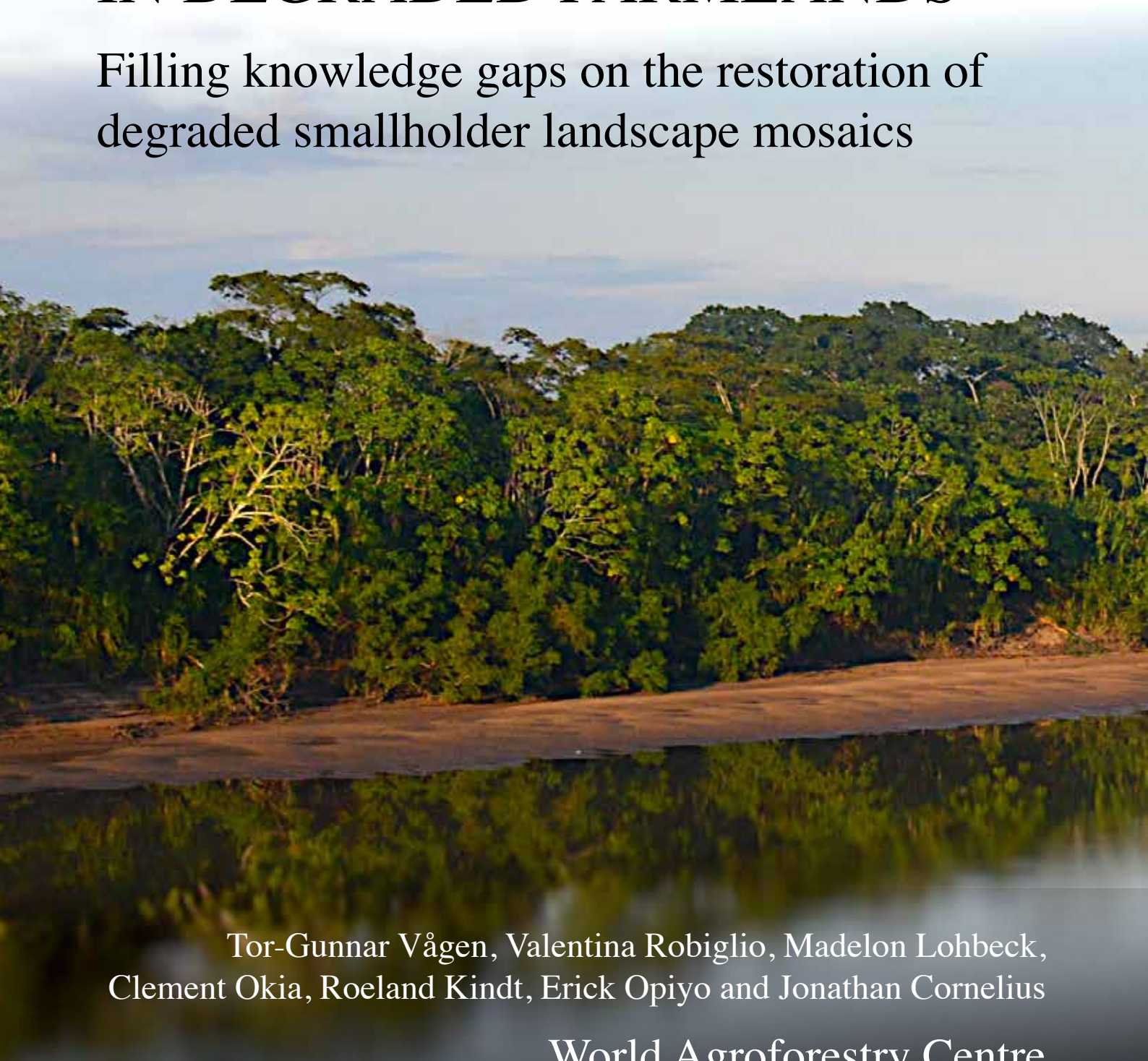
Department for International Development

IUCN

World Agroforestry Centre

THE POTENTIAL FOR FOREST LANDSCAPE RESTORATION IN DEGRADED FARMLANDS

Filling knowledge gaps on the restoration of degraded smallholder landscape mosaics



Tor-Gunnar Vågen, Valentina Robiglio, Madelon Lohbeck, Clement Okia, Roeland Kindt, Erick Opiyo and Jonathan Cornelius

World Agroforestry Centre

Executive summary

This report is the final technical report for work package 2 (WP2) of the project "Filling Knowledge Gaps on the Restoration of Degraded Smallholder Landscape Mosaics". The main objective of this work package was to illustrate the utility of comparative studies in scaling up forest landscape restoration (FLR) projects by combining systematic field-based methodologies with advanced remote sensing analyses to provide real-time as well as past assessments of key indicators of land degradation. These methods and approaches allow for the prediction of indicators of ecosystem health to be made for large landscapes.

The report also outlines advances in soil and land health monitoring that enable efficient baseline assessments and monitoring of impacts towards restoration of degraded forest landscapes. Indicators for assessment and monitoring of land degradation should be:

- Science based,
- Readily measurable (quantifiable),
- Rapid,
- Based on field assessment across multiple scales (plot, field, landscape, region) and
- Representative of the complex processes of land degradation in landscapes.

Further, when assessing the restoration potential for a given area, robust analytical frameworks are needed that explicitly incorporate scale dependencies.

We outline approaches for FLR and highlight assessments of land health, using indicators such as fractional vegetation cover, soil erosion, root-depth restrictions, soil organic carbon (SOC) and infiltration capacity. We show how these indicators can be used to identify degraded areas, and to identify options for restoration such as selection of suitable tree species for restoration of degraded soils or erosion control.

Given the complexity of the study landscapes, and their high level of fragmentation including small plot sizes, spatial assessments of land health indicators were made a spatial resolution of under 0.1 ha using and Landsat 8 remote sensing imagery. Both landscapes have a number of different agroforestry systems and practices.

Assessments of land degradation using soil erosion as a key indicator were developed indicating that 22% of the site in Peru was eroded and 83% of the two sites on Mount Elgon in Uganda were eroded. When considering cultivated areas only, 54% and 90% of the surveyed plots were eroded in Peru and Uganda, respectively.

Analysis of tree species diversity and abundance in the two study areas showed that *Eucalyptus grandis*, which is an exotic species and *Markhamia lutea*, which is indigenous were most dominant in the Uganda sites. The latter has a number of important utilities and is also a species that is suitable for restoration of degraded soil and erosion control. In Peru, the most dominant species recorded were *Cecropia sp, Vernonia sp, Inga thibaudiana* and *Vismia sp*.

We present options for restoration with native tree species in the Peruvian Amazon and in Uganda. Based on the findings of the study, the two study areas have widely different options available for FLR.

In the Mbale landscape in Uganda, lack of available land means that there is very limited scope for fallows. Only 1% of the area surveyed in Mbale was under fallow, for example. This means that the main restoration options for this landscape are native species to increase species diversity in existing agroforestry systems, reduce erosion and restore soil health.

In Campo Verde, restoration through fallows is a viable option on the other hand, but restoration efforts need to focus on increasing tree species diversity in fallows. Also, our analysis shows that this area is prone to flooding and water-logging due to very low infiltration capacity, which means that tree species that are tolerant to some water-logging should be used in restoration efforts.

The project has also developed a smart phone app for the identification of suitable species for restoration of specific areas in the Uganda study area, as well as collection of data on trees and land use through crow-sourcing.

The approaches presented as part of this report represent an important step towards more effectively identifying land degradation hotspots, targeting interventions and communicating priorities for land degradation prevention or restoration to stakeholders.

Background

Land degradation impacts the health and livelihoods of about 1.5 billion people (FAO, 2011) worldwide. Further, most of the world's remaining tropical forests are strongly modified by humans and degraded, particularly through agricultural expansion, which has caused rapid loss of tropical forest (Geist and Lambin, 2002; Kissinger et al., 2012), loss of biodiversity and a simplification of the landscape matrix. Human-modified landscapes are socio-ecological systems, whose integrity and resilience depend on complex interactions between people and their environment. Given that the state of the environment and food security are strongly interlinked in tropical landscapes, the increasing need for land for food production, urbanization and other uses pose several threats to sustainability in the long term (Beddington, 2011; Foresight, 2011).

The tropics contain ecosystems that sustain a large proportion of the world's biodiversity, but management of protected areas is often poor and hence fails to effectively protect biodiversity (Laurance et al., 2014). With human populations projected to reach 11 billion this century and agricultural expansion likely to increase, it is expected that negative trade-offs between food production and nature conservation will intensify (Laurance et al., 2014). There is increasing recognition that more integrated approaches to landscape planning are needed in order to reconcile local actor strategies with global conservation goals and to alleviate these negative trade-offs, allowing for competing land-uses while considering external influences and local needs (Sayer et al., 2013) in order to reconcile local actor strategies with global conservation goals. However, the analytical foundations for measuring and understanding socio-ecological processes remain a challenge, particularly in order to make such integrated approaches operational (Mace, 2014).

Restoration of degraded lands is now a global priority (Aronson and Alexander, 2013), with the Bonn Challenge aspiring to restore 150 million hectares of the worlds deforested and degraded lands by 2020 (Minnemeyer et al., 2011). Forest landscape restoration provides opportunities to counteract degradation and restore many ecosystem functions, including components of biodiversity (Benayas et al., 2009; Chazdon, 2008) and soil functions (Smith et al., 1999). According to Laestadius et al. (2011), more than two billion hectares offer opportunities for restoration for example through the creation of landscape mosaics and increased landscape heterogeneity. Forest restoration can increase the abundance of secondary forests, which provide many important functions and services to local people, and are often the only significant source of forest resources available to the rural poor (Smith et al., 2001). Examples of these functions and services include: hunting grounds, wood for construction, firewood, fodder and shade for animals, and access to medicinal, fibre and food species (Chambers et al., 1989; Chazdon and Coe, 1999; Junqueira et al., 2010; Voeks, 1996). In addition secondary forests have high potential rates of carbon sequestration responding to the global need to mitigate climate change (Bongers et al., 2015; Feldpausch, 2004; Guo and Gifford, 2002; Vågen et al., 2005) and may have the potential to bring benefits to local people in the form of carbon credits or REDD-like schemes in the future.

Despite clear economic, social and environmental rationales for preventing or reversing land degradation, there is often limited political will to ensure that effective measures are indeed implemented. In addition, costs can be prohibitive when attempting to restore severely degraded ecosystems and landscapes. Hence, measures or incentives that favor reforestation and restoration of degraded lands need to be combined with assessments of land degradation status and trends in order to provide farmers or other stakeholders with realistic options for restoration.

A critical component of FLR is that it needs to consider a landscape as a unit. This does not mean that complexity in landscapes is ignored, but rather that the potential for restoration explicitly considers scale across a landscape as well as the multiple functions that a landscape has, included nested scales of social and ecological systems within landscapes.

While this report focuses primarily on the biophysical aspects of restoration and hence on ecological indicators, including assessments of land degradation, social aspects also need to be considered when developing and implementing restoration options.

Acknowledgement

This study was conducted in collaboration with IUCN as part of the Knowledge and Tools for Forest Landscape Restoration (Know-For-FLR) project, which is supported by the UK government.

