friedlingstein, P., Ciais, P., Peylin, P., Zhu, B., Reichstein, M., 2009. Footprint of temperature changes in the temperate and boreal forest carbon balance. Geophysical Research Letters 36: 5.

Pineiro , G., Paruelo ,J. M., Oesterheld, M., Jobbagy, E.J., 2010. An Assessment of Grazing Effects on Soil Carbon Stocks in Grasslands. Rangeland Ecology & Management, 63(1).
Rass N., 2006. Policies and strategies to address the vulnerability of pastorlists in Sub-Saharan Africa. PPLPI Working paper No. 37. FAO Rome. Available at http://www.fao.org/ag/againfo/programmes/en/pplpi/docarc/wp37.pdf (Last accessed 12 March 2015).

Randerson, J. T., Chapin III, F. S., Harden, J. W., Neff, J. C., Harmon, M. E., 2002. Net ecosystem production: A comprehensive measure of net carbon accumulation by ecosystems. Ecological Applications 12: 937-947.

Reeder, J. D., Schuman, G.E., 2002. Influence of livestock grazing on C sequestration in semiarid mixed-grass and short-grass rangelands. Environmental Pollution, 116: 457–463.

Reeves III, JB.2009. Near- versus mid-infrared diffuse reflectance spectroscopy for soil analysis emphasizing carbon and laboratory versus on-site analysis: where are we and what needs to be done? Geoderma 158: 3-4

Reid, R., Thornton, P., McCrabb, G., Kruska, R., Atieno, F., Jones, P., 2004. Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics? Environment, Development and Sustainability 6: 91-109.

Reid, R., H. Gichohi, M. Said, D. Nkedianye, J. Ogutu, M. Kshatriya, P. Kristjanson, S. Kifugo, J. Agatsiva, S. Adanje, and R. Bagine. 2008. Fragmentation of a peri-urban savanna, Athi-Kaputiei Plains, Kenya. Pages 195-224 *in* K. A. Galvin, R. S. Reid, R. H. Behnke, and N. T. Hobbs, editors. Fragmentation in Semi-Arid and Arid Landscapes: Consequences for Human and Natural Systems. Springer, Dordrecht, The Netherlands.

Reid R., 2012a. Savannas of Our Birth - People, Wildlife, and Change in East Africa.University of California Press, California, USA.Schlesinger, W. H., 1977. Carbon balance in terrestrial detritus. Annual Review of Ecology and Systematics 8: 51-81.

Reid R., 2012b. Savannas of Our Birth - People, Wildlife, and Change in East Africa. University of California Press, California, USA. Chp 2, pp 17-46.

Ritchie, M.E. 2014. Plant compensation to grazing and soil carbon dynamics in a tropical grassland. PeerJ 2:e233:DOI 10.7717/peerj.233.

Ritchie, M. 2014a. Soil carbon dynamics in the northern Kenya rangelands. Progress report to the USAID Africa Biodiversity and Collaborative Group. The Nature Conservancy.

Robinson, T.P., Thornton P.K., Franceschini, G., Kruska, R.L., Chiozza, F., Notenbaert, A., Cecchi, G., Herrero, M., Epprecht, M., Fritz, S., You, L., Conchedda, G., See, L., 2011. Global livestock production systems. Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI) Rome, p. 152.

Rossel RAV, Walvvoort DJJ, McBratney AB, Janik LJ, Skjemstad JO. 2006. Visible, near infrared, mid infrared or combined diffuse reflectance spectroscopy for simultaneous assessment of various soil properties. Geoderma 131: 59-75.

Rossel RAV, Webster R, Bui EN, Baldoc JA. 2014. Baseline map of organic carbon in Australian soil to support national carbon accounting and monitoring under climate change. Global Change Biology 20: 2953–2970.

Sandford, S., 1983. Management of Pastoral Development in the Third World. John Wiley & Sons, Chichester, UK.

Sankaran, M., J. Ratnam, and N. Hanan. 2008. Woody cover in African savannas: the role of resources, fire, and herbivory. Global Ecology and Biogeography 17: 236-245.

Savadogo, P., Sawadogo, L., Tiveau, D., 2007. Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. Agriculture, Ecosystems and Environment 118: 80–92.

Savory, A., Butterfield, J., 1999. Holistic Management: A new framework for decision making. Pld. Island Press, Washington DC.

Scholes, M., Andreae, M.O., 2000. Biogenic and Pyrogenic Emissions from Africa and Their Impact on the Global Atmosphere. Ambio, 29: 23-29.

Seré, C., Steinfeld, H., 1996. World livestock production systems. Current status, issues and trends, Food and Agriculture Organization of the United Nations, Rome, Italy.

Shepherd KD, Walsh MG. 2007. Infrared spectroscopy-enabling an evidence-based diagnostic surveillance approach to agricultural and environmental management in developing countries. Journal of Near Infrared Spectroscopy 15: 1-19.

Shepherd KD, Shepherd G and Walsh MG. 2015. Land health surveillance and response: a framework for evidence-informed land management. Agricultural Systems 132: 93–106

Shevliakova, E., Pacala, S. W., Malyshev, S., Hurtt, G. C., Milly, P. C. D., Caspersen, J. P., Sentman, L. T., Fisk, J. P., Wirth, C., Crevoisier, C., 2009. Carbon cycling under 300 years of land use change: Importance of the secondary vegetation sink. Global Biogeochemical Cycles 23: 16.

Sitch, S., P. Friedlingstein, N. Gruber, S.D. Jones, G. Murray-Tortarolo, A. Ahiström, et al. 2013. Trends and drivers of regional sources and sinks of carbon dioxide over the past two decades. Biogeosciences Discussions 10:20113-20177.

Smith, P., J.U. Smith, D.S. Powlson, W.B. McGill, J.R.M. Arah, O.G. Chertov, K. Coleman, U. Franko, S. Frolking, D.S. Jenkinson, L.S. Jensen, R.H. Kelly, H. Klein-Gunnewiek, A.S. Komarov, C. Li, J.A.E. Molina, T. Mueller, W.J. Parton, J.H.M. Thornley, and A.P. Whitmore. 1997. A comparison of the performance of nine soil organic matter models using datasets from seven long-term experiments. Geoderma 81: 153-225.

Smith, P., Falloon, P., Coleman, K., Smith, J., Piccolo, M. C., Cerri, C. C., Bernoux, M., Jenkinson, D., Ingram, J., Szabo, J., Pasztor, L., 1999. Modeling soil carbon dynamics in tropical ecosystems. Pages 341-364 *in* Lal, R., Kimble, J., Levine, E., Stewart, B. A. (Eds.). Global climate change and tropical ecosystems. CRC Press, Boca Raton, FL, USA.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., 2007. Agriculture. Pages 499-540 *in* Metz, B., Davidson, O. R., Bosch, P. R., Dave, R., Meyer, L. A. (Eds.). Climate Change 2007: Mitigation. Contribution of Working Group III to the

Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B., Sirotenko, O., Howden, M., McAllister, T., Pan, G., Romanenkov, V., Schneider, U., Towprayoon, S., Wattenbach, M., Smith, J., 2008. Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B-Biological Sciences 363: 789-813.

Smith JU, Gottschalk P, Bellarby J., Chapman S., Lilly, A, Towers, W, Bell, B, Coleman, K., Nayak, D., Richards, M., Hillier, J., Flyn, H.[1], Wattenbach, M., Aitkenhead, M., Yeluripati, J., Farmer, J., Milne, R, Thomson, A, Evans, C, Whitmore, A, Falloon, P, Smith, P., 2010. Estimating changes in national soil carbon stocks using ECOSSE-a new model that includes upland organic soils. Part I. Model description and uncertainty in national scale simulations of Scotland. Climate Research 45: 179–192.

Smith, J., A. Abegaz, R.B. Matthews, M. Subedi, E.R. Orskov, V. Tumwesige, and P. Smith. 2014. What is the potential for biogas digesters to improve soil carbon sequestration in Sub-Saharan Africa? Comparison with other uses of organic residues. Biomass and Bioenergy 70: 73-86.

Sombroek, W. G., Nachtergaele, F. O., Hebel, A., 1993. Amounts, dynamics and sequesteting of carbon in tropical and subtropical soils. Ambio 22: 417-426.

Stephens, B. B., Gurney, K. R., Tans, P. P., Sweeney, C., Peters, W., Bruhwiler, L., Ciais, P., Ramonet, M., Bousquet, P., Nakazawa, T., Aoki, S., Machida, T., Inoue, G., Vinnichenko, N., Lloyd, J., Jordan, A., Heimann, M., Shibistova, O., Langenfelds, R. L., Steele, L. P., Francey, R. J., Denning, A. S., 2007. Weak northern and strong tropical land carbon uptake from vertical profiles of atmospheric CO2. Science 316: 1732-1735.

Still, C.J., J.A. Berry, G.J. Collatz, and R.S. DeFries. 2003. Global distribution of C3 and C4 vegetation: carbon cycle implications. Global Biogeochemical Cycles 17(1):1006, doi:10.1029/2001GB001807

Stockman, U., M.A. Adams, J.W. Crawford, D.J. Field, N. Henakaarchchi, M. Jenkins, B. Minasny, et al. 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystems and Environment 164: 80-99.

Tarnocai, C., Canadell, J. G., Schuur, E. A. G., Kuhry, P., Mazhitova, G., Zimov, S., 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochemical Cycles 23.

Tate, R. L., 1987. Soil Organic Matter. Biological and Ecological Effects. John Wiley & Sons, New York.

Teague, W.R., Dowhower, S.L., Baker, S.A., Haile, N., DeLaune, P.B., Conover, D.M., 2010. Agriculture, Ecosystems and Environment. 141: 310-322.

Tefera, S., Snyman, H.A., Smit G.N., 2007. Rangeland dynamics in southern Ethiopia: (1) Botanical composition of grasses and soil characteristics in relation to land-use and distance from water in semi-arid Borana rangelands. Journal of Environmental Management, 85: 429–442.

Thairu, D.H., Tessema, S., 1987. Research on animal feed resources: Medium-potential areas of Kenya. In: Kategile, J.A., Said, A.N., Dzowela, B.H. (Eds). Animal feed resources for small-scale livestock producers. Proceedings of the second PANESA workshop, held in Nairobi, Kenya, 1115 November 1985 IDRC manuscript report Canada. pp. 125-148.

Thomas, D., 2006. Crop–animal systems in the tropics. In: Thornton, P., (Ed), Trans-Regional Analysis of Crop–Livestock Systems, CABI Publishing, Wallingford.

Thornton P.K., K.A. Galvin, and R.B. Boone. 2003. An agro-pastoral household model for the rangelands of East Africa. Agricultural Systems 76: 601-622.

Thornton, P.K., R.H. Fawcett, K.A. Galvin, R.B. Boone, J.W. Hudson, and C.H. Vogel. 2004. Evaluating management options that use climate forecasts: modelling livestock production systems in the semi-arid zone of South Africa. Climate Research 26: 33-42.

Thornton, P.K., R.B. Boone, K.A. Galvin, S.B. BurnSilver, M.M. Waithaka, J. Kuyiah, S. Karanja, E. Gonzalez-Estrada, and M. Herrero. 2007. Coping strategies in livestock-dependent households in East and southern Africa: a synthesis of four case studies. Human Ecology 35: 461-476.

Tieszen, L.L., Senyimba, M.M., Imbamba, S.K. and Troughton, J.H., 1979. The distribution of C3 and C4 grasses and carbon isotope discrimination along an altitudinal and moisture gradient in Kenya. Oecologia (Berl.) 37: 337-350.

Traore, A.K., P. Ciais, N. Vuichard, B. Poulter, N. Viovy, M. Guimberteau, M. Jung, R. Myneni, and J.B. Fisher. 2014. Evaluation of the ORCHIDEE ecosystem model over Africa against 25 years of satellite-based water and carbon measurements. Journal of Geophysical Research: Biogeosciences 119: 1554-1575.

Tredennick, A.T. Hanan, N.P. ,2015. Effects of Tree Harvest on the Stable-State Dynamics of Savanna and Forest. The American Naturalist 185(5): E153-E165.

Tsegaye, D., Vedeld, P., Moe, S. R., 2013. Pastoralists and livelihoods: A case study from northern Afar, Ethiopia. Journal of Arid Environments 91: 138-146.

Turner, M. D., 1999. The role of social networks, indefinite boundaries and political bargaining in maintaining the ecological and economic resilience of the transhumance systems of Sudano-Sahelian West Africa. Pages 97–123 in Niamir-Fuller, M. (Eds). Managing mobility in African rangelands. Intermediate Technology Publications, London, UK.