1. Ecological indicators

The purpose of ecological indicators is to measure the state of the environment, including to diagnose environmental problems and their causes (Dale and Beyeler, 2001). While many efforts to assess the condition of the environment have used a very limited set of indicators that do not allow one to capture the complexity of ecosystems, there is sometimes also a conundrum of indicators, which are difficult to measure or to handle analytically. Further, many efforts to develop indicators have failed due to a lack of scientific rigor and/or defined protocols for data collection. Hence, systematic data collection and management approaches are needed, backed by scientifically rigorous methods for data analysis and inference. Ideally, indicators are easy-to measure proxies that reflect more complex underlying processes.

Parrott (2010) suggested a number of measures of complexity, including (i) temporal measures or time-series which have Shannon entropy or derivatives thereof, (ii) spatial measures such as soil properties, vegetation cover or land degradation (Vågen et al., 2015a, 2013), (iii) spatiotemporal measures which integrate both space and time, and (iv) structural measures which describe how components of a system are organized. In the measurements of all of the above indicators, the various scales in space and time they play out at also need to be considered. This means that methods for sampling of ecological indicators should take scale implicitly into consideration.

1.1. Land degradation

Indicators of land degradation are a subset of ecological indicators related to degradation processes such as soil erosion, salinization, water stress, forest fires and overgrazing. In the current project we measure or observe a number of different indicators of land degradation at multiple spatial scales. These indicators were chosen to reflect land degradation risk, vegetation condition, land use, biodiversity and soil health, and included:

- Soil erosion prevalence
- Root-depth restrictions (compaction)
- Vegetation
  - Herbaceous vegetation cover density
  - Woody vegetation cover and structure
  - Forest cover and distribution
  - Tree biodiversity
- Soil health
- Land use

2. Forest landscape restoration

Forest landscape restoration aims to restore ecological integrity while enhancing human well-being. It is a process of reconciling competing interests to formulate viable restoration goals, matching them with realistic restoration strategies and balancing the localities within the landscape context (Figure 1 and Table 1). Notably, forest landscape restoration is about restoring ecological integrity, given current land uses, local actor interests and strategies, and land degradation status.

What results from successful restoration is a multi-functional landscape in which nature and people thrive. Therefore the aim is generally a set of functions specified by local actors.

An important consideration in any landscape restoration effort is to determine what one is trying to restore. Forest systems generally have ecologies that are complex so restoring a forested landscape is about much more than simply replacing a community of trees (Wilson et al., 1999). We often do not know what the original forest ecosystem was in a particular location and even where we do know a great deal about it, reestablishing what was originally there is often not an option due to considerations such as current land use, people and land degradation status. The aim will therefore generally be to restore important functions of forest ecosystems and set the stage for natural regeneration to happen where possible. In cases where ecosystems are severely degraded, it may not make economic sense so actively restore them.

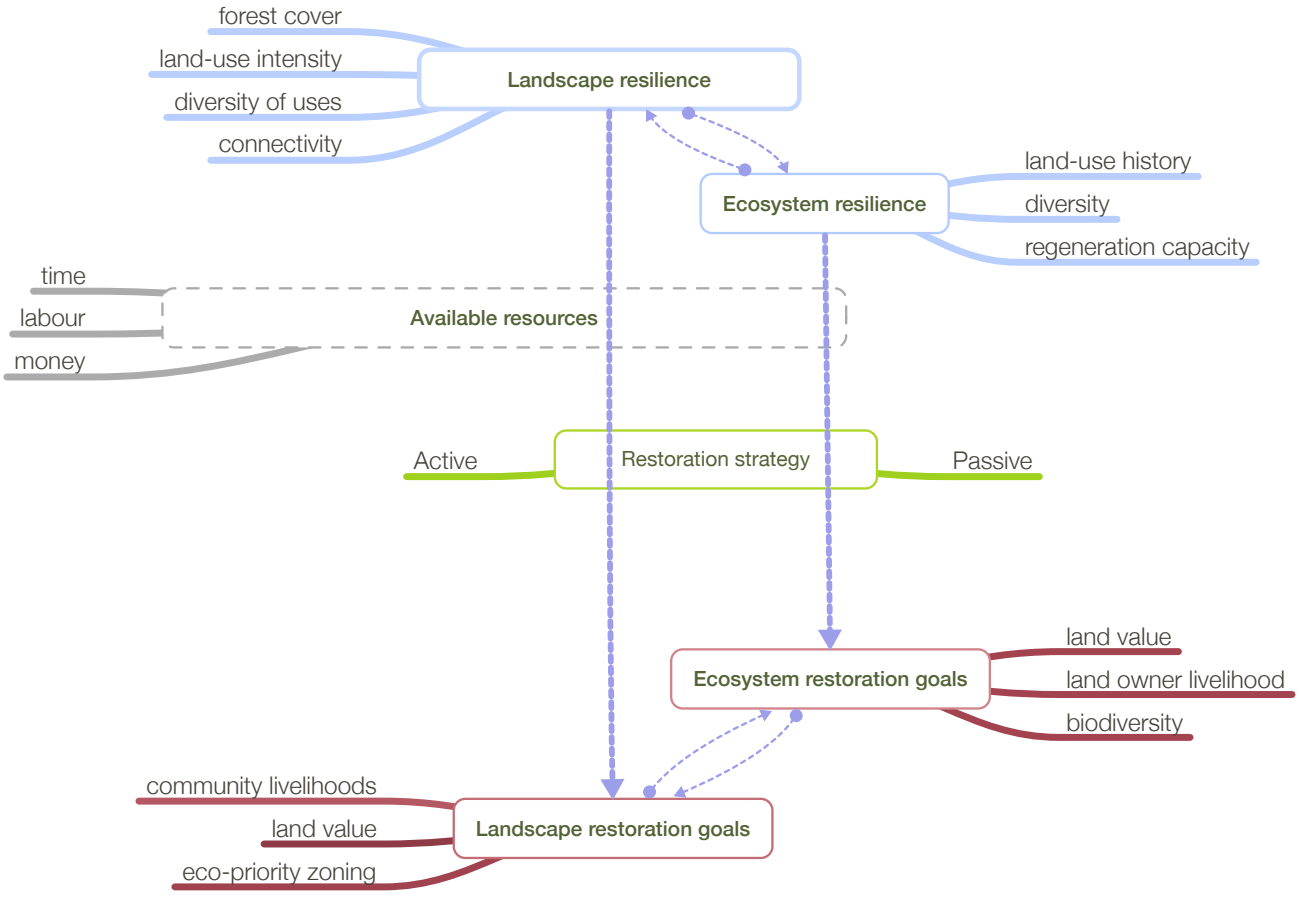


Figure 1. Mosaic restoration framework illustrating how, based on the resilience of the ecosystem (less degradation leads to higher resilience), a given restoration strategy needs to contribute towards specified restoration goals. The resources available, as well as the resilience of the system, limit the strategy that can be taken and thereby the goals that can be achieved. Restoration strategies range from passive restoration (having low costs & labour requirements, longer timeframe, only works in relatively resilient systems) to active restoration (high costs & labour requirements, faster, suitable for very degraded systems). Ecosystem resilience is embedded into landscape resilience. Similarly ecosystem restoration goals are embedded in the landscape restoration goals. Orange boxes give some important variables that shape the factor (these are not exhaustive). See Table 1 for an example of how this framework can be applied to different landscape elements.

Table 1. Linkages between the state of the landscape or landscape elements (blue), restoration strategies (green), and restoration goals (red) (colors match the mosaic restoration framework in Figure 1). Landscape elements range from a. specific types of land use (drivers of degradation relate to the intensified use thereof), to b. more general types of land use elements (drivers of degradation can occur across land uses), to c. landscape mosaics that contain a combination of different drivers. Note that different restoration goals can be thought of, not just the ones in the table, though changing the goal will concomitantly change the strategy. Also different drivers of degradation can occur simultaneously, often exacerbating degradation rapidly.

Landscape element	Driver of degradation	Indicator of degradation	Restoration strategy	Restoration goal
a Old fallow	Overuse (intensive crop system)	Low diversity, high stem density	Enrichment planting (active) or natural regeneration (passive) depending on level of degradation	Conservation forest
Plantation	Overuse (intensive use)	Monospecific, poor productivity and regeneration, exotic spp	Enrichment planting (active) or natural regeneration (passive) depending on level of degradation	Diverse native species plantation
Logged forest	Overuse (overharvesting)	Damage, lianas, logging trails, stumps	Liana cutting, silvicultural treatments, enrichment planting or natural regeneration depending on level of degradation	Reduced impact logging
Recent fallow	Overuse (many crop cycles, short fallows)	Dominating persistent weeds	Enrichment planting of trees that outshade the weed, natural regeneration probably not feasible	Agrosuccessional system
Cropfield	Overuse (intense use)	Poor productivity, soil exhausted, possibly erosion	Enrichment planting with trees and herbs that improve and stabilize the soil, or natural regeneration (introducing a fallow) depending on level of degradation	Agroforestry system
Pasture	Overuse (overgrazing)	Bare soil, erosion, trampling	Active planting with fodder trees, grasses and legumes, depending on level of degradation natural regeneration (passive) is an option	Silvopastoral system
b Any (applies to different land uses/ landscape elements)	Fire Weeds Erosion/ landslides	Burnmarks, little & low diverse regeneration Dominating weeds, little & low diverse regeneration Gullies, bare soil, substrate absent	Firebreaks, prescribed burning, enrichment planting of trees that outshade the weed or and/or ground-covering herbs for stabilisation. Focus on slopes and hilltops. Natural regeneration probably not feasible.	Healthy & productive ecosystems and landscapes
Protective land (riverbanks, slopes, hilltops)	Erosion/ landslides	Gullies, bare soil, substrate absent	Enrichment planting with trees and ground covering herbs for stabilisation, probably natural regeneration is no option. Focus on slopes and hilltops. Terracing.	Stabile and healthy soils
Forest (including regrowth, plantation, conserved)	dispersal limitation (hunting or source)	Little & low diverse regeneration	Enrichment planting with trees that attract animals/dispersers (large-seeded spp). Landscape level measures that promote connectivity between forests (corridors, stepping stones).	Healthy & resilient forests
c Landscape mosaic	Combination of the above	Diversity of land uses, % forest cover and connectivity	Combination of the above	Multifunctional & resilient landscapes

Also, degradation and restoration pathways tend to be very different, which has major implications for the selection of restoration strategies.

In the following sections we outline some of the main strategies for restoration of forest landscapes, focusing particularly in natural- and active restoration strategies.

2.1. Natural regeneration

Natural regeneration is a form of passive restoration where active forest management is discouraged or strongly limited in order for natural succession to take place. Depending on the level of disturbance, natural regeneration can lead to a close replication of the forested ecosystem that was there prior to it being degraded. However, in most cases the level of disturbance is such that the naturally regenerated system is quite different from what was originally there.

Secondary succession leads to a forested ecosystem that harbors many of the functions (such as soil nutrient stocks, productivity) that were present in the forest prior to it being degraded, although the original composition and structure may never return. The time it takes to restore ecosystem functions depends on the specific function, the level of degradation (e.g. land-use history and land degradation status) and on the quality of the landscape matrix.

Despite numerous studies of secondary succession, our ability to predict successional pathways remains poor. Restoration through secondary succession is generally a relatively low-cost restoration strategy. However, in most cases the level of disturbance is such that the natural regeneration will not take place, is very slow, or where the naturally regenerated system leads to a very different and undesirable state of the ecosystem. In those cases active restoration is a better option.

2.2. Assisted restoration

Given the time required for natural regeneration to take place, some form of active restoration is often needed to accelerate the (natural) recovery process. Where forest degradation is severe and soil erosion has resulted in the loss of topsoil, for example, active restoration might be necessary as the system would not recover otherwise (Lamb et al., 2005). An important aspect of forest landscape restoration is therefore to identify areas where thresholds may have been exceeded due to for example soil erosion in order to spatially target efforts of active restoration.