Turner, M. D. 2011. The new pastoral development paradigm: engaging the realities of property institutions and livestock mobility in dryland Africa. Society and Natural Resources 24: 469-484.

UNCCD 2012. Zero net land degradation. A sustainable development goal for Rio 20+. UNCCD Secretariat Policy Brief. Available at: www.unccd.int/Lists/SiteDocumentLibrary/Rio+20/UNCCD_PolicyBrief_ZeroNetLandDegradation.pdf (Last accessed 22 July 2015).

UNEP. 2012. Land Health Surveillance: An evidence-based approach to land ecosystem management, illustrated with a Case Study in the West Africa Sahel. United Nations Environment Program. Nairobi, Kenya.

UNFCCC. 2006. Approved afforestation and reforestation baselines methodology AR-AM0002: "Restoration of degraded lands through afforestation/reforestation"

Upton, M., 2012. Is Africa different? Economics of the livestock sector. GalvMed Abstracts. Available at http://www.galvmed.org/wp-content/uploads/2013/07/Abstract-paper-1-Upton-Final-18June13.pdf. (Last accessed March 16th 2015).

Vågen, T.-G., Lal, R., Singh, B.R., 2005. Soil carbon sequestration in Sub-Saharan Africa: a review. Land Degrad. Develop., 16: 53–71.

Vågen T-G, Davey FA, Shepherd KD. 2012. Land health surveillance: Mapping soil carbon in Kenyan rangelands. In Nair PKR and Garrity D (Eds) Agroforestry- The global land use, Advances in Agroforestry pp 455-462

Vågen T-G, Winowiecki L, Abegaz A, Hadgu KM. 2013. Landsat-based approaches for mapping of land degradation prevalence and soil functional properties in Ethiopia. Remote Sensing of Environment 134: 266-275

Vågen, T.-G., Leigh, A.W., 2013. Mapping of soil organic carbon stocks for spatially explicit assessments of climate change mitigation potential. Environmental Research Letters 8: 015011.

van Wesemael, B., Paustian, K., Andrén, O., Cerri, C.E.P., Dodd, M., Etchevers, J., Goidts, E., Grace, P., Kätterer, T., McConkey, B.G., Ogle, S., Pan, G., Siebner, C., 2011. How can soil monitoring networks be used to improve predictions of organic carbon pool dynamics and CO2 fluxes in agricultural soils? Plant and Soil 338: 247-259.

VEMAP Members (J.M. Melillo, J. Borchers, J. Chaney, H. Fisher, S. Fox, A. Haxeltine, A. Janetos, D.W. Kicklighter, T.G.F. Kittel, A.D. McGuire, R. McKeown, R. Neilson, R. Nemani, D.S. Ojima, T. Painter, Y. Pan, W.J. Parton, L. Pierce, L. Pitelka, C. Prentice, B. Rizzo, N.A. Rosenbloom, S. Running, D.S. Schimel, S. Sitch, T. Smith, I. Woodward). 1995. Vegetation/Ecosystem Modeling and Analysis Project (VEMAP): Comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and CO2 doubling. Global Biogeochemical Cycles 9: 407-437.

Vetter, S., 2005. Rangelands at equilibrium and non-equilibrium: recent developments in the debate. Journal of Arid Environments 62: 321-341.

Vital Signs Protocol. 2014. E-Plot Soil Sampling and Processing. Version 3.0 Available at http://vitalsigns.org/files/VS_Protocol_Soils_March2014_update.pdf. (Last accessed 22 July 2015).

von Wehrden, H., Hanspach, J., Kaczensky, P., Fischer, J., Wesche, K., 2012. Global assessment of the non-equilibrium concept in rangelands. Ecological Applications 22(2): 393–399.

Walkley A, Black IA. 1934. An examination of the Degtjareff method for determining organic carbon in soils: Effect of variations in digestion conditions and of inorganic soil constituents. Soil Sci. 63: 251-263.

Waswa BW, Vlek PLG, Tamene LD, Okoth P, Mbakaya D, Zingore S. 2013. Evaluating indicators of land degradation in smallholder farming systems of western Kenya, Geoderma 195-196: 192–200.

Weber, U., M. Jung, M. Reichstein, C. Beer, M.C. Braakhekke, V. Lehsten, D. Ghent, J. Kaduk, N. Viovy, P. Ciais, N. Gobron, and C. Rödenbeck. 2009. The interannual variability of Africa's ecosystem productivity: a multi-model analysis. Biogeosciences 6: 285-295.

Weber, K.T. Gokhale, B.S., 2011. Effect of grazing on soil-water content in semiarid rangelands of southeast Idaho. Journal of Arid Evironments 75: 464-470.

West, A.G., G.F. Midgley, and W.J. Bond. 2012. The reforestation of Africa? South African Journal of Science 108:1448. http://dx.doi.org/10.4102/sajs.v108i11/12.1448.

Western, D., 1982. The environment and ecology of pastoralists in arid savannas. Development and Change 13:183-211.

Williams, C.A., Hanan, N.P., Neff, J.C., Scholes, R.J., Berry, J.A., Denning, A.S. et al., 2007. Africa and the global carbon cycle. Carbon Balance and Management*,* 2(March 2007): Art. No. 3, Available at http://www.cbmjournal.com/articles/browse.asp). doi:10.1186/1750-0680-2-3.

Wilson, R.T., de Leeuw, P.N. and de Haan, C., 1983. Systems research in the arid zones of Mali, preliminary results (Research Report No. 5). International Livestock Centre for Africa, Addis Ababa, Ethiopia, 189 pp.

Wilts, A. R., Reicosky, D. C., Allmaras, R. R., Clapp, C. E., 2004. Long-Term Corn Residue Effects: Harvest Alternatives, Soil Carbon Turnover, and Root-Derived Carbon. Soil Sci Soc Am J 68: 1342-1351.

Winrock International, 1992. Assessment of animal agriculture in Africa. Winrock International, Morilton, Arkansas, USA.

WOCAT, 2008. A framework for documentation and evaluation of sustainable land management: technologies basic questionnaire. World Overview of Conservation Approaches and Technologies. Available at http://www.wocat.net (Last accessed 22 July 2015).

World Bank, 2008. World Development Report 2008. Agriculture for Development, Washington, DC. Available at http://siteresources.worldbank.org/INTWDR2008/Resources/WDR_00_book.pdf. (Last accessed March 16th 2015).

Woomer, P.L., Toure, A., Sall, M., 2004. Carbon stocks in Senegal's Sahel Transition Zone. Journal of Arid Environments 59: 499–510.

WRI, 2005. The wealth of the poor: Managing ecosystems to fight poverty. World Resources Institute, Washington DC.

Wynn JG, Bird MI, Vellen, L, Grand-Clement E, Carter J, Berry SL. 2006. Continental-scale measurement of the soil organic carbon pool with climatic, edaphic, and biotic controls 20, GB1007, doi:10.1029/2005GB002576.

Yayneshet, T., Treydte, A.C.,2015. A meta-analysis of the effects of communal livestock grazing on vegetation and soils in Sub-Saharan Africa. Journal of Arid Environments, 116: 18-24.

Appendix 1. Key points by citation

Introduction	
Citation	**Key point relevant to this study**
Batjes, 2004	Estimated that grasslands in Africa have the potential to sequester between 7 – 42 Tg C $year^{-1}$.
Follet and Reed, 2010	It has been estimated that grazing lands account for a quarter of potential carbon (C) sequestration in world soils.
Gerber et al., 2013	Grazing lands have an estimated sequestration rate of ~ 0.6 Gt CO2 equivalents yr^{-1}.
IFPRI and ILRI, 2000	There are an estimated 25 million pastoralists and 240 million agro-pastoralists in Sub-Saharan Africa.
Neely et al., 2010	Grazing lands support some of the world's poorest people.
Petri et al., 2010	The world's land surface is one third (31%) grasslands, shrublands or savannah.
van de Steeg et al., 2013	Climate change is already having an impact (changing rainfall and temperature regimes) in arid and semi-arid regions of Sub-Saharan Africa where livelihoods are mainly supported by livestock with future impacts predicted to be most negative for grasslands around Senegal and in Southern Africa, where Namibia, Botswana and South Africa meet.
Thornton et al., 2002	Sub-Saharan Africa is home to 23% of the world's poor the majority of who depend on livestock for some part of their livelihoods.
World Bank, 2007	Livestock is growing as a sector, accounting for up to 50% of GDP in countries with significant areas of rangeland.

Chapter 1. Current types of grazing lands in Sub-Saharan Africa and associated management practices	
Citation	**Key point relevant to this study**
AU-IBAR, 2012	Controlled grazing management practice is considered beneficial in conditions of poor vegetation cover, overgrazing and degraded soils, and is considered as the most promising sustainable land management practice to restore degraded rangelands as it enhances the vigor of mature perennial grasses.
Behnke, 1999	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Bekure et al., 1991	Cases where the fields and the grazing areas may be more distant from each other and grazing more extensive with less frequent return to the central place are becoming more common in East Africa wherever rainfall is sufficient to support production of a crop fairly consistently and where there is movement towards privatization of the lands.
Boone and Hobbs, 2004	Pastoral systems have been developed to sustain life in rangelands where vegetation and water resources are often ephemeral in time and patchy in space, and pastoral movement to these ephemeral patches is a critical strategy to improve milk and meat production.
Brottem et al., 2014	In West Africa there exist transhumant systems in which livestock

	are taken north into the Sahel during the wet season to graze and then return south to the higher rainfall zones where they graze on crop residues.
Butt et al., 2009	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Coppock, 1990	There are also agro-pastoral systems in Eastern Africa, but there, the movements of livestock are not transhumant.
Devendra et al., 2005	Mixed crop-livestock systems, where crops and livestock are integrated on the same farm, are widespread in rain-fed systems and form the backbone of smallholder agriculture.
Ellis and Galvin, 1994	In very general terms it can be said that historically pastoralism dominates in the drylands of eastern Africa with limited crop-livestock integration and agropastoralism in the dryland ecosystems of west Africa and that this can be attributed partially to bi-modal versus uni-modal rainfall patterns.
Ellis et al., 1987	In the drier areas of northern Kenya and Southern Ethiopia, crop agriculture is impossible or unreliable and the pastoral scene is dominated by extensive nomadic pastoralism as in Turkana and Borana.
Ellis and Swift, 1988	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Fernandez-Gimenez, 2002	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Fernandez-Gimenez and Le Febre, 2002	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Goldman and Riosmena, 2013	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Greiner et al., 2013	Agropastoralism is growing in Eastern Africa in areas traditionally used for nomadic pastoralism which are becoming ever more fragmented by the encroachment of cropland.
Hodgson, 1999	In Sub-Saharan Africa, much of the rangeland is utilized by nomadic pastoralists who move their herds and flocks over extensive areas during the course of the year to find water and grazing for their livelihood.
Homewood, 2004	Most rural people living in SSA rangelands are agro-pastoral, combining small-scale farming with livestock keeping, or specialize in herding (pastoralists) or farming; there are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
IAASTD, 2009	Over 60% of the population of Sub-Saharan Africa depends on agriculture for their livelihood and agriculture accounted for on average 29% of GDP between the years 1998 to 2000.
Kiage, 2013	Soil erosion and land degradation represent a major threat to food security in SSA rangelands.
Marshall, 1998	Pastoral institutions have developed over the last 7000 years, since pastoralists became the first people to produce food in much of Sub-Saharan Africa.