Examples of active restoration include:

- Stabilization of soils through soil and water conservation measures
- Restoration plantings by:
    1. Creating a canopy cover by planting a small number of fast-growing but short-lived tree species, or
    2. Using a greater diversity of trees that represent more mature successional stages.

2.2.1. Restoration in mosaic landscapes

Most degraded tropical landscapes consist of a mosaic of land uses and land cover types, including residual forests and cultivated land (Figure 2). Mosaic landscape restoration aims to restore overall landscape productivity by targeting both restoration of natural vegetation cover (promoting succession) and restoration of degraded agricultural areas (Chazdon, 2008).

The overall goal of mosaic restoration is to improve local livelihoods, biodiversity conservation and socio-ecological resilience. Generally, this type of restoration strategy is suitable where population densities are higher. The result is a mix of forest, trees and other land uses including agroforestry and smallholder agriculture, ideally including corridors of native vegetation to enable wildlife movement.

An important consideration in the restoration of mosaic landscapes is the restoration of individual sites versus landscapes. The latter is generally more effective, but means that entire communities have to be involved in restoration efforts, including in deciding the proportional cover and distribution of forest patches and/or corridors within a landscape. A landscape approach is also more likely to produce the desired outcome of improved livelihoods (Lamb et al., 2005).

2.2.2. What species to plant?

Selection of species to plant should be guided by the restoration goals, and may be limited by the ability to collect or buy seeds, grow them into seedlings or by seedling availability on the market. In cases where natural regeneration is taking place, selected species can be promoted over others, for example by removing weeds and other competing species.

Different tree species are known to differ in their ability to grow under certain conditions. For example, old-growth forest species may be important for restoration goals (for example due to their high-quality

Figure 2. Typical mosaic landscape on Mt Elgon in Uganda.

wood or desired fruits), but typically these will not grow well in very degraded sites. Pioneer species on the other hand are specifically adjusted to these conditions and would be better candidates for restoring degraded sites.

Framework tree species are indigenous forest tree species, planted to complement and accelerate natural regeneration of forest ecosystems and encourage biodiversity recovery on degraded sites. Functional traits can be central to predicting species success in a given environment (Lohbeck et al., 2013) and thereby for predicting species success in restoration plantings (Gondard et al., 2003; Martínez-Garza et al., 2013). Functional traits may also determine or enhance the overall functioning of the ecosystem, such as decomposition rates or productivity (Lohbeck et al., 2015).

For example in a secondary forest in Panama, N2 fixing tree species had up to nine times faster carbon sequestration rates than non-fixing species (Batterman et al., 2013). Further, when planting species that increase the functional diversity or response diversity of an ecosystem, the system's resilience may be enhanced (Laliberté et al., 2010; Lohbeck et al., 2012).

Another aspect to consider in the selection of species to plant is the use or utility of the trees. Often landowners will want to increase the number of useful species that can generate revenues or increase food security, for example. A list of useful species for restoration plantings can be found through the World Agroforestry Centre website by following the following link:

http://www.worldagroforestry.org/resources/databases/agroforestree

Finally, information on useful tree species can be combined with information on the potential natural vegetation for a specific intervention area, provided this information is available. For Uganda, we use the map of potential natural vegetation of eastern Africa (VECEA) to identify species that are suitable for restoration in the project area. This ties in with work package 3 of the project and the development of a smart phone application for tree species selection based on the VECEA maps.

2.3. Landscape resilience

This is objective differs from the previous sub-objectives by explicitly including the landscape as a unit and thereby recognizing larger temporal and spatial scales. By doing so we consider complex mosaic landscapes as social-ecological systems whereby the resilience - the ability to withstand pressures and disturbances - depends on the interactions between land-use types and between people and the environment they depend on.

Indicators for socio-ecological resilience at the landscape level have been developed, and include; the retention and acquisition of traditional ecological knowledge, the use of indigenous languages, demographic characteristics of the communities, cultural values attached to the land and its species, customary laws, social institutions and autonomy, food sovereignty and self-sufficiency, multiple uses of land and species, complexity and intensity of interactions with the ecosystems and the conservation of resources (van Oudenhoven et al., 2012).

3. Case studies

3.1. Campo Verde, Peru

The Campo Verde site is located in the province of Coronell Portillo, Department of Ucayali in the Central /Eastern Amazon sector of Peru. It is 4.5 kilometers from the town of Campo Verde, in Campo Verde and Nueva Requena districts (Figure 3).

This area can be characterized as an "old deforestation frontier", and is currently predominantly degraded pasture land. Establishment of colonists in the areas started in the 1960s when families benefited from a national program in support to the "colonization" of the Amazon (Fomento del Gobierno Central para la Colonización de la Región Amazónica). Migrants started to convert forest into annual and biannual crops (rice, cassava, pineapple, beans, plantain) and fruit trees. Each family was allocated 20ha. Today, the growing influence of the booming center of Pucallpa, located at 30 km from the site is also evident.

In the 1970s, "Boot Peru" acquired some 2,892 ha of the site and established a cashew plantation, which was later sold to a group of investors who started to establish pastures (mostly with *Brachiaria brizantha* and *Brachiaria decumbens*). This land now belongs to the Compañía Agrícola Ganadera Pucallpa S.A., which generally limits its interventions to the maintenance of pastures, often rented to family farmers from bordering settlements. There are about 200 ha of remnant forest in the holding that has not been converted into pasture for lack of funds.

Another part of the site was allocated to the Sociedad Agrícola de Interés Social (SAIS) Túpac Amaru, a social enterprise that was established at the time of the agrarian reform with the purpose of employing 2600 families of 16 farmers communities (comunidades campesinas, originary from the Andean regions). At the time of the allocation of land to SAIS the enterprise decided to maintain 50% of the forest land for timber production, the rest was converted into pasture land with *Brachiaria decumbens*, associated with Kudzu (*Pueraria phaseoloides*) in parcels of 15 ha surrounded by a 50 m buffer of forest.

Besides planned land occupation by colonists and enterprises, unplanned spontaneous occupation of land by Andean migrants and migrants from the north and coastal areas started around the same years. Many of them invaded the land allocated to the private and social enterprises above. Three settlements were established in the study site (caserios) with conflicts over

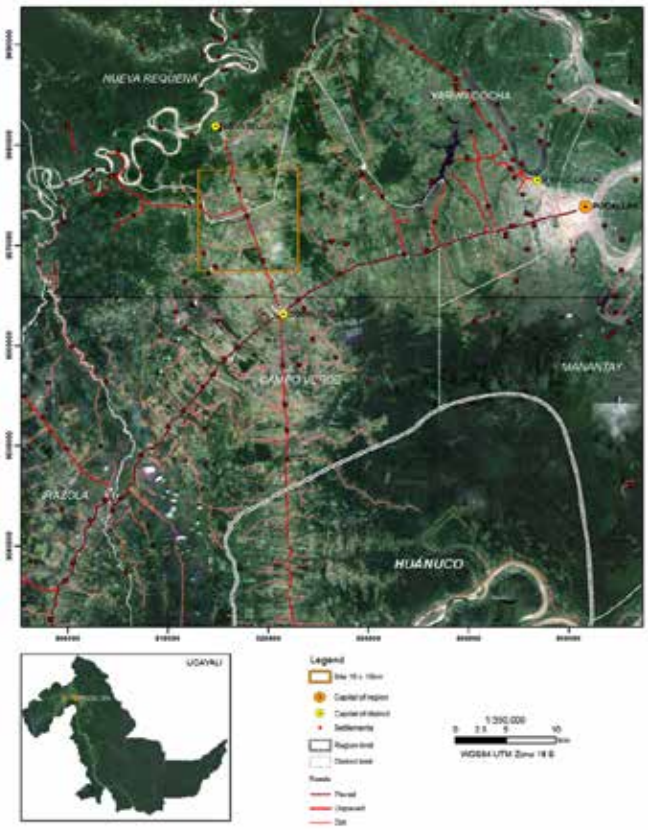


Figure 3. Location of the LDSF site of Campo Verde.

land continuing until 2007 with the legal entitlement of occupying informal farmers to their land. All these different actors contributed to forest conversion processes in different ways and magnitudes. The main difference between the big companies and small family farmers was that the companies converted land mostly to establish pastures, whereas family farmers, if not supported by large livestock sector projects, farmed mostly annual cash crops.

Generally, farmers would establish maize and rice fields after clearing the forest, sometimes including cassava for self-consumption. Richer farmers also established pastures. Progressively due to a lowering of productivity (due to a persistent use of fire and lack of soil conservation practices) production shifted towards less demanding cassava and many farmers started to grow coca (*Erythroxylum coca*) to complement their income. The presence of illicit crops attracted the intervention of public programs for eradication (PDA, Alternative Development Program), promoting permanent crops such as oil palm (*Elaeis sp*) and cacao (*Theobroma cacao*) to offset coca farming. Today, both cacao and oil palm are rapidly expanding in the area with strong support from public investments, in particular during the initial establishment phase.

Figure 5. Photographs from Campo Verde, showing (clockwise from top-left): Aguaje Palms in the northern part of the study area; Field establishment in a former fallow area; Cacao plantation; Herd of cattle Pueblo Libre.

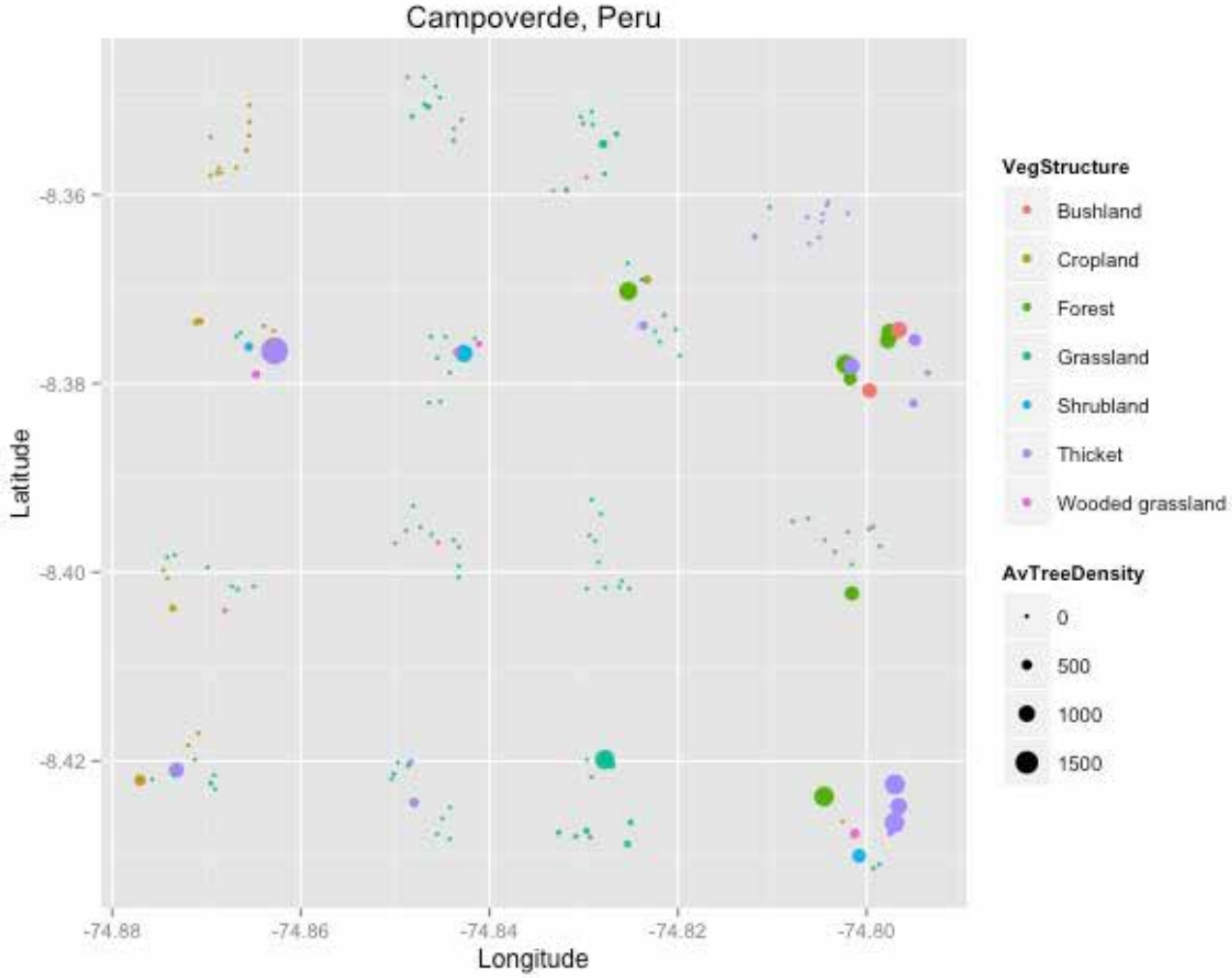


Figure 4. Spatial representation of vegetation structure classes and tree densities in Campo Verde.

3.2. Mount Elgon, Uganda

The Mt Elgon study area is located near the town of Mbale in western Uganda, which is the main municipal, administrative, and commercial center of Mbale District and the surrounding sub-region. According to estimates from 2014, the total population of Mbale town is just under 100,000. The study areas is characterized by smallholder mosaic farming systems with cultivation of maize, legumes, banana, coffee and a number of different annual cash crops, including cabbage. Eucalyptus plantations are widespread in the area, as well as a number of other exotic fruit and timber tree species.

The Ugandan National Environment Regulations for Mountainous and Hilly Areas Management established strict regulations that prohibit cultivation of slopes steeper than 15% to reduce land degradation in this area, but farming is widespread also on very steep slopes in the study area (Figure 6).

The main processes leading to land degradation in the area include intensive cultivation, fragmentation and loss of biodiversity, and indiscriminate removal of vegetation cover through grazing and fuel-wood collection. Population pressure is high, which presents some challenges in terms of restoration options, including potential leakage into surrounding areas and encroachment into the forest reserve. The latter is already under great pressure from shifting cultivation, and timber and firewood collection. A number of forest dwellers still live inside the park, primarily indigenous Benet people; pastoralists that practice some subsistence agriculture.

Soil erosion has been reported to be widespread in a number of studies, as a result of cultivation of steep slopes. Loss of soil fertility through soil degradation is a major challenge in the study area, although diverse agroforestry systems do exists, particularly around homesteads. Landslides are also a major problem in, having caused numerous fatalities.

The case study focuses on two landscapes on a topographic gradient from the town of Mbale, going east up Mount Elgon (Figure 7). Both sites are located in the district of Mbale, in Manjiya county of eastern Uganda and border Mount Elgon National Park, which was established in 1992. Mount Elgon (4321 m) is a solitary extinct volcano that strides the Kenya-Uganda border.

The area is characterized by a mountainous topography, slopes in the west and north include quite spectacular cliffs. Mean annual temperature is about 23°C and mean annual rainfall about 1,800 mm. The area is an important water catchment for the Turkwell and Lake Turkana systems, Lake Victoria, Lake Kyoga and the Nile Basin.

Figure 6. Cultivation of steep sloped in the upper part of the Mbale LDSF site.

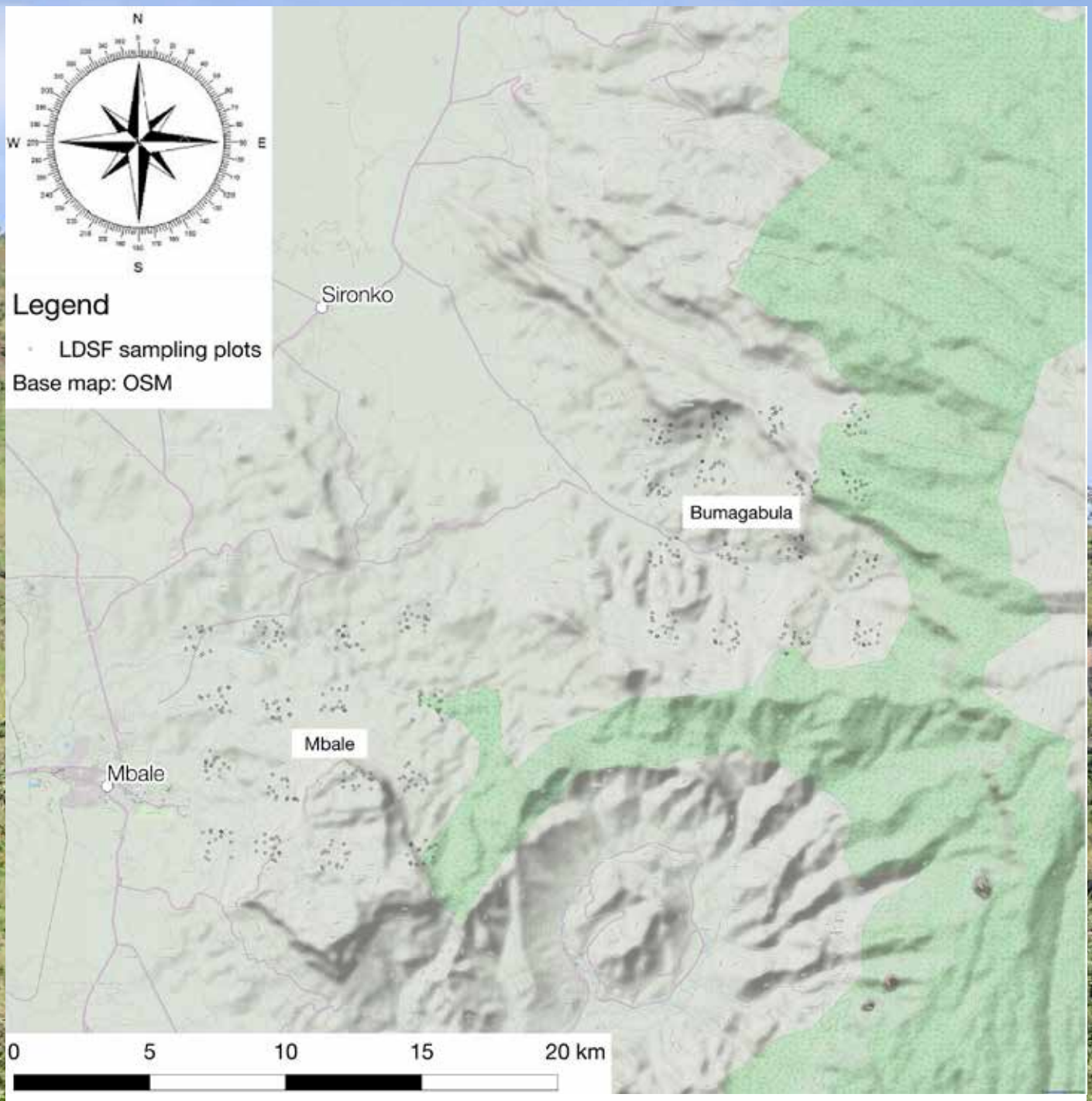


Figure 7. The Mbale and Bumagabula LDSF sentinel sites on Mt Elgon in Uganda. Source for background map: Open Street Maps.

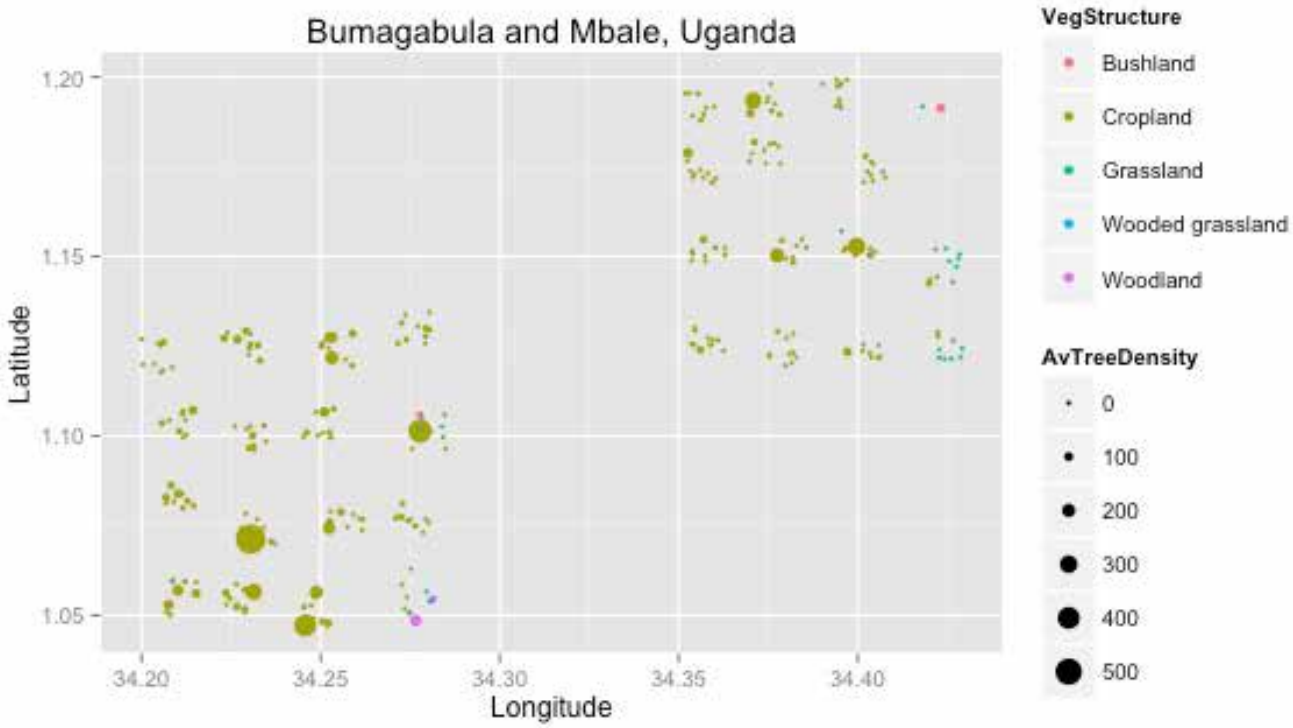


Figure 8. Spatial distribution of vegetation structure classes and tree densities in Mbale and Bumagabula.