Marshall and Hildebrand, 2002	Pastoral institutions have developed over the last 7000 years, since pastoralists became the first people to produce food in much of Sub-Saharan Africa.
McCabe, 1990	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
McCabe, 1994	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
McCabe, 2004	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
McCabe et al., 1999	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
McCarthy et al., 2000	The ability of the land to sustain increasing numbers of livestock owners without damaging the environment will be determined in part by the way the users themselves can govern access and use of this vital resource.
Niamir-Fuller, 1999	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Nkedianye et al., 2011	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Okayasu et al., 2011	Studies suggests that most arid and semi-arid rangeland systems exhibit both equilibrium and non-equilibrium states at different scales, which implies a management approach that takes into account temporal variability and spatial heterogeneity.
Oldeman, 1994	Approximately 65% of agricultural land, 35% of permanent pastures and 19% of forest and woodland in the region are estimated to be affected by some form of degradation.
Reid et al., 2008	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods, still being negotiated today in the face of major developmental changes.
Robinson et al., 2011	Map: Livestock production systems in SSA
Sandford, 1983	Pastoral systems are one of the dominant forms of livestock production system in SSA and are found mainly in arid and semi-arid areas and limited areas in the sub-humid zones in East Africa and West Africa.
Seré and Steinfeld, 1996	Ago-pastoralism is found in the semiarid, sub-humid and humid tropics and in tropical highland areas.
Smith et al., 2007	Proposed measures for mitigating greenhouse gas emissions from grazing lands and livestock production systems, their apparent effects on reducing emissions of individual gases where adopted (mitigative effect), and gave an estimate of scientific confidence that the proposed practice can reduce overall net emissions at the site of adoption.
Thomas, 2006	Mixed crop-livestock systems, where crops and livestock are integrated on the same farm, are widespread in rain-fed systems and form the backbone of smallholder agriculture.
Tsegaye et al., 2013	There are traditional rules concerning grazing movement to sustain

	grazing productivity and thus the resilience of pastoral livelihoods.
Turner, 1999	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Turner, 2011	There are traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods.
Vetter, 2005	Studies suggests that most arid and semi-arid rangeland systems exhibit both equilibrium and non-equilibrium states at different scales, which implies a management approach that takes into account temporal variability and spatial heterogeneity.
Wilson et al., 1983	Pastoral systems are one of the dominant forms of livestock production system in SSA and are found mainly in arid and semi-arid areas and limited areas in the sub-humid zones in East Africa and West Africa.
WOCAT, 2008	Grazing lands are defined as "land used for animal production e.g. natural or semi- natural grasslands, open woodlands, improved or planted pastures".
WRI, 2005	Approximately 65% of agricultural land, 35% of permanent pastures and 19% of forest and woodland in the region are estimated to be affected by some form of degradation.

Chapter 2. Analysis of the key determinants of C sequestration in grass/rangeland systems in different conditions	
Citation	**Key point relevant to this study**
Adams et al., 2009	Disturbance, such as fire, drought, or disease, can lead to substantial losses of carbon from both soils and vegetation.
Allison, 1973	Soil organic matter is an important source of plant nutrients and can enhance production, increase soil aggregation, limit soil erosion, and increase cation exchange and water holding capacities.
Bai et al., 2008	The Land Degradation Assessment in Drylands (LADA) concluded that about 16% of rangelands are currently undergoing degradation and that rangelands comprise 20-25% of total land area currently being degraded; with current degradation likely occurring in addition to historic degradation.
Brandt et al., 2010	Limited studies in the US have shown that photodegradation plays an important role in the decomposition process but the interaction with aridity is not well understood.
Bridges and Oldeman, 1999	Nearly 270 Mha of grassland worldwide have been degraded to some degree by mismanagement.
Chapman and Lemaire, 1993	Primary production in overgrazed grasslands can decrease if herbivory decreases plant growth capacity, vegetation density, community biomass, or, if community composition changes.
Ciais et al., 2005	Disturbance, such as fire, drought, or disease, can lead to substantial losses of carbon from both soils and vegetation.
Conant et al., 2015	Improved grazing management (management that increases production), leads to an increase of soil carbon stocks by an average

	of 0.35 t C ha-1 yr-1.
Conant et al., 2001	Grassland management to enhance production (through sowing improved species, irrigation or fertilization), minimize negative impacts of grazing, or rehabilitate degraded lands can each lead to carbon sequestration; adding manure to soil builds soil organic matter in grasslands.
Conant and Paustian, 2002	Removal of large amounts of aboveground biomass, continuous heavy stocking rates, and other poor grazing management practices are important human-controlled factors that influence grassland production and have led to depletion of soil carbon stocks.
Craine et al., 2008	The co-limitation of N and P in grasslands, particularly those dominated by C4 species (as is the case in SSA) is not well understood.
DeFries et al., 1999	Emissions from deforestation have dominated carbon losses from terrestrial ecosystems.
Donigan et al., 1994	Soil organic matter losses due to conversion of native grasslands to cultivation are both extensive and well documented.
FAO, 2007	Integrating data on grassland area and grassland soil carbon stocks results in a global estimate of about 343Bt C – nearly 50% more than is stored in forests worldwide.
FAO, 2009	Over the last decade grassland area has been growing declining while arable land area has been growing, suggesting continued conversion of grassland to croplands.
Field et al., 2007	Future carbon balance in ecosystems will be affected by a climate, though net global response of ecosystems remains uncertain due in part to several feedbacks.
Follett et al., 2001a	When management practices that deplete soil carbon stocks are reversed, grassland ecosystem carbon stocks can be rebuilt, sequestering atmospheric CO_2.
Follett et al., 2001b	Grassland management to enhance production (through sowing improved species, irrigation or fertilization), minimize negative impacts of grazing, or rehabilitate degraded lands can each lead to carbon sequestration.
Friedlingstein et al. 2006	Carbon stored in vegetation and soil is vulnerable to loss at warmer temperatures.
Gough et al., 2008	In undisturbed ecosystems, the carbon balance tends to be positive: carbon uptake through photosynthesis exceeds losses from respiration; even mature, old-growth forest ecosystems take up slightly more carbon than they release.
Guo and Gifford, 2002	When grasslands are converted to agricultural land, soil carbon stocks tend to decline by an average of about 60%.
Houghton et al. 1983	Deforestation, degradation of native grazing lands, and conversion to cropland have prompted losses of 450-800 Gt CO_2 from biomass and soil carbon pools – equivalent to 30-40% of cumulative fossil fuel emissions.
IPCC, 2000	Improved management in existing agroforestry systems could sequester 0.012 Tg C yr^{-1} while conversion of 630 Mha of

	unproductive or degraded croplands and grasslands to agroforestry could sequester as much as 0.59 Tg C annually by 2040, which would be accompanied by modest increases in N_2O emissions as more nitrogen circulate in the system.
Jose, 2009	When agroforestry systems are introduced in suitable locations, carbon is sequestered in the tree biomass and tends to be sequestered in the soil as well.
Kern, 1994	Historically, intensive use of grasslands has resulted in the transfer of 993 Mt of soil carbon to the atmosphere in the form of CO_2 just from land use change in the United States.
Kononova, 1966	Soil organic matter is an important source of plant nutrients and can enhance production, increase soil aggregation, limit soil erosion, and increase cation exchange and water holding capacities.
Limpens et al. 2008	Whether disturbances decline, mitigating future carbon losses, or the scope of human-induced disturbances continues to expand in the future is a very important determinant of future concentrations of atmospheric $CO_{2.}$
Luyssart et al. 2008	In undisturbed ecosystems, the carbon balance tends to be positive: carbon uptake through photosynthesis exceeds losses from respiration; even mature, old-growth forest ecosystems take up slightly more carbon than they release.
Marland et al. 2000	Deforestation, degradation of native grazing lands, and conversion to cropland have prompted losses of 450-800 Gt CO_2 from biomass and soil carbon pools – equivalent to 30-40% of cumulative fossil fuel emissions.
Milchunas and Lauenroth, 1993	It is widely accepted that continuous excessive grazing is detrimental to plant communities and soil carbon.
Miller and Donahue, 1990	Soil organic matter is an important source of plant nutrients and can enhance production, increase soil aggregation, limit soil erosion, and increase cation exchange and water holding capacities.
Nair et al., 2009	Agroforestry enhances carbon uptake by lengthening the growing season, expanding the niches from which water and soil nutrients are drawn, and in the case of nitrogen-fixing species, enhancing soil fertility.
Ogle et al., 2004	Grassland management to enhance production (through sowing improved species, irrigation or fertilization), minimize negative impacts of grazing, or rehabilitate degraded lands can each lead to carbon sequestration.
Ojima et al., 1993	Removal of a large amounts of aboveground biomass, continuous heavy stocking rates, and other poor grazing management practices are an important human-controlled factors that influences grassland production and haves led to depletion of soil carbon stocks.
Oldeman, 1994	As a result of past practices, somewhere between five and ten percent of the world's grasslands have been degraded by overgrazing; previous research has documented that improved grazing management could lead to greater forage production, more

	efficient use of land resources, and enhanced profitability and rehabilitation of degraded lands.
Olofsson and Hickler, 2008	Deforestation, degradation of native grazing lands, and conversion to cropland have prompted losses of 450-800 Gt CO_2 from biomass and soil carbon pools – equivalent to 30-40% of cumulative fossil fuel emissions.
Page et al., 2002	Disturbance, such as fire, drought, or disease, can lead to substantial losses of carbon from both soils and vegetation.
Paustian et al., 1997	Just as is the case for forest biomass carbon stocks, grassland soil carbon stocks are susceptible to loss upon conversion to other land uses.
Piao et al., 2009	Carbon stored in vegetation and soil is vulnerable to loss at warmer temperatures.
Randerson et al., 2002	Disturbance is a defining element of all ecosystems that continues to influence the carbon uptake and losses that determine long-term ecosystem carbon balance.
Reid et al., 2004	Much of the world's grazing lands are under pressure to produce more livestock by grazing more intensively, particularly in Africa's rangelands, which are vulnerable to climate change and expected to still supply most of the beef and milk demanded in Africa.
Schlesinger, 1977	One of the reasons for the intensive use of grasslands is the high natural soil fertility. Grasslands characteristically have high inherent soil organic matter content averaging 331 Mg ha-1.
Shevliakova et al., 2009	Substantial amounts of carbon have been lost from biomass and soils of systems other than forests as well.
Smith et al., 1999	Carbon dioxide is produced when biomass is burned, and soil carbon stocks begin to decline soon after disturbances to the soil.
Smith et al., 2007	Globally an estimated 0.2-0.8 Gt CO_2 yr^{-1} could be sequestered in grassland soils by 2030 given prices for CO_2 of 20-50 $US per tonne.
Smith et al., 2008	Disturbance through overgrazing, fire, invasive species, etc. can deplete grassland systems of carbon stocks; adding manure or biosolids to soil could sequester between 0.42 and 0.76 t C ha^{-1} yr^{-1} depending on region (sequestration rates tend to be greater in moist regions than in dry).
Sombroek et al., 1993	Integrating data on grassland area and grassland soil carbon stocks results in a global estimate of about 343Bt C – nearly 50% more than is stored in forests worldwide.
Stephens et al., 2007	In undisturbed ecosystems, the carbon balance tends to be positive: carbon uptake through photosynthesis exceeds losses from respiration; even mature, old-growth forest ecosystems take up slightly more carbon than they release.
Tarnocai et al., 2009	Carbon stored in soil: at least 1600 Pg C, but perhaps as much as twice that.
Tate, 1987	Soil organic matter is an important source of plant nutrients and can enhance production, increase soil aggregation, limit soil erosion, and increase cation exchange and water holding capacities.
West and Post, 2002	Rotations with grass, hay, or pasture tend to have the largest

	impact on soil carbon stocks.
Wilts et al., 2004	Harvesting a large proportion of plant biomass enhances yields of useful material (e.g., for forage or fuel), but decreases carbon inputs to the soil.

Chapter 3. Carbon impacts of grazing land management systems and practices in Sub-Saharan Africa	
Citation	**Key point relevant to this study**
Andreae and Merlet, 2001	The amount of aboveground C removed by fire depends on many interacting factors including vegetation type, climate and fire regime.
Angassa, 2012	Found that light to moderate grazing increased herbaceous species diversity whereas heavy grazing, without ‘overgrazing’ reduced it in rangelands in Borana, Ethiopia
Ardo and Olson, 2003	In modeling carbon dynamics in semi- arid Sudan found that whilst high grazing intensity decreased SOC on grasslands it increased SOC in savannas.
Bagchi and Richie, 2010	Found SOC to be 49% lower in areas being grazed by livestock when compared to areas being grazed by native herbivores despite grazing intensities being comparable, attributing the difference to the different diets of the animals.
Bennett, 2000	In Africa more than 300 million people live under severe water restrictions, the majority of whom rely on grazing lands to support them.
Beukes and Cowling, 2003	Looked at the impacts of short periods of high intensity grazing on SOC in the Nama Karoo district South Africa and found this type of grazing reduced organic carbon in the topsoil which they attributed to increased microbial activity accelerating turnover of soil organic matter.
Bird et al., 2000	Working on a long term (52 yr) trial in Zimbabwean savanna, found significant SOC losses in the upper soil layer (0-5cm) after annual burning with a decrease of 20-25% compared with pre-fire levels and found annual burning reduced SOC by almost twice as much in sandy soils compared to clay soils.
Bond et al., 2004	In the longer term, fires restrict seedling survival and thus eventually reduce adult populations and thus biomass and changes in fire regime may therefore be associated with long-term change in carbon stocks in biomass.
Bond and Keely, 2005	Burning is a widespread management practice and indeed a natural and necessary phenomenon in many grass and shrublands.
Bombelli et al., 2009	The CarboAfrica Project estimated GHG from fires in different biomes in SSA including grasslands and savannas and estimated C losses from fires in savannas in SSA as 0.79 Pg C y^{-1}, concluding that the savannas of SSA are currently a net C sink.

Chapter 3. Continued.	
Citation	**Key point relevant to this study**
Briske et al., 2011	Examined the debate surrounding rotational grazing and concluded that there were no significant benefits of rotational grazing over continuous grazing.
Catley et al., 2013	Fencing or protecting an area to exclude grazing animals is increasingly being used in parts of SSA (particularly Ethiopia) as an adaptive strategy to provide livestock fodder in times of stress.
Ciais et al.,2011	African ecosystems currently contribute an estimated 40 % of GHG emissions from fire, mostly from savanna burning.
Conant and Paustian; 2002	Estimated that 10.4% of grasslands in Africa are 'overgrazed' with most being located in Eastern and southern Africa and that in dryland landscapes, heavy grazing can increase soil erosion by leaving bare patches of soil (prone to wind and water erosion) and by compacting the soil which increases the amount and intensity of runoff.
Derner et al., 2006	Showed that in grasslands of North America, grazing increased SOC stocks at 0-30 cm in dry shortgrass steppe ecosystems and decreased stocks in more humid mid- and tallgrass prairie ecosystems.
Derner and Schuman, 2007	On reviewing precipitation and grazing effects on SOC in rangelands concluded that soil carbon sequestration at 0-30 cm, was increased only at sites with a mean annual precipitation of 600 mm or less which they attributed to changes in the amount of root material in the upper soil profile.
Diaz, 2010	Potential for C sequestration in soil inorganic carbon (SIC) through changes in land management is deemed to be low with processes being extremely slow and poorly understood.
Dohn et al., 2013	Fires don’t typically carry in the drier savannas (e.g. the Sahel), and tree-grass competition means that fires often don’t carry in the mesic (wetter) savannas either, thus most fires are concentrated in the 400-1000 mm MAP rainfall zones.
Du Preez et al., 2011	Reviewed the effects of land management, including fire frequency, on SOC in South Africa.
Ellis and Swift, 1988	Put forward the theory that when high inter annual rainfall variability is experienced, land degradation tends to be relatively low as periods of drought cause herbivore populations to collapse, allowing vegetation to recover, whereas in areas where rainfall patterns are stable grazing pressure remains high throughout drought periods as herbivores stay in the same area leading to potential degradation.
Eswaran et al., 1997	An estimated 50% of land in Africa (40% in Sub-Saharan Africa) is considered to be suited uniquely for pastoralism due to limited and highly variable rainfall.
Everson et al., 1989	Found wet season burning caused more runoff and erosion than burning during the dry season, which would presumably lead to increased loss of SOC.

Chapter 3. Continued.	
Citation	**Key point relevant to this study**
FAO/LEAD, 2006	Some 75% of grasslands in Africa undergo burning every year.
Fynn et al., 2003	Working in South Africa on a long term (50 yr) field trial in a native grassland, showed that annual burning caused a decrease in aboveground litter inputs (0-2cm), led to increased turnover of root material below the surface (4 – 10cm), but only found significant change in organic C (a decrease) in the 0-2cm soil layer.
Gray and Bond, 2009	Looked at thicket invasion and effect on C stocks in AGB and soil in South Africa and found total carbon was highest in forests (485 Mg C ha-1), second highest in thickets (397Mg C ha- 1) and lowest in grasslands (283Mg C ha-1).
Haile, 2005	There can be great variability in rainfall in many areas especially the Sahel region, eastern Africa and southern Africa.
Hanan et al., 2008	Argued that in dry savanna systems fire tends to remove young seedlings and herbaceous ground vegetation, with little immediate impact on long-term carbon stock in biomass.
Hiernaux et al., 1999	Grazing shown to have negative effects on above and below ground carbon stocks.
Holechek et al., 2000	Reviewed studies considering short term high intensity grazing and concluded the system led to soil degradation with implications for SOC.
Hulbert, 1988	Pointed out that in drier climatic zones, some grasses that are not disturbed by grazing, trampling or fire, can deteriorate and become senescent leading to loss of land cover and subsequent degradation.
IEM, 2009	Water harvesting techniques such as bunds or micro-catchments have been shown to increase forage production and therefore have potential to increase both above and below ground C in areas with erratic rainfall.
IPCC, 2006	For the purposes of GHG accounting, CO2 is assumed to be taken up again after burning by the growth of new plants and therefore has a ‘neutral’ impact on the atmosphere if land use remains the same.
IRIN, 2007	An estimated 50% of land in Africa (40% in Sub-Saharan Africa) is considered to be suited uniquely for pastoralism due to limited and highly variable rainfall
IUCN, 2010	Different species are suited to different ecological niches and herd diversity is an important adaptation strategy of traditional pastoral systems.
Jones et al., 1990	34 year study in Kruger National Park, which showed a greater decline of SOC in the 0- 15 cm depth in plots burned every 1,2 or 3 years.

Chapter 3. Continued.	
Citation	**Key point relevant to this study**
Kay, 2001	Water harvesting in grazing lands is not widespread in Sub-Saharan Africa and its potential is therefore under researched.
Lehmann et al., 2014	Found in a global analysis a general relationship where fire reduced tree basal area, having implications for above ground C stocks with the relationship varying depending on moisture, suggesting more than one model is needed.
Macharia et al., 2011	Carried out a trial in a semi-arid area in Kenya to determine the impact of legume introduction on soil fertility in natural pastures and found a significant increase in SOC after 2 years due to the increased volume of crop residues produced by the legumes.
Materechera et al., 1998	Found soils on which vegetation had been burned annually for a 17 year period had lower SOC at 0-40cm (0.71%) than soils which were burned every 3 years (0.83%) but that levels in soils burned every 6 years were similar to those burned annually (0.77%) showing that the relationship between burn frequency and SOC is not a straightforward one and further investigation is needed.
McNaughton, 1979	Pointed out that in drier climatic zones, some grasses that are not disturbed by grazing, trampling or fire, can deteriorate and become senescent leading to loss of land cover and subsequent degradation.
McSherry and Ritchie, 2013	For the 17 studies considered they did not find a direct relationship between annual precipitation and grazing effect on SOC but did find an interaction between soil type, annual precipitation and grazing impact on SOC but did find higher grazing intensity was associated with increased SOC in grasslands dominated by C4 grasses and lower SOC in grasslands dominated by C3 grasses.
Mills and Fey, 2003	Found SOC decline in a savanna in the Eastern Cape, South Africa after intense grazing by goats.
Neely et al., 2010	Overgrazing, a function of both grazing and recovery time, results when livestock either overgraze the plants to a point of non-recovery or access the plants before they have had time to recover.
Nelson and Sommers, 1996	Total carbon (TC) includes both organic and inorganic soil carbon.
NEPAD, 2002	Approximately 4% of the cultivated area in Sub-Saharan Africa is irrigated, and this is generally confined to high production prime agricultural land.
Niamir-Fuller, 1999	Many pastoralists are subsistence herders often living in multiple family groups with sufficient labour to ‘track’ changes in pastures closely by moving and splitting herds quickly.
Ojima et al., 1994	Burning can increase annual dry matter production by encouraging early growth at the start of the new season.
Oluwole et al., 2008	Reported an increase in SOC in a 25 yr trial in burned versus unburned grassland plots in the 0-15 cm layer and attributed this to turnover of root material after burning.

Chapter 3. Continued.	
Citation	**Key point relevant to this study**
Pineiro et al., 2010	Compaction caused by trampling reduces infiltration of water which in turn can reduce plant productivity again limiting inputs to the soil.
Rass, 2006	Estimated 50 million pastoralists and agropastoralists in SSA.
Reeder and Schuman, 2002	Grazing shown to have positive effects on above and below ground carbon stocks.
Reid, 2012	Pastoralist systems, developed over thousands of years and allowing herders to use resources as and when they become available and giving grazing lands time to recover before return, can break down when movement and resources are restricted.
Ritchie, 2014	Argued that warm season C4 species, which tend to dominate in tropical grasslands, respond differently to heavy grazing than temperate C3 species and argued that warm season C4 species invest more in belowground growth in response to intermediate grazing pressure thereby increasing C inputs to the soil and therefore SOC.
Savadogo et al., 2007	Found increased grazing pressure led to lower SOC at 0-10 cm on both burned and unburned plots after 12 years but found no direct interactive effect and found the impact of grazing on species composition was different in burned and unburned plots which could potentially lead to changes in SOC in the longer term.
Savory and Butterfield, 1999	The original premise behind rotational grazing systems arose from the insight that, historically, African landscapes were grazed by large bunched and moving herds of ungulates that were responding to predators.
Scholes and Andreae, 2000	Following a fire, C can be lost from the soil due to an alteration in soil respiration and CO2 can be emitted from the soil due to decomposition of material killed but not fully burned in the fire.
Teague et al., 2010	Found that multi-paddock, adaptive management strategies for improving vegetation and animal performance were superior to continuous grazing in terms of ecosystem services, conservation and restoration.
Tefera et al., 2007	Working in Ethiopia found perennial grasses were replaced by annual grasses in heavily grazed areas, a phenomenon which didn't occur in lightly grazed areas.
Thairu and Tessema, 1987	Herbaceous legumes are not found in many grazing land in Sub-Saharan Africa, especially those in the drylands, attributable to susceptibility to drought, failure to cope with heavy grazing and high cost of seeds.
Tieszen et al., 1979	In SSA low altitude grasslands are dominated by C4 species and high altitude grasslands by C3 species.
Upton, 2012	The change in livestock production in SSA was driven by increasing livestock numbers rather than gains in livestock productivity putting pressure on SSA grazing lands and traditional grazing systems.

Chapter 3. Continued.	
Citation	**Key point relevant to this study**
Tredennick and Hannan, 2015	Trees can provide sources of fuel, forage and other products of benefit to local communities.
Upton, 2012	Between 2000 and 2010, production of livestock increased by 3% in Sub-Saharan Africa ; driven by increasing livestock numbers rather than gains in livestock productivity.
Vågen et al., 2005	Burning is a common management practice in many grazing lands in Sub-Saharan Africa dating back to prehistoric times.
von Wehrden et al., 2012	Reviewed 58 studies where rainfall variability and land degradation had been studied, finding that degradation caused by grazing was found almost exclusively in areas with relatively stable annual precipitation.
Weber and Gokhale, 2011	Called for transformation of the debate over specific grazing systems into a greater understanding of adaptive management processes and attention to better management.
Western, 1982	Many pastoralists are subsistence herders often living in multiple family groups with sufficient labour to 'track' changes in pastures closely by moving and splitting herds quickly.
Williams et al., 2007	Had similar estimated for C losses from fires in savannas as found by the CarboAfrica Project.
Winrock International, 1992	In Sub-Saharan Africa cattle, goats, sheep and camels are the most common types of livestock.
World Bank, 2008	Livestock are an irreplaceable source of livelihood for the poor and are one of the fastest growing agricultural sectors.
Woomer et al., 2004	In Senegal, estimated 0.77 t C ha_1 yr_1 could be sequestered over 20 years by restoring degraded grasslands to woody grasslands and proposed that each percent increase in woody canopy cover results in 800 kg C ha_1 in woody biomass.
Yayneshet and Treydte, 2015	Conducted a meta-analysis that included consideration of 34 studies from SSA which considered the effects of livestock grazing on SOC and found no differences between rangelands, commercial ranches and game reserves.

Chapter 4. Measuring rangeland health and soil carbon in Africa	
Citation	**Key point relevant to this study**
AfSIS, 2014.	The Land health surveillance approach is being applied at Sub-Saharan Africa scale in the Africa Soil Information Service.
AfSIS, 2015a	Deploying centralised calibration development and an online spectral prediction application to reduce the need for each laboratory to develop its own calibrations.
Aynekulu et al., 2014	The land health surveillance approach has been applied to assess impacts of land management interventions such as the effects of livestock exclusion and of fire on rangeland carbon stocks in Ethiopia and Burkina Faso.
Bellon-Maurel and McBratney, 2011	A review showing an exponential increase in the use of near infrared (NIR) and mid infrared (MIR) reflectance spectroscopy for soil analysis

Bird et al., 2010	Found in their review of existing methods of carbon accounting that the IPCC Good Practice Guidance for Land Use, Land-Use Change and Forestry (GPG-LULUCF) and the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2006 IPCC Guidelines) and The Voluntary Carbon Standard (VCS) are the widely referred carbon accounting methods.
Cobo et al., 2010	IR data can be integrated with geostatistic data.
Conant et al., 2013	A number of carbon measurement schemes are emerging for specific applications. However, lack of robust, rapid and cost-effective methods is a major problem.
Croft et al., 2012	IR data can be integrated with remote sensing and topographic information.
de Gruijter et al., 2006	Monitoring of natural resources has become increasingly important over the last five decades and this trend is likely to continue.
Donovan, 2013	The dry combustion or elemental analysis procedure is the most accurate common test for soil carbon, and is often cheaper than other tests.
Ellis and Larsen, 2008	A number of carbon measurement schemes are emerging for specific applications. However, lack of robust, rapid and cost-effective methods is a major problem.
EthioSIS, 2014	The land health surveillance is being applied at national scale by the Ethiopia Soil Information System
Harris et al., 2001	To remove the influence of inorganic carbon (carbonate) soil carbon is determined on acidified samples, i.e. fumigated with hydrochloric acid to remove inorganic carbon.
Henry et al., 2009	African soils are important carbon reservoirs in which the top 100 cm constitutes about 68% of the terrestrial carbon pool of Africa.
Herrick et al., 2013	Monitoring of natural resources has become increasingly important mainly due to human pressure on natural resources and increasing awareness of its negative impact on ecosystem health and human wellbeing.
Huang et al., 2007	IR data can be integrated with remote sensing and topographic information for digital soil mapping at landscape level.
IPCC, 2003	The development of locally calibrated models that can use easily collected data, minimizing the cost of demonstrating a change in soil organic carbon stock is a research priority.
Lee et al., 2009	Estimates of soil carbon stocks to a fixed depth using single depth bulk density are often biased therefore it is necessary to consider corrections for spatial and temporal variation in bulk density in quantifying SOC stocks along a soil profile.