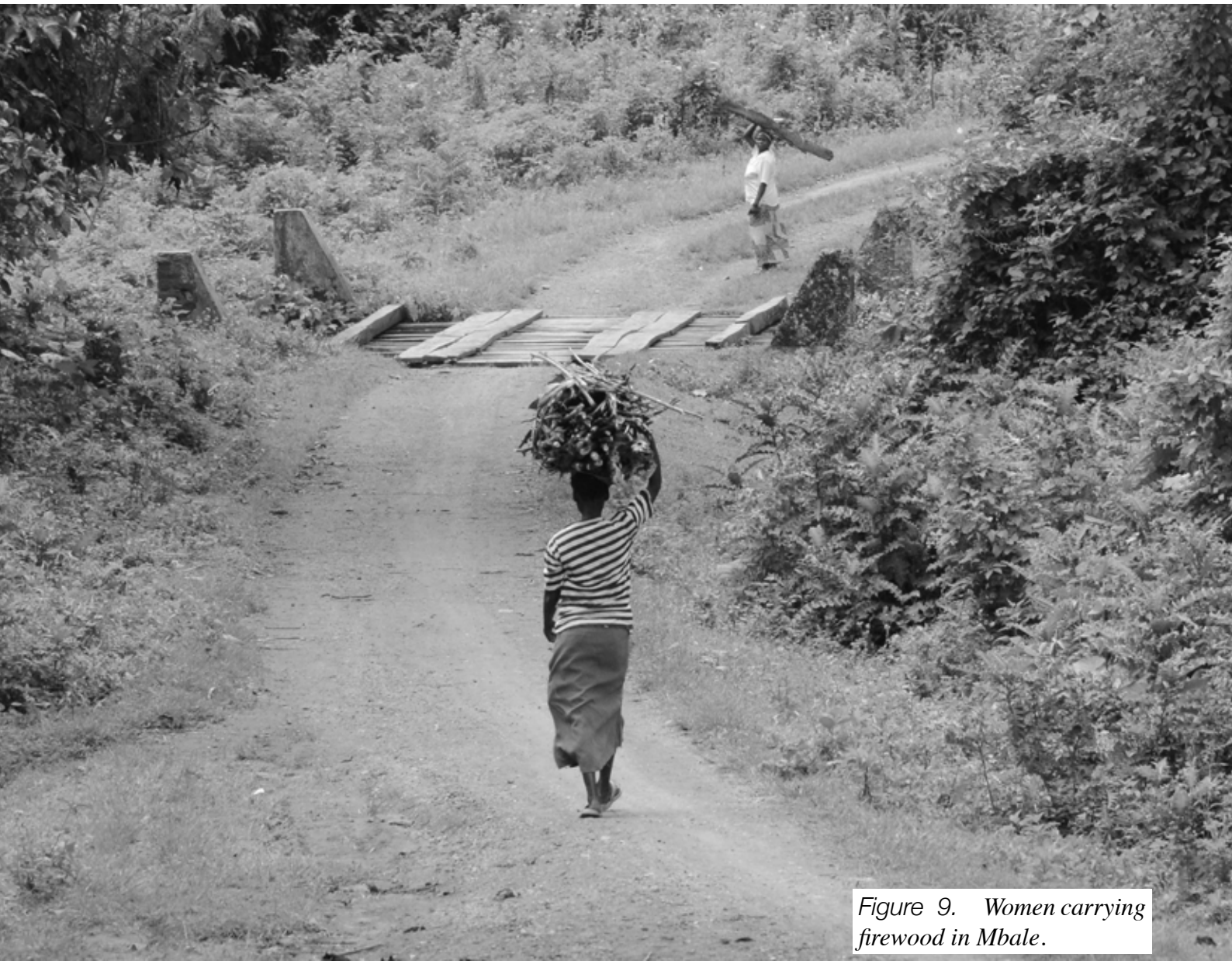


Figure 9. Women carrying firewood in Mbale.

4. Land health

Indicators of land health are important to effectively communicate information about land and soil health both to stakeholders such as farmers or advisory services, and to policy makers. Indicators are also critical when assessing environmental conditions and progress made towards the mitigation or avoidance of land degradation, through for example restoration of forest landscapes. However, many attempts at quantifying land degradation or assessing restoration potential have failed to operationalize the use of indicators of land health, mainly because indicators are used that are often not feasible to measure, combined with a lack of rigorous science-based analytical frameworks.

In this project we show the utility of systematic field surveys, combined with the application of robust analytical methods and tools to simultaneously summarize variations in multiple measures of ecosystem function, providing critical spatially explicit evidence for the assessment of land degradation status and restoration potential. Examples of some of the most important indicators include soil erosion, soil organic carbon concentrations and stocks, soil compaction, soil acidity, soil fertility parameters, vegetation dynamics (rather than just productivity), infiltration capacity, and biodiversity. All of these indicators can be readily measured on the ground and/or from remote sensing.

Accelerated soil erosion is arguably the most important indicator of land health and also one of the most widespread forms of land degradation worldwide. Interactions between soil erosion and climate change, alterations of the hydrological cycle and vegetation composition shifts need to be considered explicitly. Even though the role of erosion and vegetation dynamic processes in accelerating land degradation is well recognized, few studies have focused on these processes and their interactions.

Landscape-level information on land health at nested spatial scales is needed to bridge this gap. The Land Degradation Surveillance Framework (LDSF) is an example of a method that has been applied in a number of projects in the global tropics to provide more rigorous, science-based, assessments of land degradation risk and status. In this study, we use field datasets collected using the LDSF to provide spatially explicit assessments of land health on Mount Elgon in Uganda and Campo Verde in Peru, respectively. Each LDSF site covers a landscape that is 10x10 km in size (100 km^2), where 160 sampling plots are established, each representing an area of 0.1 ha. We conduct comparative assessments of land health indicators, including tree biodiversity, soil health and land degradation (soil erosion and compaction) in these landscapes.

Finally, this information is used in conjunction with information on the potential natural vegetation of the study areas to explore options for restoration of forest landscapes through the use of native tree species.

4.1. Land cover and land use

The sites on Mt Elgon were mostly cropland (between 89 and 91% of each site), followed by grasslands or pastures (5.6 to 9.8%), while Campo Verde was predominantly grassland (56.3%), followed by croplands (16.5%) and thickets (15.8%) (Table 2). None of the surveyed plots were in forests in the Mt Elgon study area, but rather open stands of trees (woodlands) or

Table 2. Vegetation structure classes for the three sites, based on the LDSF field surveys.

Vegetation structure	Site		
	Mbale	Bumagabula	Campo Verde
	---------- ha ----------		
Forest			443
Woodland	130		
Bushland	60	75	127
Shrubland			190
Thicket			1,582
Wooded grassland	60		380
Pasture (grassland)	560	977	5,632
Cropland	9,130	8,947	1,646
Other	60	1	

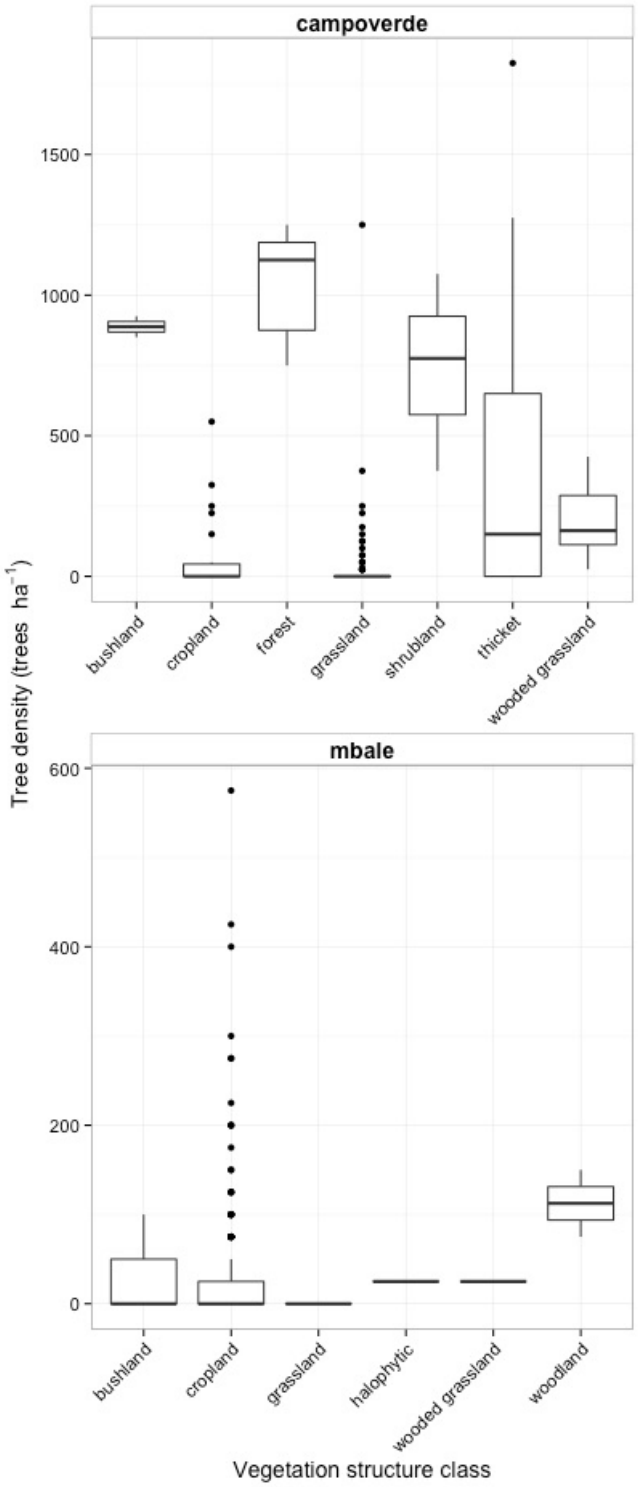


Figure 10. Tree density (trees ha^{-1}) by sampling cluster in the two study areas. Note that units on y-axis are different in the two plots.

bushland (a mix of trees and shrubs with a canopy cover of 40% or more), the latter found mostly within the forest reserve and along rivers and streams. The surveyed sections of the forest reserve on Mt Elgon were heavily disturbed with low tree densities.

The number of trees in Mbale (Bumagabula and Mbale combined) is high, considering that this landscape is predominantly agricultural, as shown in the species dominance graph for this area in Figure 15. The most common species in Mbale is *Markhamia lutea* (nsambya or lusambya in Luganda), which is native to Ethiopia, Kenya, Tanzania and Uganda and common particularly in the lake basins and highland areas. The tree is drought resistant but sensitive to water-logging, and often promoted for soil conservation. *Markhamia lutea* has a number of different uses, including as a source of firewood and charcoal, shade in and around homesteads (Figure 11), for construction of furniture and timber, and the leaves have medicinal value. Other native tree species that are common in Mbale, include *Cordia africana, Vernonia auriculifera* and *Croton macrostachys*.

The species inventory for Campo Verde has a number of unidentified species, but if we consider the species that were identified during the survey, Cetico (*Cecropia spp*) is the most common genus in the area (Figure 15). This genus consists of several pioneer tree species that are common in the Neotropics. Other common species include ocuera blanca (*Vernonia sp*), shimbillo (*Inga thibaudiana*) and pichirina (*Vismia spp*). Shimbillo is a multi-purpose tree that is nitrogen-fixing, fast growing and easily adapts to agroforestry systems. Some varieties can withstand prolonged flooding, but are sometimes considered a nuisance by farmers. Removal of shimbillo is not easy as it regenerates with ease. Hence, Inga species have great potential for use in the restoration of degraded lands.

Cecropia spp species are also major pioneer species, and in some cases invasive as some of the species in this genus are able to self-pollinate. They are often the first pioneer species to occupy former forest areas, being aggressive and growing rapidly, including in former forest areas cleared for pasture such as in Campo Verde.