Chapter 4. Continued.	
Citation	**Key point relevant to this study**
McBratney et al., 2003	SOC monitoring at fine temporal resolution can be also achieved using a digital soil mapping approach where SOC is predicted using covariates such as terrain attributes e.g. topographic wetness index, spectral reflectance bands from satellite imagery, land cover and natural vegetation, and parent material
McCarty et al., 2002	To remove the influence of inorganic carbon (carbonate), soil carbon is determined on acidified samples, i.e. fumigated with hydrochloric acid to remove inorganic carbon.
Moreira et al., 2009	Have shown the potential application of IR to measure bulk density.
Murty et al., 2002	Found that the impact of conversion of forests into cultivated lands on the changes in soil carbon stock was often inflated due to the confounding influence of changes in bulk density.
Palmer et al., 2002	It is possible that the cheapest way to measure changes in soil carbon with any of these methods is to collect, store, and then analyze a time series of samples from the same soil (to determine any limitations resulting from soil sample preparation and processing), in the same lab, at the same time.
Nocita et al., 2015	MIR is more reproducible and robust in measuring soil carbon than NIR.
Reeves III, 2009	Has shown the potential application of IR to measure soil carbon onsite.
Rossel et al., 2006	MIR is more reproducible and robust in measuring soil carbon than NIR.
Rossel et al., 2014	Used IR data to develop soil carbon map of Australia.
Shepherd and Walsh, 2007	Soil sampling and infrared analysis can be integrated into study designs to accumulate evidence on the impact of grazing land management interventions on soil health and soil carbon stocks.
Shepherd et al., 2015	The key objectives are to: (i) identify land health problems, (ii) establish quantitative objectives for land health promotion, (iii) provide information for the design and planning of land management intervention programmes and resource allocation priorities, (iv) determine the impact of specific interventions, and (v) identify research, service and training needs for different stakeholder groups.
Skjemstad and Baldock, 2007	Thermal oxidation - one of two widely used methods to measure soil carbon content.
UNEP, 2012	The key objectives are to: (i) identify land health problems, (ii) establish quantitative objectives for land health promotion, (ii) provide information for the design and planning of land management intervention programmes and resource allocation priorities, (iv) determine the impact of specific interventions, and (v) identify research, service and training needs for different stakeholder groups.
UNFCCC, 2006	Although the change in SOC stock varies with factors that influence the rate of production and decomposition of carbon, a five-year monitoring cycle is recommended by IPCC, whereas UNFCCC recommend a monitoring interval of between 10 and 20 years.
Vågen et al., 2013.	The land health surveillance applied at regional scale.
Vågen et al., 2012	Infrared spectroscopy data was used in mapping soil carbon in Kenyan rangelands.
Vågen et al., 2006	At a continental level, climate tends to explain more variation in soil organic

	carbon than any other single factor but locally-historic land use often has a dominant influence, and this may not be well reflected by current land use.
Walkley and Black, 1934	Walkley-Black procedure is one of two widely used methods to measure soil carbon content.
Waswa et al., 2013	The land health surveillance applied at landscape scale.
Wynn et al., 2006	At a continental level, climate tends to explain more variation in soil organic carbon than any other single factor.

Chapter 5. Map-based estimates of present carbon stocks of grazing lands in Sub-Sahara Africa	
Citation	**Key point relevant to this study**
AfSIS, 2015a	Landscape scale effects may be considered once high resolution soil property maps, derived from digital soil mapping become available for Sub-Saharan Africa.
Banwart et al., 2014	Effects of land use and management interventions on SOC dynamics largely occur at the local scale.
Banwart et al., 2015	Changes in land use and management practices may result in significant losses of SOC, with concomitant emissions of CO2 to the atmosphere, as well as a deterioration of soil health and human livelihood associated with a loss of soil organic matter content unless soils are judiciously managed.
Bationo et al. 2007	Changes in land use and management practices may result in significant losses of SOC, with concomitant emissions of CO2 to the atmosphere, as well as a deterioration of soil health and human livelihood associated with a loss of soil organic matter content unless soils are judiciously managed.
Batjes, 2004	Earlier estimates of SOC reserves for Africa were based on coarser resolution soil-geographic databases and a more limited set of soil profiles.
Bernoux et al., 2011	Possible changes in SOC content and net CO2eq emissions can best be assessed using appropriately-scaled and regionally validated modeling tools supported by long-term monitoring systems.
Conant et al., 2010	Possible changes in SOC content and net CO2eq emissions can best be assessed using appropriately-scaled and regionally validated modeling tools supported by long-term monitoring systems.
de Brogniez et al., 2011	Possible changes in SOC content and net CO2eq emissions can best be assessed using appropriately-scaled and regionally validated modeling tools supported by long-term monitoring systems.
Du Preez et al., 2011	Effects of socio-economically and biophysically driven changes in land use and management on SOC content, including the occurrence of natural and man-made fires, within the various livestock production systems, cannot be distilled from the present continental-scale data sets.
Govender et al., 2006	Effects of socio-economically and biophysically driven changes in land use and management on SOC content, including the occurrence of natural and man-made fires, within the various livestock production systems, cannot be distilled from the present continental-scale data sets.
Gray et al., 2011	SOC as a reflection of regional differences in the type and intensity of the soil forming factors of climate, parent material, relief, organisms (notably humans) and time.
Hengl et al., 2014	Landscape scale effects may be considered once high resolution soil property maps, derived from digital soil mapping become available for Sub-Saharan Africa.
Henry et al., 2009	Indicated that use of different soil profile databases, spatial datasets and/or mapping approaches can easily lead to differences of up to 30% in the estimates for SOC stocks for Africa.
Izac, 1997	Effects of land use and management interventions on SOC dynamics largely occur at the local scale.

Chapter 5. Continued.	
Citation	**Key point relevant to this study**
Jenny, 1941	SOC as a reflection of regional differences in the type and intensity of the soil forming factors of climate, parent material, relief, organisms (notably humans) and time.
Milne et al., 2015	Changes in land use and management practices may result in significant losses of SOC, with concomitant emissions of CO2 to the atmosphere, as well as a deterioration of soil health and human livelihood associated with a loss of soil organic matter content unless soils are judiciously managed.
Milne et al., 2012	Possible changes in SOC content and net CO2eq emissions can best be assessed using appropriately-scaled and regionally validated modeling tools supported by long-term monitoring systems.
Robinson et al., 2011	Estimates for mean SOC content, as observed in different livestock production systems.
Vågen and Leigh, 2013	Landscape scale effects may be considered once high resolution soil property maps, derived from digital soil mapping become available for Sub-Saharan Africa.
van Wesemael et al., 2011	Possible changes in SOC content and net CO2eq emissions can best be assessed using appropriately-scaled and regionally validated modeling tools supported by long-term monitoring systems.

Chapter 6. Modeling Soil Carbon in African grazing lands	
Citation	**Key point relevant to this study**
Ardö and Olsson, 2003	Linked CENTURY to a spatial database to make estimates of carbon sequestration in a region in semi-arid Sudan; included grasslands and savannas for which they simulated grazing intensity and fire regimes.
Barnwell et al., 1992	Model classification may include aspects of use of the model, such as how well the model has been field tested and its performance, how well it is documented, the agreement between the data needed by the model and available datasets, its flexibility for use in addressing questions, and its relative ease of use.
Bombelli et al., 2009	A lack of data on the carbon budget for African landscapes makes dynamic modeling results more uncertain.
Boone et al., 2002	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Boone, 2005	Used SAVANNA to assess the means in which vegetation changed under increasing levels of landscape fragmentation in South Africa, Tanzania, and Kenya.
Boone, 2007	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Boone et al., 2011a	Some models are relevant to broader scales, e.g. G-Range.
Boone et al., 2011b	Linked SAVANNA model to agent-based household models.
Boone et al., 2013	Some models are relevant to broader scales, e.g. G-Range.
Ciais et al., 2009	Africa accounts for 20% of the total land surface of the earth, making its carbon dynamics particularly relevant globally, however, these dynamics are poorly understood.
Ciais et al., 2011	A lack of data on the carbon budget for African landscapes makes dynamic modeling results more uncertain; proposed an African integrated carbon observing system, where an array of eddy covariance flux towers in grazing lands, ongoing satellite mapping of CO, CO2, CH4, fire, and land use change, and modeling fuse to provide information to guide planning.
Coleman et al., 1997	The Rothamsted Carbon Model (i.e., RothC) is a simple and popular choice for analyses of carbon dynamics, explicitly for non-flooded soils only.
Colomb et al., 2013	An example of this for GHG accounting models in general has been developed by FAO and partners and consists of an Excel tool which guides the user through a number of decision trees to arrive at the most appropriate tool.
Conant et al., 2001	Existing soil carbon models can yield surfaces that align relatively closely with observed values, and comparisons among modeling results are promising but with sometimes high uncertainties – example: CENTURY model.
Conant and Paustian, 2002	Existing soil carbon models can yield surfaces that align relatively closely with observed values, and comparisons among modeling results are promising but with sometimes high uncertainties – example: CENTURY model.
Coughenour, 1992	Some models are relevant to landscapes, e.g. SAVANNA, an ecosystem model created working in the Turkana District, Kenya.
Diels et al., 2001	Using RothC model in weighing the benefits of fertilization of West African savanna crops.
Denish et al., 2015	Used G-Range model (a streamlined representation of the CENTURY model)

	in planning documents.
Easter et al., 2007	Working at the national scale, Kamoni et al. (2007) used the GEFSOC System which links the CENTURY and the RothC models to a GIS to estimate C stock changes for Kenya.
Ellis and Coughenour, 1998	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Eve et al., 2014	Standardization of methods is steadily improving and made more straightforward through web-based tools.
Fisher et al., 2013	In a nine model approach, noted two effects for increasing uncertainty between models, droughts and increased observed CO2 – reduced precipitation and hotter temperatures decrease CO2 uptake, and may change the humid tropics from a net sink to a net source of atmospheric CO2.
Galvin et al., 2004	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Gilhespy et al., 2014	Provides a type of model 'family tree' for the DNDC (DeNitrification DeComposition) model which researchers have modified to apply to many questions, conditions, and regions.
Grote et al., 2009	Used DNDC model joined that to a land-surface model to yield a tool that simulated climate chemistry and biogeochemical processes in a coupled way, applied to a reserve in Burkina Faso.
Higgins and Scheiter, 2012	Landscapes in Africa are changing in ways that could change the regions' capacity to store carbon, for example reforestation.
Huntzinger et al., 2013	Explicit comparisons between models are done, and help to identify both the range of variation expected in results and the structural differences in models that may lead to differences in outcomes.
Kamoni et al., 2007	Working at the national scale, used the GEFSOC System which links the CENTURY and the RothC models to a GIS to estimate C stock changes for Kenya.
Kaonga and Coleman, 2008	Using RothC model, carbon turnover was judged for coppicing fallows in eastern Zambia.
Lardy et al., 2011	RothC, SOMA, and SOCRATES models use five carbon pools, whereas CENTURY uses three pools plus aboveground and belowground plant residual pools.
Li et al., 1992	The DNDC (DeNitrification DeComposition) model has been used all over the world to simulate N20, CO2 and N2 emissions from agricultural soils.
Liu et al., 2011	Responses to disturbances such as droughts and fires are not simulated well with current methods.
Manzoni and Porporato, 2009	Reviewed ~250 biogeochemical models, and classified their theoretical and mathematical frameworks.
McGill, 1996	A practical means to speed analysts' identification of appropriate models would be helpful; classification may include attributes of the models, such as time step, relative complexity, spatial and temporal scale, cost, and data requirements.
Melillo et al., 1995	Quantitative comparisons of ecosystem models are difficult because any disagreements in output may be due to different processes incorporated in the models or different spatial or temporal scales.