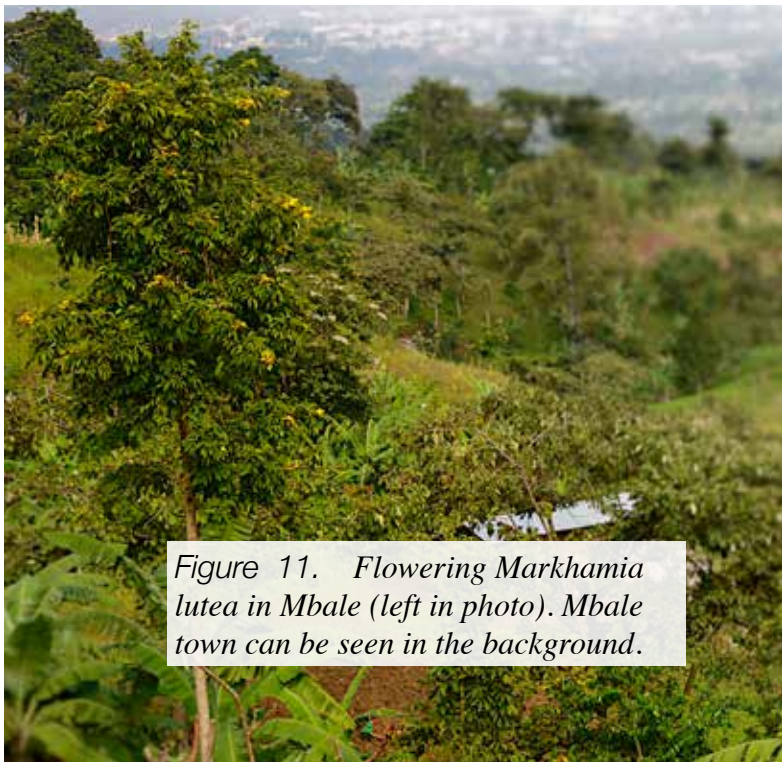


Figure 11. Flowering Markhamia lutea in Mbale (left in photo). Mbale town can be seen in the background.

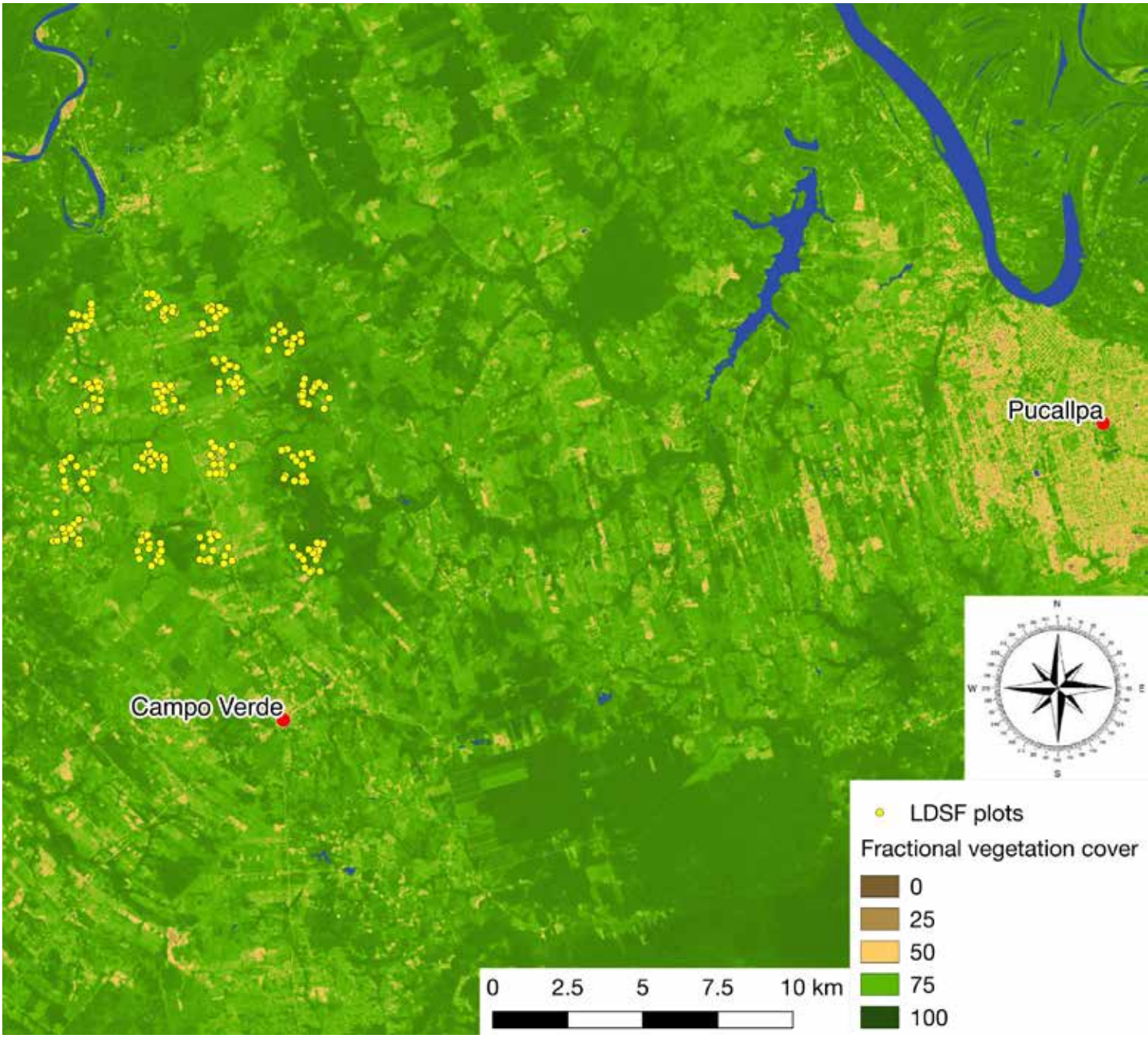


Figure 12. Fractional vegetation cover in Campo Verde (Pucallpa), based on the Soil Adjusted Total Vegetation Index (SATVI) (Marsett et al., 2006; Qi et al., 2002).

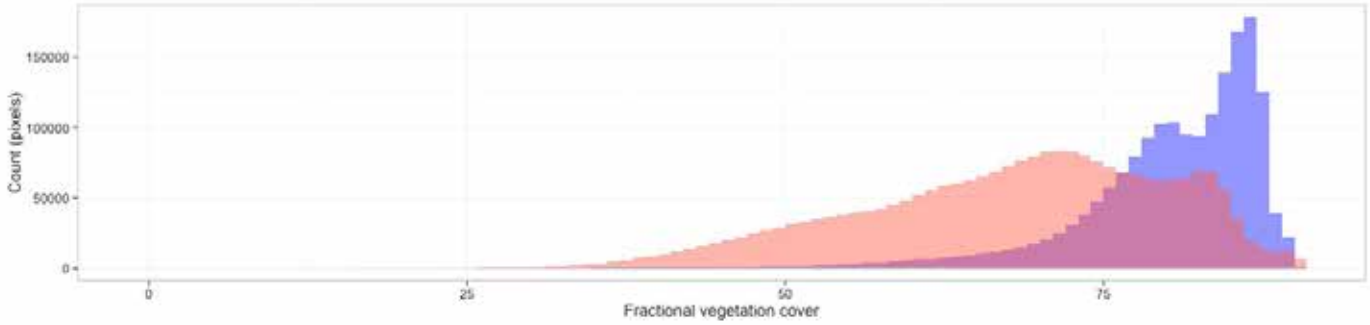


Figure 13. Distribution of fractional vegetation cover in the two study landscapes (blue=Campo Verde; red=Mbale), based on Landsat 8 imagery from 2013 and 2014 (see maps in Figure 12 and Figure 14). Note that these distributions represent the broader landscapes, not just the LDSF sites. Fractional vegetation cover values above 85 represent forest, while values below 50 represent low vegetation cover. The SATVI index used is also sensitive to senescent vegetation, such as dry grasslands.

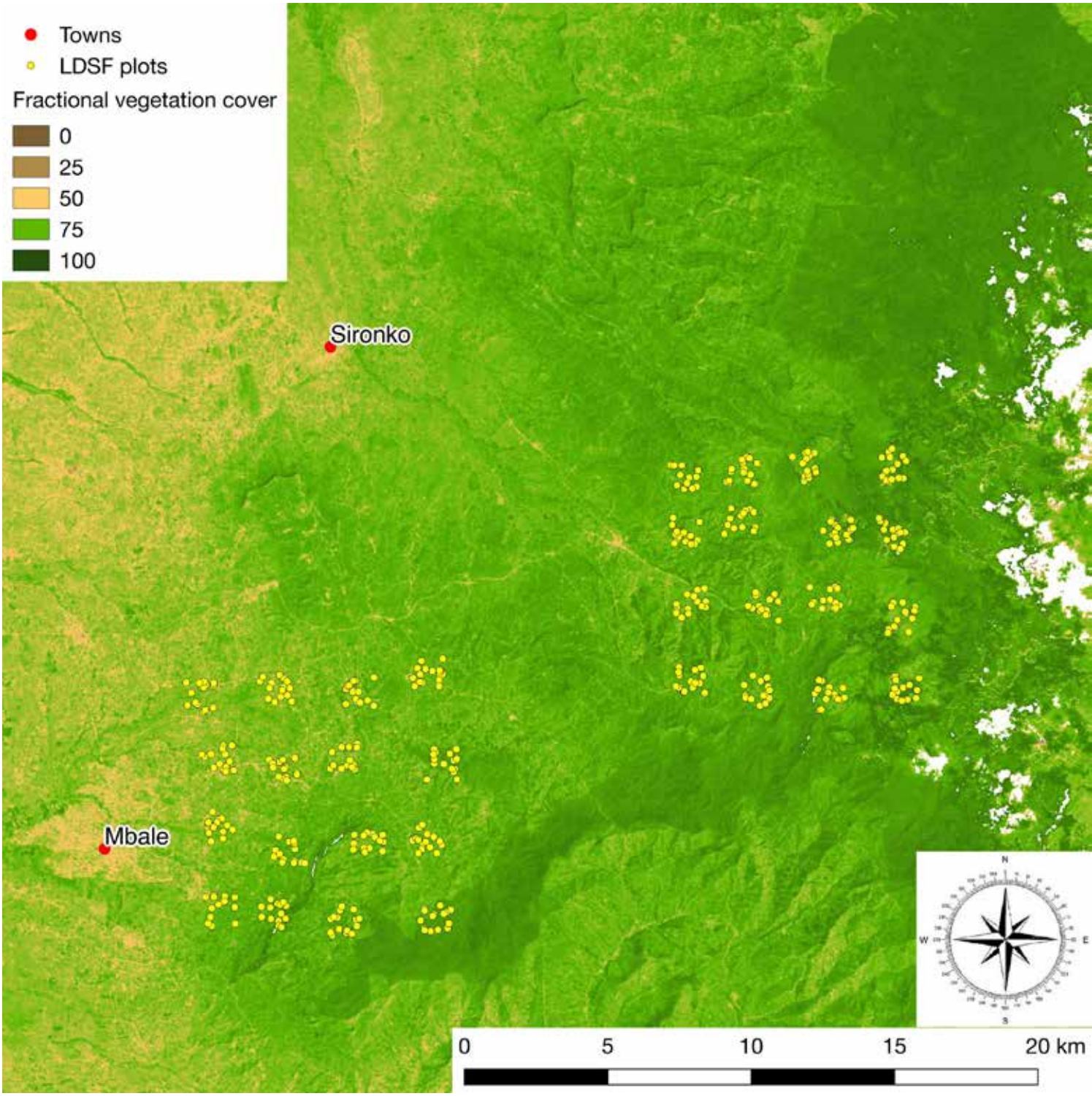


Figure 14. Fractional vegetation cover in the Mt Elgon study area, based on the Soil Adjusted Total Vegetation Index (SATVI) (Marsett et al., 2006; Qi et al., 2002).

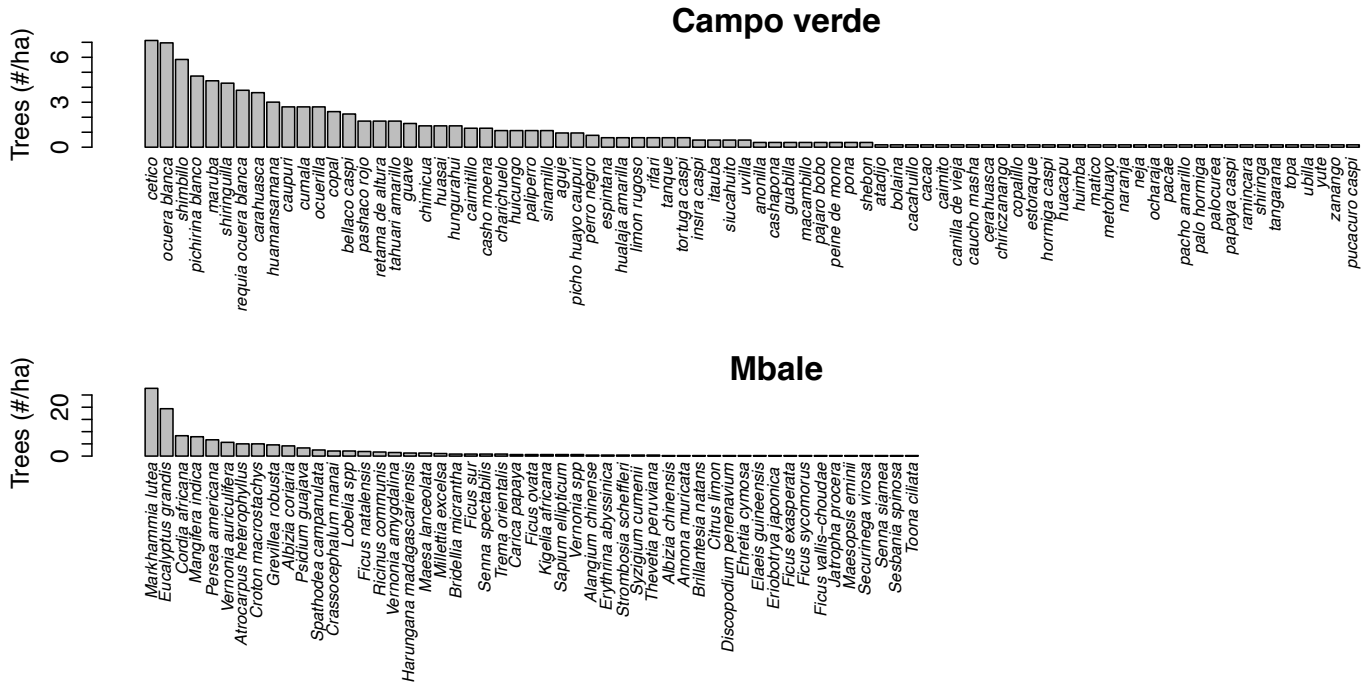


Figure 15. Species dominance graphs for the two study landscapes.

We use species richness as a simple way to denote the number of different species in the study sites. This is a simple measure of species number relations, and we calculate richness here for the different vegetation structure classes in the study areas (Figure 16).

Forest ecosystems in the Amazon typically have high species richness, with some studies reporting about 300 species in a single hectare of undisturbed forest. Central African forests can have comparable species richness to the Amazon. Studies conducted on the slopes of Mount Elgon have identified at least 400 different tree species in this ecosystem.

Overall, the highest species richness is found in forest and bushland systems, although the diversity of trees in these systems is a fraction of what one would expect in an undisturbed forest system. Land degradation in terms of loss of biodiversity is therefore significant even in remaining forests in Campo Verde.

Grasslands and croplands have very low species richness in terms of tree species. What is perhaps surprising is the low level of species richness in thickets, or fallow areas, but this may be due to a dominance of pioneer species in these areas.

Interestingly, species richness in croplands in Mbale is higher than in shrublands and thickets within this site. This may be attributed to the long tradition of agroforestry in this area, which has resulted in a relatively rich agroforestry system. However, it is worth noting that many of these species are exotic, such as Eucaluptus, mango and avocado trees. Wooded grasslands and woodlands have predominantly *Eucalyptus grandis* and relatively low species richness.

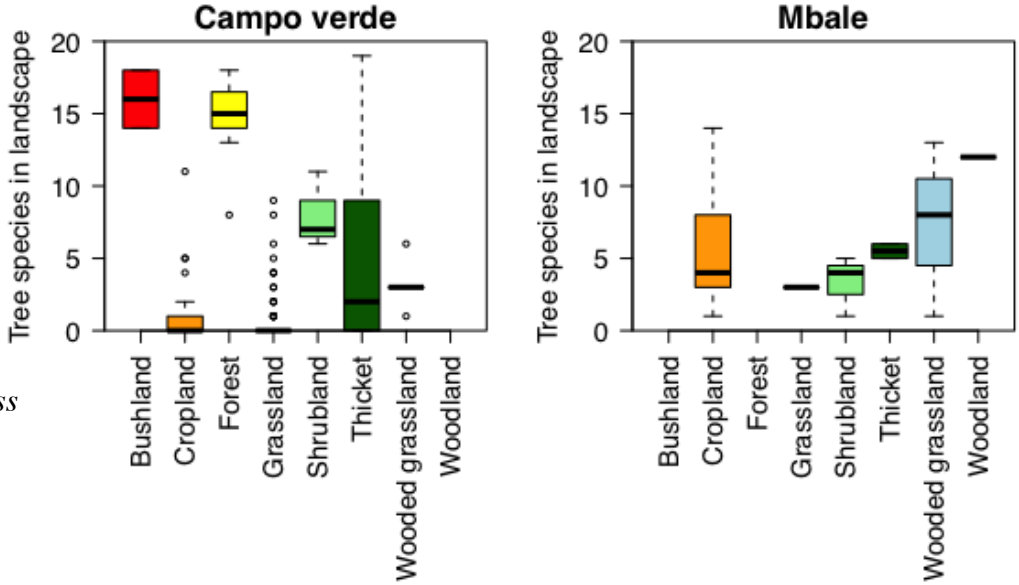


Figure 16. Species richness by vegetation structure class in Campo Verde and Mbale.

4.2. Land degradation

4.2.1. Soil erosion

Soil erosion by wind and water depends on several interrelated factors such as climate, moisture availability, soil properties, topography, land cover and management. We model and map the prevalence of soil erosion in the two study landscapes using a combination of LDSF field survey data, machine learning methods and remote sensing. These approaches allow us to detect soil erosion directly, and hence quantify erosion risk in a spatially explicit way through the identification of soil erosion "hotspots".

Since the predictions are performed based on models developed using systematically collected LDSF field observations of different forms of soil erosion and calibrated satellite imagery, comparative studies can be made between different the study areas. Also, these methods can be applied in the estimation of past erosion, and hence changes over time can be assessed. Where soil erosion is severe, irreversible denudation can occur due to self-sustained interactions between soil erosion, soil degradation and nutrient loss, and loss of vegetation cover, making restoration challenging.

Understanding interactions between land degradation processes such as soil erosion and vegetation cover and soil carbon storage is of critical importance for management of ecosystem health, and for determining restoration potential.

Soil erosion was observed in 22% of the plots in Campo Verde and 83% of the plots in the Mount Elgon study area. About 54% of croplands in Campo Verde were eroded, while almost 90% of croplands in Mbale showed visible signs of erosion. No erosion was observed in forest, while 4% of the sampled plots in thickets were eroded. In the grasslands, 20% of the plots in Campo Verde were eroded, while the frequency of eroded grassland plots in Mbale was 26%.

We predicted erosion prevalence using Landsat 8 data with good accuracy and precision overall, when testing the model on an independent dataset (Figure 17)., and proceeded to map soil erosion prevalence based on Landsat 8 for the two study landscapes.

Figure 19 shows predicted erosion prevalence in the Campo Verde study area at 30m resolution. Erosion in this area is low to moderate (Figure 20). Some hot-spots can be seen within the Campo Verde site, as well as along riverbanks north of the site. As expected, forested areas show no occurrences of soil erosion in the predicted maps.

Soil erosion patterns in the Mbale area are complex (Figure 21), reflecting the complex mosaic of smallholder farms in this area. The map shows higher erosion in cultivated areas in both study landscapes, showing the importance of cultivation as a driver of land degradation in these landscapes.

The higher overall risk of erosion in Mbale is particularly evident in Figure 20, highlighting the need

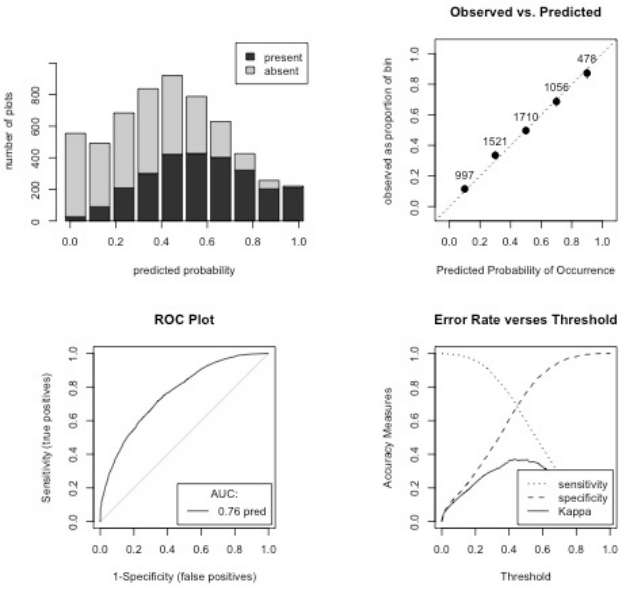


Figure 17. Validation prediction results for erosion, based on Landsat 8.

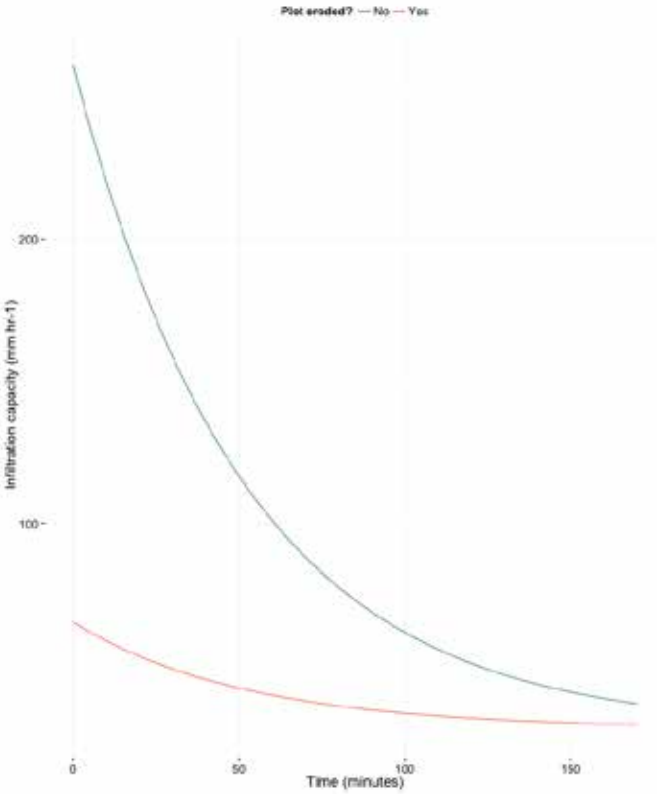


Figure 18. Infiltration curves for plots with observed soil erosion and plots without visible patterns of erosion in Mbale.