Milne et al., 2007	Working at the national scale, Kamoni et al. (2007) used the GEFSOC System which links the CENTURY and the RothC models to a GIS to estimate C stock changes for Kenya.
Milne and Brown, 1997	Landscapes in Africa are changing in ways that could change the regions' capacity to store carbon, for example the loss of wetlands.
Nie et al., 1992	Increases in atmospheric CO2 could provide advantages to C3 woody plants, causing large shifts in forest cover, which in turn could change soil carbon balances, although increasing temperatures and reduced precipitation make outcomes uncertain.
Parton, 1996	Suggests that sufficient modularity in simulation tools would allow procedures from different models to be swapped, and the implications for results explored.
Parton et al., 1987	The CENTURY model is a dynamic ecosystem model that simulates the turnover of soil organic matter and plant nutrients, originally developed to simulate conditions in the North American Great Plains grasslands but has been adapted and applied to many ecosystems worldwide.
Paustian et al., 1997	Some of factors are not well represented in soil carbon models that may be used in Africa due to a lack of observational data from the region and because the majority of models have been developed and parametrized for temperate conditions.
Ritchie, 2014a	Some models are relevant to landscapes, e.g. SNAP model which explicitly incorporates plant compensation to grazing at an annual time-step; model does not include nitrogen dynamics, but does include pathways for decomposition by invertebrate detritivores such as termites.
Ritchie, 2014b	Adapted the SNAP model to the Northern Rangelands Trust conservancies of northern Kenya.
Rosenzweig et al., 2014	Map data from the Agricultural Model Intercomparison and Improvement Project.
Sankaran et al., 2008	C4 grasses, and the fires they support, have played a major role in preventing trees from establishing and have led to the creation of vast areas of savanna.
Sitch et al., 2013	Ensemble model results (e.g., the TRENDY collection of eight dynamical global vegetation models, where mean responses from multiple models incorporate more relationships than any one model.
Smith et al., 2014	Using RothC model, compared the carbon balance of biogas digestion, composting, and biochar in Sub-Saharan Africa.
Smith et al., 2010	The ECOSSE model (based on Roth-C) has been developed to deal with flooded soils.
Smith et al., 1997	Existing soil carbon models can yield surfaces that align relatively closely with observed values, and comparisons among modeling results are promising.
Still et al., 2003	Increases in atmospheric CO2 could provide advantages to C3 woody plants, causing large shifts in forest cover, which in turn could change soil carbon balances, although increasing temperatures and reduced precipitation make outcomes uncertain.
Stockman et al., 2013	A difficult question regarding model portability is the degree to which models must be assessed for each new application - using an application in discovery and decision making without some calibration using local observations would be risky.

Thornton et al., 2003	Linked SAVANNA model to population-based household models.
Thornton et al., 2004	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Thornton et al., 2007	There have been many subsequent improvements to the SAVANNA model and it has been applied to systems world-wide.
Traore et al., 2007	New satellite sensors are becoming available, such as microwave sensors that allow for estimation of tree biomass, which will reduce modeling uncertainties
VEMAP members, 1995	A practical means to speed analysts' identification of appropriate models would be helpful; classification may include attributes of the models, such as time step, relative complexity, spatial and temporal scale, cost, and data requirements.
Weber et al., 2009	Used remotely sensed data to assess carbon balance results from the ORCHIDEE, LPJ-DGVM, LPJ-Guess, and JULES models, with simulations done using standardized protocols and forcing data (e.g., total precipitation, temperature, humidity, wind); found the modeled results differ in magnitude, but that the overarching patterns were similar.
West et al., 2012	Africa may be poised to experience more vegetation changes than other regions of the world due to the commonality of C4 plants.
Williams et al., 2007	From the African Carbon Exchange project came the ACE model, focused on carbon dynamics and the ways in which disturbances may alter land cover.
Woomer et al., 2004	Parameterized and applied CENTURY to grasslands with varying amounts of woody cover in Senegal, using the analysis to estimate that restoring degraded grasslands to woody grasslands in Senegal over a 20 year period could sequester 0.77 t C ha-1 yr-1.

Chapter 7. Model based estimates of potential carbon stocks in grass/rangelands in Sub-Saharan Africa	
Citation	**Key point relevant to this study**
Conant, 2011	All assessments of GHG mitigation potential in grazing lands were based on the concept that a change in management practices can lead to a change in C stocks and/or N2O emissions.
Conant and Paustian, 2004	To-date, limited data on grazing land management have constrained the ability of researchers to delineate areas amenable to improved management from those that are not amenable.
Conant et al., 2001.	Another limitation of most broad-scale assessments is that they have relied on emission factors generated from the synthesis or meta-analysis of published studies.
Henderson et al., 2015	Sought to estimate the effectiveness of mitigation practices in grazing lands using process-based models and detailed spatial information.
Lal, 2004	Estimates for C stocks though improved management in grazing lands.
Ogle et al., 2004.	Another limitation of most broad-scale assessments is that they have relied on emission factors generated from the synthesis or meta-analysis of published studies.
Parton et al., 1998	Process-based model – the Daycent model.
Parton et al., 1987	Process-based model – the Century model.
Paustian et al., 1997	All estimates of mitigation potential were constructed using (1) information about changes in soil C storage and N2O emission rates given a change in land management and (2) information about where land management changes were feasible.
Smith, 2008	Another limitation of most broad-scale assessments is that they have relied on emission factors generated from the synthesis or meta-analysis of published studies.
Smith et al., 2007	Estimates for C stocks though improved management in grazing lands.

Chapter 8. Synthesis	
Citation	**Key point relevant to this study**
Ardo and Olsen, 2003	Examples of application to situations in Africa including grazing lands are limited but increasing in number.
Aynekulu et al., 2011	Although infrared spectroscopy significantly reduces the analytical cost of measuring soil carbon, costs incurred in soil sampling and preparation still form the largest component cost.
Conant and Paustian, 2002	Examples of application to situations in Africa including grazing lands are limited but increasing in number.
Hengl, 2015	Such landscape scale effects may be more easily considered now as high (250 m) resolution soil property maps, derived from digital soil mapping, have very recently become available for SSA.
IPCC, 2003	In some cases the cost of demonstrating the change in carbon stocks in soils to the required accuracy and precision may exceed the benefits that accrue from the increase in stocks.
Kamoni et al., 2010	Examples of application to situations in Africa including grazing lands are limited but increasing in number.
UNCCD, 2012	The United Nations Convention to Combat Desertification (UNCCD) aims for zero net land degradation by 2030.

Appendix 2. Misconceptions about pastoralism

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Pastoral grazing strategies in sub-Saharan Africa form sophisticated management systems to sustain human life in harsh environments. These pastoral institutions need be understood and built upon when contemplating actions to improve carbon management. For example, there is a wide literature about traditional rules concerning grazing movement to sustain grazing productivity and thus the resilience of pastoral livelihoods (Ellis and Swift, 1988a; McCabe, 1990, 1994; Behnke ,1999; McCabe et al., 1999; Niamir-Fuller, 1999c, a; Turner, 1999; Fernandez-Gimenez, 2002; Homewood et al., 2004; McCabe, 2004; Vetter, 2005; Fernandez-Gimenez and Le Febre, 2006; Reid et al., 2008; Butt et al., 2009; Nkedianye et al., 2011; Turner, 2011; Goldman and Riosmena, 2013 Tsegaye et al., 2013). These institutions (rules and norms) determine when and where people graze, for how long, with what types of livestock, who uses which water source, and how to judge grassland health. These pastoral institutions and norms have developed over the last 7000 years or so, when pastoralists became the first people to produce food in much of sub-Saharan Africa (Marshall, 1998; Marshall and Hildebrand, 2002). And they are still negotiated today, especially in the face of major development changes in pastoral systems, especially those pastoral areas near towns and market centers, but also in more remote rangelands where changes in climate are starting to occur.

Generally, dryland development efforts and policy are built on misconceptions about pastoral management strategies and their impacts, even today. Unfortunately, this means that projects and policy often promote changes to pastoral management that are neither helpful nor adaptive, based on these basic misconceptions (Niamir-Fuller, 1999c;(Reid et al., 2014). Part of the reason for these misconceptions is that most of the world's people live far from rangelands (Safriel et al., 2005) in urban, forested or farm-based systems, where climate is predictable and sedentary livestyles are viable. It is difficult, and sometimes impossible, for people with this life experience to understand how to sustain life in rangelands where vegetation and water resources are often ephemeral in time and patchy in space, and pastoral movement to these ephemeral patches is a critical strategy to improve milk and meat production (Boone and Hobbs, 2004).

Other misconceptions about pastoral management of rangelands, especially in Africa, include the following (Reid et al., 2014), p. 220-221:

'In the past, for example, we assumed most rangelands used in common by herders to be overgrazed and degraded. Now we understand that many drier rangelands have non-equilibrium dynamics, where climate has more impact on vegetation than grazing does (but these rangelands can still be overgrazed) (Ellis and Swift 1988b, Vetter 2005, von Wehrden et al. 2012). In wetter rangelands, exhibiting equilibrium dynamics, overgrazing of vegetation is more widespread. We now also understand that rangelands often exhibit non-linear dynamics, where vegetation can shift from one state to another over thresholds or past tipping points, rather than smoothly following a predictable vegetation succession (Briske et al. 2005). Globally, many rangelands are shifting from grass to shrub or tree dominated ecosystems (Asner et al. 2004), which we attributed to livestock grazing in the past. Today there is no universally accepted explanation of those changes (D'Odorico et al. 2013). In the past, pastoralism was thought to be the root cause of the spread of deserts, especially in Africa (Sinclair and Fryxell 1985). But recent evidence suggests that pastoralists can promote greening by planting or conserving trees (Sendzimir et al. 2011, Roba and Oba 2013), and that global rangelands are greening the world over (Cook and Pau 2013). This leads some to suggest that degradation is not a widespread phenomenon, at least in Africa's Sahel (Fensholt et al. 2013).'

‘Conventional wisdom about how pastoralists manage rangelands has changed just as dramatically. In 1776, Adam Smith’s Wealth of Nations popularized a notion that the natural advancement of livelihoods and land use is from hunter to pastoralist to farmer. We now know that pastoralism evolved from crop agriculture in many parts of the world, rather than vice versa. And it is clear that pastoralism is one of the most efficient ways to turn sunlight into food in marginal lands. Misunderstanding of pastoral risk management led to the idea that pastoralists are irrational to hold large herds (Herskovits 1926), whereas today, this is understood to be a sound way to manage risk in the face of recurrent dry seasons, drought and winter storms. The ‘tragedy of the commons’ promoted the idea that joint grazing of rangelands by multiple herders inevitably leads to over use, because no herder has the incentive to restrain from overgrazing the commons (Hardin 1968). In our view, this tragedy is misnamed and should be called the ‘tragedy of open access’, since most pastoral communities have agreed upon rules of use for their commons (McCabe 1990). Further work suggested what might be called a ‘tragedy of enclosure’, when common lands become privatized and fragmented by boundaries, like fencing (Galvin et al. 2008). Settlement of nomadic pastoralists and intensification of livestock production was promoted in the past to exert political control over mobile populations, collect taxes, provide health and education services, and increase productivity. As climate becomes more variable and pastoral movement more important, it is not clear that privatization, settlement and agricultural intensification are the right approaches to these problems. Comparing livestock weights on commercial ranches and pastoral common lands gave the impression that commercial ranches are more productive. But the opposite is true because high labor inputs and opportunistic grazing over large landscapes gives common land pastoralists the edge in production per unit area over commercial enterprises (Sandford 1983).’

Pastoral subsistence strategies and movement systems in Africa

Most pastoralists in Africa are largely *subsistence* herders whose goal is to produce edible animal products (milk, meat, ghee) as many days of the year as possible and to produce extra animals for sale. To accomplish this, they keep a milking herd of females and a herd with multiple species (some combination of camels, cattle, donkeys, sheep and goats) to avoid the risk of herding just one species. Subsistence herders often live in multiple family groups to have sufficient labor to ‘track’ changes in pastures closely by moving and splitting herds quickly (Western, 1982; Niamir-Fuller, 1999b), and thus are able to support more animals on the same pasture than a *commercial* rancher can (Behnke, 1983, 1985).

There are five types of major movement management strategies that herders use in Africa that differ in: 1) how far the family moves, 2) how long they live in a place, 3) how they own the land, and 4) how much they depend on growing livestock or crops to support their families (Reid, 2012). *Nomadic herders* (‘pure’ pastoralists) move their families and herds frequently to greener pastures, live almost entirely off their livestock, maintain only loose connections to markets and the state, often own land in common, and often have no specific dry- and wet-season pastures. *Seasonally moving herders* (or transhumant pastoralists) move livestock and part of the family between wet and dry season pastures that are reasonably well defined, sometimes cultivate, and also own land in common (Bromley, 1991; Behnke, 1999; Zeidane, 1999; Little, 2003; McPeak and Little, 2005). *Settled families with nomadic herds* have part of the family live permanently near a market center to access markets, schools and health care but send livestock out with a herder to camps far from town (McPeak and Little, 2005). If it is wet enough, the settled family might grow some crops.. *Sedentary herder-farmers* have permanent settlements, herd animals in pastures around their settlements, grow crops and usually own their land (Turton, 1991; Coppock, 1994; Fratkin et al., 1994; Lane, 1996; Niamir-Fuller, 1999a). Most rare in Africa are *fenced* *ranchers* who manage large tracts of fenced land and are well connected to meat and milk markets (Blench, 2001).

Literature cited

Asner, G. P., A. J. Elmore, L. P. Olander, R. E. Martin, and A. T. Harris. 2004. Grazing systems, ecosystem responses, and global change. Annual Review Of Environment And Resources **29**:261-299.

Behnke, R. H. 1983. Production rationales: the commercialization of subsistence pastoralism. Nomadic Peoples **14**:3-27.

Behnke, R. H. 1985. Measuring the benefits of subsistence versus commercial livestock production in Africa. Agricultural Systems **16**:109-135.

Behnke, R. H. 1999. Stock-movement and range-management in a Himba community in north-western Namibia. Pages 184-216 *in* M. Niamir-Fuller, editor. Managing Mobility in African Rangelands. FAO and Beijer International Institute of Ecological Economics, London.

Blench, R. M. 2001. 'You can't go home again': Pastoralism in the new millenium. FAO: Animal Health and Production Series, No. 50, Rome.

Boone, R. B., and N. T. Hobbs. 2004. Lines around fragments: Effects of fencing on large herbivores. African Journal of Range & Forage Science **21**:79-90.

Briske, D. D., S. D. Fuhlendor, and E. E. Smeins. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology & Management **58**:1-10.

Bromley, D. W. 1991. Environment and Economy: Property Rights and Public Policy. Blackwell Publishers, Oxford.

Butt, B., A. Shortridge, and A. WinklerPrins. 2009. Pastoral herd management, drought coping strategies, and cattle mobility in southern Kenya. Annals of the Association of American Geographers **99**:309-334.

Cook, B. I., and S. Pau. 2013. A global assessment of long-term greening and browning trends in pasture lands using the GIMMS LAI3g dataset. Remote Sensing **5**:2492-2512.

Coppock, D. L. 1994. The Borana Plateau of southern Ethiopia: Synthesis of pastoral research, development and change, 1980-1991. International Livestock Centre for Africa, Addis Ababa, Ethiopia.

D'Odorico, P., A. Bhattachan, K. F. Davis, S. Ravi, and C. W. Runyan. 2013. Global desertification: Drivers and feedbacks. Advances in Water Resources **51**:326-344.

Ellis, J. E., and D. M. Swift. 1988a. Stability Of African Pastoral Ecosystems - Alternate Paradigms And Implications For Development. Journal of Range Management **41**:450-459.

Ellis, J. E., and D. M. Swift. 1988b. Stability of African pastoral ecosystems: Alternative paradigms and implications for development. Journal of Range Management **41**:450-459.

Fensholt, R., K. Rasmussen, P. Kaspersen, S. Huber, S. Horion, and E. Swinnen. 2013. Assessing Land Degradation/Recovery in the African Sahel from Long-Term Earth Observation Based Primary Productivity and Precipitation Relationships. Remote Sensing **5**:664-686.

Fernandez-Gimenez, M. E. 2002. Spatial and social boundaries and the paradox of pastoral land tenure: A case study from postsocialist Mongolia. Human Ecology **30**:49-78.

Fernandez-Gimenez, M. E., and S. Le Febre. 2006. Mobility in pastoral systems: Dynamic flux or downward trend? International Journal of Sustainable Development and World Ecology **13**:341-362.

Fratkin, E., K. A. Galvin, and E. A. Roth. 1994. African Pastoralist Systems: An Integrated Approach. Lynne Rienner Publishers, Boulder.

Galvin, K. A., R. S. Reid, R. H. Behnke, and N. T. Hobbs, editors. 2008. Fragmentation in Semi-arid and Arid Landscapes: Consequences for Human and Natural Systems. Springer, Dordrecht.

Goldman, M. J., and F. Riosmena. 2013. Adaptive capacity in Tanzanian Maasailand: Changing strategies to cope with drought in fragmented landscapes. Global Environmental Change-Human and Policy Dimensions **23**:588-597.

Hardin, G. 1968. Tragedy of the commons. Science **162**:1243-&.

Herskovits, M. 1926. The cattle complex in East Africa. American Anthropologist **28**:230-272, 361-388, 494-528, 633-664.

Homewood, K., E. Coast, and M. Thompson. 2004. In-migrants and exclusion in east African rangelands: Access, tenure and conflict. Africa **74**:567-610.

Lane, C. 1996. Pasture Lost: Barabaig Economy, Resource Tenure, and the Alienation of their Land in Tanzania. Initiatives Publishers, Nairobi, Kenya.

Little, P. D. 2003. Somalia: Economy without a State. International African Institute, James Currey, Indiana University Press, Btec Books, Oxford.

Marshall, F. 1998. Early food production in Africa. The Review of Archaeology:47-58.

Marshall, F., and E. Hildebrand. 2002. Cattle before crops: The beginnings of food production in Africa. Journal of World Prehistory **16**:99-143.

McCabe, J. T. 1990. Turkana pastoralism: A case against the tragedy of the commons. Human Ecology **18**:81-103.

McCabe, J. T. 1994. Mobility and land use among African pastoralists: old conceptual problems and new interpretations. Pages 69-90 *in* E. Fratkin, K. A. Galvin, and E. A. Roth, editors. African Pastoralist Systems: An Integrated Approach. Lynne Rienner Publishers, Boulder.

McCabe, J. T. 2004. Cattle Bring Us to Our Enemies. University of Michigan Press, Ann Arbor.

McCabe, J. T., R. Dyson-Hudson, and J. Wienpahl. 1999. Nomadic movements. Pages 108-121 *in* M. A. Little and P. W. Leslie, editors. Turkana Herders of the Dry Savanna. Oxford University Press, Oxford, England.

McPeak, J., and P. D. Little. 2005. Cursed if you do, cursed if you don't: The contradictory processes of sedentarization in northern Kenya. Pages 87-104 *in* E. Fratkin and E. A. Roth, editors. As Pastoralists Settle. Kluwer Academic Publishers, Dordrecht, Netherlands.

Niamir-Fuller, M. 1999a. Conflict management and mobility among pastoralists in Karamoja, Uganda. Pages 149-183 *in* M. Niamir-Fuller, editor. Managing Mobility in African Rangelands. FAO and Beijer Institute of Ecological Economics, London.

Niamir-Fuller, M. 1999b. Introduction. Pages 1-9 *in* M. Niamir-Fuller, editor. Managing Mobility in African Rangelands. FAO and Beijer International Institute of Ecological Economics, London.

Niamir-Fuller, M. 1999c. Managing mobility in African rangelands. Pages 102-131 *in* N. McCarthy, B. Swallow, M. Kirk, and P. Hazell, editors. Property Rights, Risk and Livestock Development in Africa. International Food Policy Research Institute, International Livestock Research Institute, Washington, D.C., Nairobi.

Nkedianye, D., J. de Leeuw, J. O. Ogutu, M. Y. Said, T. L. Saidimu, S. C. Kifugo, D. S. Kaelo, and R. S. Reid. 2011. Mobility and Livestock mortality in communally used pastoral areas: the impact of the 2005-2006 drought on livestock mortality in Maasailand. Pastoralism, Research, Policy, and Practice **1**:1-17.

Reid, R., H. Gichohi, M. Said, D. Nkedianye, J. Ogutu, M. Kshatriya, P. Kristjanson, S. Kifugo, J. Agatsiva, S. Adanje, and R. Bagine. 2008. Fragmentation of a peri-urban savanna, Athi-Kaputiei Plains, Kenya. Pages 195-224 *in* K. A. Galvin, R. S. Reid, R. H. Behnke, and N. T. Hobbs, editors. Fragmentation in Semi-Arid and Arid Landscapes: Consequences for Human and Natural Systems. Springer, Dordrecht, The Netherlands.

Reid, R. S. 2012. Savannas of Our Birth: People, Wildlife and Change in East Africa. University of California Press, Berkeley.

Reid, R. S., M. E. Fernández-Giménez, and K. A. Galvin. 2014. Dynamics and resilience of rangelands and pastoral peoples around the globe. Annual Review Of Environment And Resources **39**:217-242.

Roba, H. G., and G. Oba. 2013. Understanding the role of local management in vegetation recovery around pastoral settlements in northern Kenya. Environmental Management **51**:838-849.

Safriel, U., Z. Adeel, D. Niemeijer, J. Puigdefabregas, R. White, R. Lal, M. Winslow, J. Ziedler, S. Prince, E. Archer, C. e. a. King, B. Shapiro, K. Wessels, T. Nielsen, B. Portnov, I. Reshef, J. Thonell, E. Lachman, D. McNab, M. El-Kassas, and E. Ezcurra. 2005. Dryland Systems. Pages 623-662 Millennium Ecosystem Assessment: Condition and Trends, Volume 1. Island Press, Washington, DC, USA.

Sandford, S. 1983. Management of Pastoral Development in the Third World. John Wiley & Sons, Chichester, UK.

Sendzimir, J., C. P. Reij, and P. Magnuszewski. 2011. Rebuilding resilience in the Sahel: regreening in the Maradi and Zinder regions of Niger. Ecology And Society **16**:http://dx.doi.org/10.5751/ES-04198-160301.

Sinclair, A. R. E., and J. M. Fryxell. 1985. The Sahel of Africa - Ecology of a disaster. Canadian Journal Of Zoology-Revue Canadienne De Zoologie **63**:987-994.

Tsegaye, D., P. Vedeld, and S. R. Moe. 2013. Pastoralists and livelihoods: A case study from northern Afar, Ethiopia. Journal of Arid Environments **91**:138-146.

Turner, M. D. 1999. The role of social networks, indefinite boundaries and political bargaining in maintaining the ecological and economic resilience of the transhumance systems of Sudano-Sahelian West Africa. Pages 97–123 *in* M. Niamir-Fuller, editor. Managing mobility in African rangelands. Intermediate Technology Publications, London, UK.

Turner, M. D. 2011. The new pastoral development paradigm: engaging the realities of property institutions and livestock mobility in dryland Africa. Society and Natural Resources **24**:469-484.

Turton, D. 1991. Movement, warfare, and ethnicity in the lower Omo Valley. Pages 145-169 *in* J. G. Galaty and P. Bonte, editors. Herders, Traders, and Warriors: Pastoralism in Africa. Westview Press, Boulder.

Vetter, S. 2005. Rangelands at equilibrium and non-equilibrium: recent developments in the debate. Journal of Arid Environments **62**:321-341.

von Wehrden, H., J. Hanspach, P. Kaczensky, J. Fischer, and K. Wesche. 2012. Global assessment of the non-equilibrium concept in rangelands. Ecological Applications **22**:393-399.

Western, D. 1982. The environment and ecology of pastoralists in arid savannas. Development and Change **13**:183-211.

Zeidane, M. O. 1999. Le rôle des institutions provisoires dans la gestion des ressources à propriété commune des terries arides en Mauritanie. Pages 47-75 *in* M. Niamir-Fuller, editor. Managing mobility in African rangelands. FAO and Beijer International Institute of Ecological Economics, London.

Appendix 3. Supplemental materials for Chapter 5

Niels H. Batjes – ISRIC, World Soil Information

This Appendix summarizes the data and GIS-based mapping procedures used to assess SOC stocks and SOC content in "grazing lands" of Sub Sahara Africa, as presented in Chapter 5 of the report, as well as some additional results and discussion.

Data

The study uses the following point and spatial datasets:

a) Soil-geographic data: The mapping unit composition for Africa, in terms of their component FAO soil units (FAO, 1988), was derived from the Harmonised World Soil Database, v1.2 (FAO *et al.*, 2012). For selected regions, such as sections of Namibia, the soil-geographic was updated using more recent materials (Jones *et al.*, 2013). The grid cell size of HWSD is 30x30 arc-second or ~1km x 1km at the equator; the underpinning base maps are at a broad scale of 1:1-1:5 million.
b) Soil profile data: Data for organic carbon content, bulk density and proportion of coarse fragments (>2 mm), for defined soil depths (horizons), were extracted from the ISRIC-WISE database (Batjes, 2009, 2011). These data were subsequently used to calculate depth-weighted SOC contents for each profile using procedures described in Batjes et al (2011).
c) Land cover: Areas mapped for various typologies of "grazing lands" vary substantially between GIS datasets as definitions are not consistent between studies (White *et al.*, 2000; Mayaux *et al.*, 2003; Suttie *et al.*, 2005), despite ongoing standardization efforts (Allen *et al.*, 2011). Based on intensive discussions, the SOC-SSA project team finally adopted the 1x1km "Global Livestock Production Systems" (GLPS) map of Robinson et al. (2011, p.146) as most appropriate for the present broad scale analyses.
d) Climate: Differences in regional climate, as considered in the GLPS definitions (Robinson *et al.*, 2011), provided the basis for the present stratification by climate.
e) Country information: Sub-Sahara Africa (SSA) is defined according to United Nations (UN, 2012) definitions; it consists of Africa minus Northern Africa (Algeria, Egypt, Libya, Morocco, Tunisia, the Sudan and disputed Western Sahara). Country boundaries were taken from ESRI's World Administrative Regions GIS-layer.

Methodology

The GIS-based mapping procedure include three main steps:

a) The global complement of soil profiles held in the ISRIC-WISE database (> 13,000) was analysed using taxotransfer procedures that consider differences in FAO soil type (28) (FAO, 1988), soil texture classes (5) and soil depth (7) (see Batjes *et al.*, 2007; Batjes, 2012), plus climate (Peel *et al.*, 2007) as an additional criterion for clustering the profile data (For example, sandy soils (Arenosols) from Aridic zones will generally have lower mean SOC contents than those from Temperate zones).
b) Procedures were developed for computing soil carbon stocks and area-weighted SOC content to 0-0.3m, 0-0.5, 0-1 m and 0-2 m that consider LGPS class, broad climate classes as well as FAO soil type, building on earlier automated procedures (Batjes *et al.*, 2007; Batjes, 2012).

c) The various spatial data layers were combined and analysed using GIS overlays and subsequent SQL queries.

Results

Main differences in area-weighted mean SOC content by GLPS (Robinson *et al.*, 2011) are discussed in the body of this report (Section 1.X). However, in view of imposed limitations of length for the body of the report, the importance of regional differences in main soil type in determining mean SOC content in a given GLPS, as expressed in its classification, will be illustrated here for the Rangelands (LG, Table 5) and topsoil (0-0.3m).

Table 5. Mean soil organic carbon (SOC) content for selected FAO soil types represented in rangeland livestock production systems in Sub Sahara Africa

Major soil group (FAO, 1988)	Livestock production system [a]	SOC content (kg C m^{-2})				
		0-0.3 m	0-0.5 m	0-1m	0-0.5 m	0-2 m
AC – Acrisols[b]	LG Rangelands Arid	3.58[c]	4.71	6.63	8.02	9.07
	LG Rangelands Humid	4.09	5.46	7.74	9.26	10.50
	LG Rangelands Hyperarid	3.24	4.26	6.21	7.72	8.89
	LG Rangelands Temperate	4.11	5.34	7.05	8.10	9.01
AL – Alisols	LG Rangelands Arid	5.04	7.14	10.15	12.05	14.05
	LG Rangelands Humid	5.93	7.56	9.62	10.94	12.05
	LG Rangelands Temperate	7.33	9.98	14.48	17.17	18.80
AN – Andosols	LG Rangelands Arid	9.88	14.10	20.27	25.26	29.68
	LG Rangelands Humid	10.11	14.48	20.61	25.25	29.02
	LG Rangelands Hyperarid	1.85	2.77	4.13	5.11	9.04
	LG Rangelands Temperate	10.12	14.40	20.5	25.12	29.91
AR – Arenosols	LG Rangelands Arid	1.47	2.08	3.02	3.75	4.42
	LG Rangelands Humid	2.86	3.89	5.53	6.80	7.80
	LG Rangelands Hyperarid	1.21	1.71	2.46	3.05	3.86
	LG Rangelands Temperate	2.85	3.80	5.14	6.18	7.03
VR – Vertisols	LG Rangelands Arid	4.71	6.65	10.26	12.60	14.82
	LG Rangelands Humid	5.78	7.99	11.88	14.20	16.50
	LG Rangelands Hyperarid	4.70	6.43	9.35	11.19	13.02
	LG Rangelands Temperate	6.19	8.57	12.70	15.10	17.63

[a] Global livestock production systems according Robinson et al. (2011) on et al, see Table 1. [b] For a brief characterization of these soil types see (Bridges *et al.*, 1998). [c] Coefficients of variation for SOC content, per stratum, are generally greater than 50%.

Coarse textured, droughty and nutrient poor Arenosols, for example, have a mean SOC content of 1.2 kg C m^{-2} in the Hyperarid zone, 1.5 kg C m^{-2} in the Arid zone, and about 2.9 kg C m^{-2} in the Humid and Temperate zone. In comparison, for the same livestock production system, relatively fertile, fine-textured, swelling-and-shrinking clays soils (Vertisols) have greater mean SOC contents than coarse textured (sandy) Arenosols: 4.7 kg C m^{-2} in the Arid zones, 5.8 kg C m^{-2} in the Humid zone and 6.2 kg C m^{-2} in the Temperate zone.

The importance of nutrient status and clay mineralogy and is illustrated further for Acrisols (base saturation <50% and CEC_{clay} <24 $cmol_c$ kg^{-1}) and Alisols (base saturation <50% and CEC_{clay} >24 $cmol_c$ kg^{-1}), that typically are medium to fine-textured with a clay-illuviation horizon (Bridges *et al.*, 1998). Based on the available data, mean SOC content of Acrisols is 3.24 kg C m^{-2} in the Hyperarid zone, 3.58 kg C m^{-2} in the Arid zone, and some 4.1 kg C m^{-2} in the Humid zone and Temperate zone. Alternatively, Alisols, which are high activity clay soils (CEC_{clay} > 24 $cmol_c$ kg^{-1}), have a mean SOC content of 5.3 kg C m^{-2} in the Arid, 5.9 kg C m^{-2} in the Humid and 7.3 kg C m^{-2} in the Temperate zone. The importance of texture and clay mineralogy (as expressed here by clay activity), hence water holding capacity and infiltration rate, and overall soil fertility (expressed above by the base saturation), is also reflected in the figures for naturally fertile soils formed over volcanic ash (Andosols): 1.9 kg C m^{-2} in the Hyperarid zone, where rainfall is strongly limiting for biomass production, to around 10 kg C m^{-2} in the other zones.

Discussion

More detailed analyses of "present" SOC levels in SSA would require a wider selection of soil profiles for the region with well-documented land use and management histories. However, for most legacy soil profiles this information has not been reported in the source materials or, when available, using a diverse range of classification systems that prove cumbersome to harmonize to a common standard (Batjes, 2011; Leenaars *et al.*, 2014). Alternatively, in the near future, digital soil mapping procedures that consider a temporal dimension (4-D; x,y, z and time), may be able to account for possible effects of land use and management changes on regional SOC stocks over the last decades (see Minasny *et al.*, 2010; Ross *et al.*, 2013; Hengl *et al.*, 2014), and this with defined levels of uncertainty at an increasingly fine resolution.

References (cited in Appendix)

Allen, V.G., Batello, C., Berretta, E.J., Hodgson, J., Kothmann, M., Li, X., McIvor, J., Milne, J., Morris, C., Peeters, A., Sanderson, M., The, F., Grazing Terminology, C., 2011. An international terminology for grazing lands and grazing animals. Grass and Forage Science 66, 2-28.

Batjes, N.H., 2009. Harmonized soil profile data for applications at global and continental scales: updates to the WISE database. Soil Use and Management 25, 124-127

Batjes, N.H., 2011. Soil organic carbon stocks under native vegetation - revised estimates for use with the simple assessment option of the Carbon Benefits Project system. Agriculture, Ecosystems & Environment 142, 365-373.

Batjes, N.H., 2012. ISRIC-WISE derived soil properties on a 5 by 5 arc-minutes global grid (ver. 1.2). ISRIC - World Soil Information, Wageningen, p. 57.

Batjes, N.H., Al-Adamat, R., Bhattacharyya, T., Bernoux, M., Cerri, C.E.P., Gicheru, P., Kamoni, P., Milne, E., Pal, D.K., Rawajfih, Z., 2007. Preparation of consistent soil data sets for SOC modelling purposes:

secondary SOTER data sets for four case study areas. Agriculture, Ecosystems and Environment 122, 26-34.
Bridges, E.M., Batjes, N.H., Nachtergaele, F.O. (Eds.), 1998. World Reference Base for Soil Resources: Atlas. ISSS Working Group RB, ISRIC-FAO-ISSS-Acco. Acco Press, Leuven.
FAO, 1988. FAO-Unesco Soil Map of the World, Revised Legend, with corrections and updates. p. 140.
FAO, IIASA, ISRIC, ISSCAS, JRC, 2012. Harmonized World Soil Database (version 1.2). Food and Agriculture Organization of the United Nations, International Institute for Applied Systems Analysis, ISRIC - World Soil Information, Institute of Soil Science - Chinese Academy of Sciences, Joint Research Centre of the European Commission, Laxenburg.
Hengl, T., Mendes de Jesus, J., MacMillan, R.A., Batjes, N.H., Heuvelink, G.B.M., Ribeiro, E.C., Samuel-Rosa, A., Kempen, B., Leenaars, J.G.B., Walsh, M.G., Gonzalez, M.R., 2014. SoilGrids1km— global soil information based on automated mapping. PLoS ONE 9, e105992.
Jones, A., H., B.-M., Brossard, M., Dampha, A., Deckers, J., Dewitte, O., Gallali, T., Hallett, S., Jones, R., Kilasara, M., Le Roux, P., Micheli, E., Montanarella, L., Spaargaren, O., Thiombiano, L., Van Ranst, E., Yemefack, M., R., Z. (Eds.), 2013. Soil Atlas of Africa. European Commission, Publications Office of the European Union, European Union, Luxembourg.
Leenaars, J.G.B., Kempen, B., van Oostrum, A.J.M., Batjes, N.H., 2014. Africa Soil Profiles Database: a compilation of georeferenced and standardised legacy soil profile data for Sub-Saharan Africa In: Arrouays, D., McKenzie, N., Hempel, J., de Forge, A.R., McBratney, A. (Eds.), GlobalSoilMap - Basis of the global spatial information system. Taylor & Francis Group, Orléans pp. 51-57.
Mayaux, P., Bartholomé, E., Massart, M., Cutsem, C.V., Cabral, A., Nonguierma, A., Diallo, O., Pretorius, C., Thompson, M., Cherlet, M., Pekel, J.-F., Defourny, P., Vasconcelos, M., Gregorio, A.D., S.Fritz, Grandi, G.D., C.Elvidge, P.Vogt, Belward, A., 2003. A land cover map for Africa. Joint Research Centre, European Commission, Luxembourg.
Minasny, B., Sulaeman, Y., McBratney, A.B., 2010. Is soil carbon disappearing? The dynamics of soil organic carbon in Java. Global Change Biology 17, 1917–1924.
Peel, M.C., Finlayson, B.L., McMahon, T.A., 2007. Updated world map of the Köppen-Geiger climate classification. Hydrol. Earth Syst. Sci. Discuss 11, 1633-1644.
Robinson, T.P., P.K., T., Franceschini, G., Kruska, R.L., Chiozza, F., Notenbaert, A., Cecchi, G., Herrero, M., Epprecht, M., Fritz, S., You, L., Conchedda, G., See, L., 2011. Global livestock production systems. . Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI), rOME, p. 152.
Ross, C.W., Grunwald, S., Myers, D.B., 2013. Spatiotemporal modeling of soil organic carbon stocks across a subtropical region. Science of The Total Environment 461–462, 149-157.
Suttie, J.M., Reynolds, S.G., Batello, C., 2005. Grasslands of the world. FAO, Rome, p. 495 + app.
UN, 2012. Classification of countries by major area and region of the World. Population Divison, Department of Economic and Social Affairs, United Nations.
White, R.P., Murray, S., Rohweder, M., 2000. Pilot Analysis of Global Ecosystems: Grassland Ecosystems. World Resources Institute, Washington, DC.