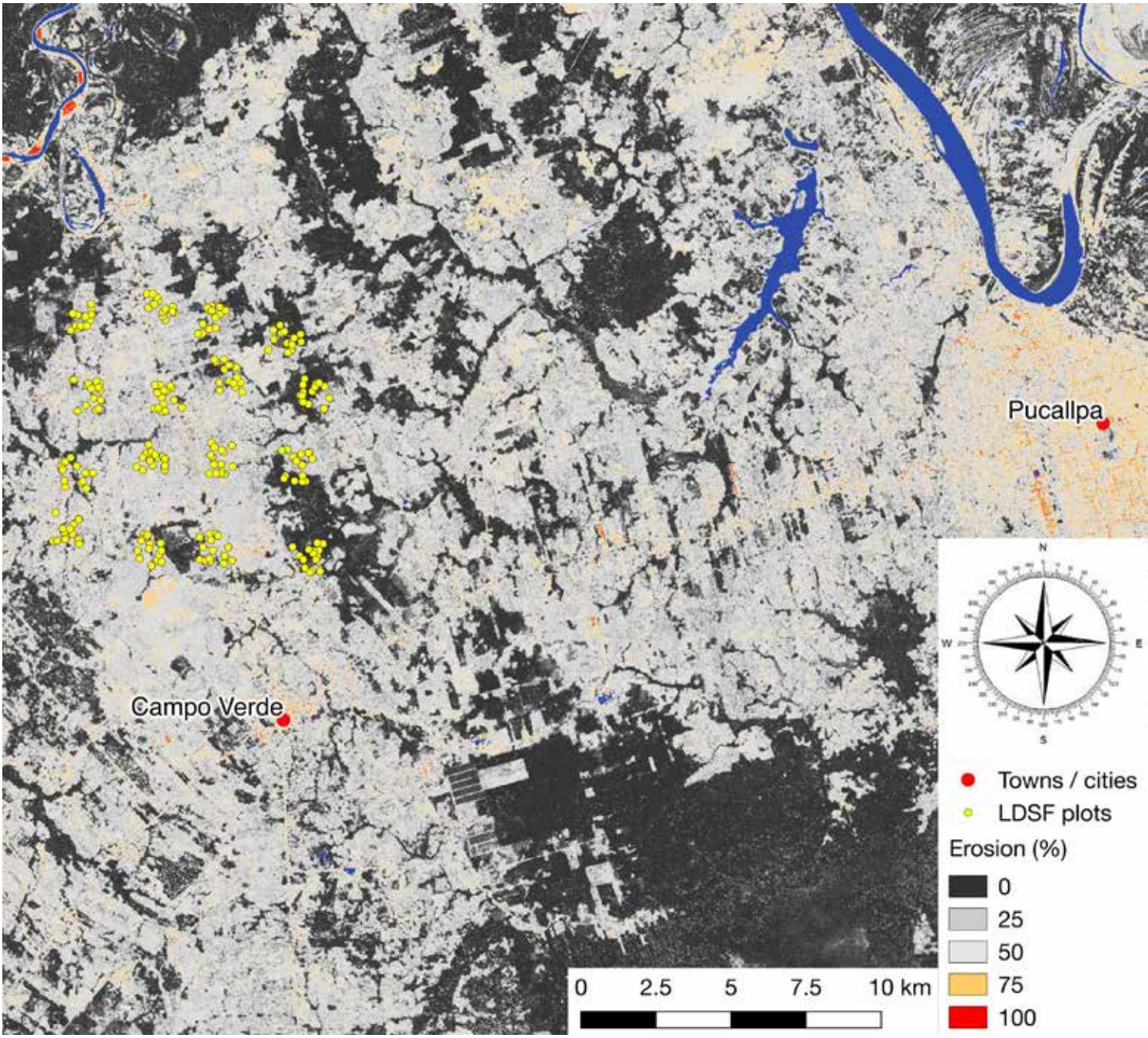


Figure 19. Map of predicted erosion prevalence (%) in the Campo Verde study area, based on Landsat 8 from March 2013.

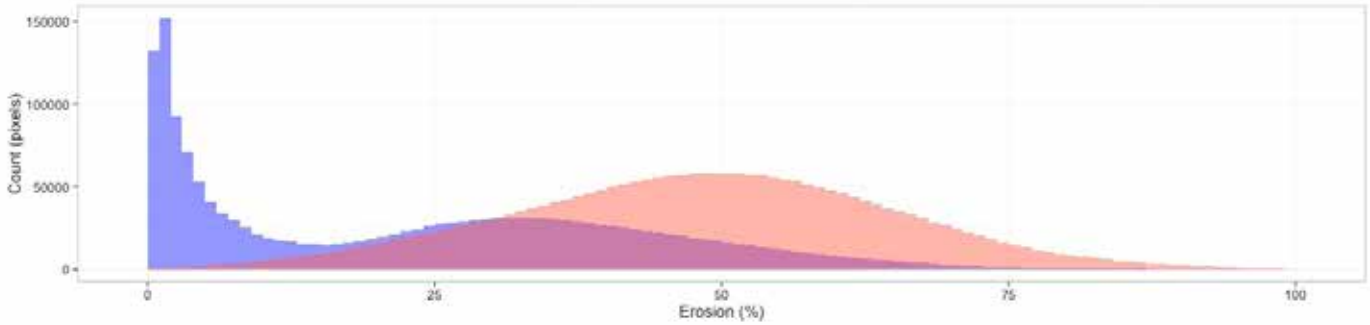


Figure 20. Distribution of soil erosion prevalence in the two study landscapes (blue=Campo Verde; red=M-bale), based on Landsat 8 imagery from 2013 and 2014 (see maps in Figure 19 and Figure 20). Note that these distributions represent the broader landscapes, not just the LDSF sites. Erosion values higher than 50% are considered high, while values higher than 75% indicate severe erosion.

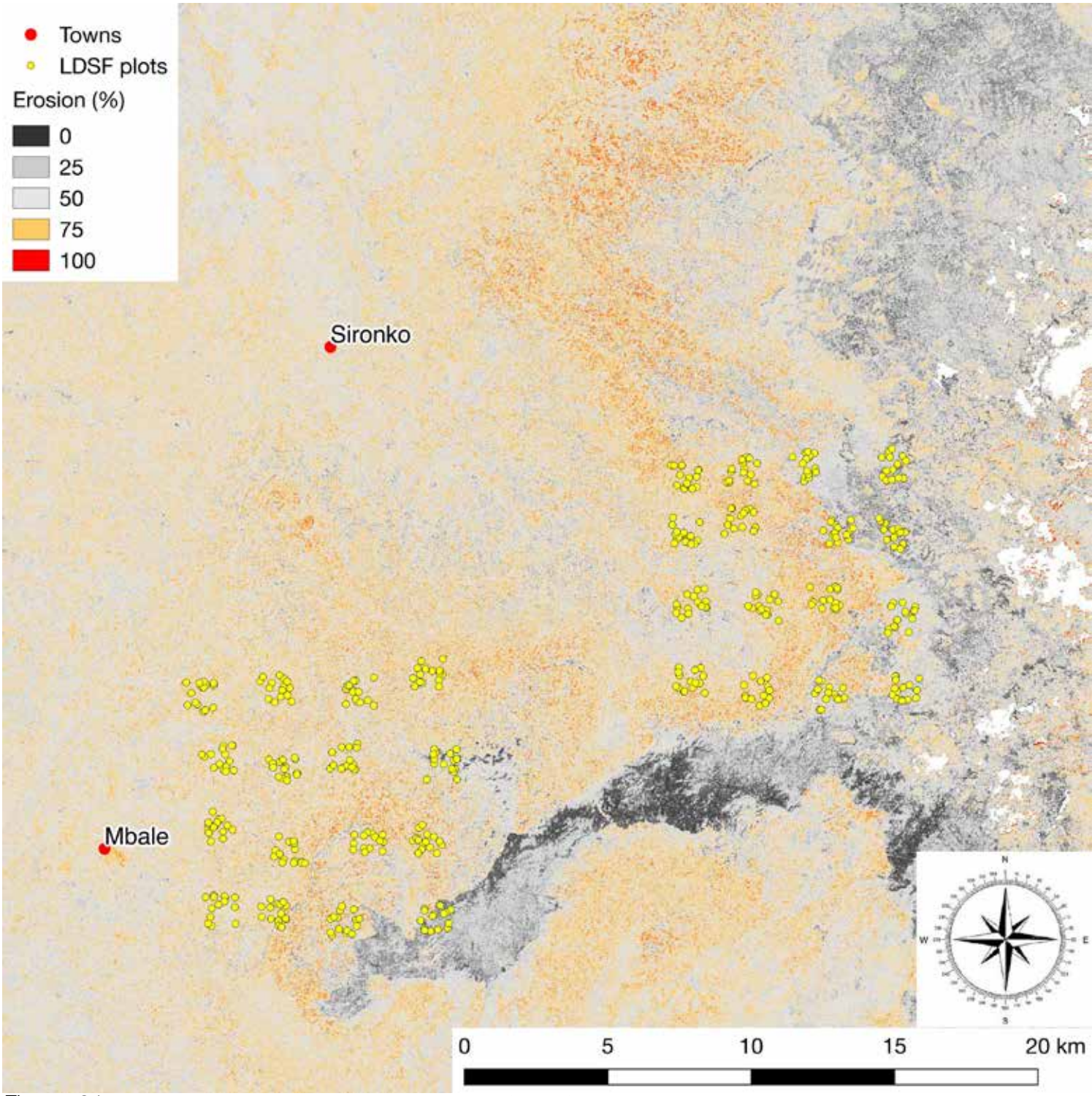


Figure 21. Map of predicted erosion prevalence (%) in the Mount Elgon study area, based on Landsat 8 from February 2014.

for soil and water conservation measures in this area to control erosion. Initial infiltration rates are much lower in eroded plots in Mbale than in non-eroded plots, while saturated hydraulic conductivity is similar (Figure 18). This is particularly important when considering the steep topography of the Mbale study area, as the low initial infiltration rates are likely to result in overland flow during heavy rainfall. In the next section of the report we explore interactions between root-depth restrictions and infiltration capacity further.

4.2.2. Root-depth restrictions

Root-depth restrictions can occur due to soil compaction, which is where soil particles are pressed together, reducing the pore space between them. Compaction can occur as a result of a number of different human-induced factors such as (i) machinery, (ii) tillage, (iii) trampling by animals and even from the force of raindrops when the soil is exposed. The latter is particularly a problem in soils that are low in soil organic matter, and tends to occur after deforestation, for example.

When soils become heavily compacted, they have fewer large pores and water infiltration tends to become slower. This occurs because the larger pores in the soil are most effective in moving water through the soil when it is saturated. Finally, roots have to exert more energy or force to penetrate compacted soil, which tends to inhibit plant growth.

Effective rooting depth for agricultural crops such as wheat and maize are commonly reported to be between 60 and 180 cm, when these crops are grown in unrestricted conditions. Root-depth restrictions in the upper 50 cm of the soil profile therefore present challenges for agricultural management by potentially limiting plant growth and soil infiltration capacity both in agricultural and forest systems.

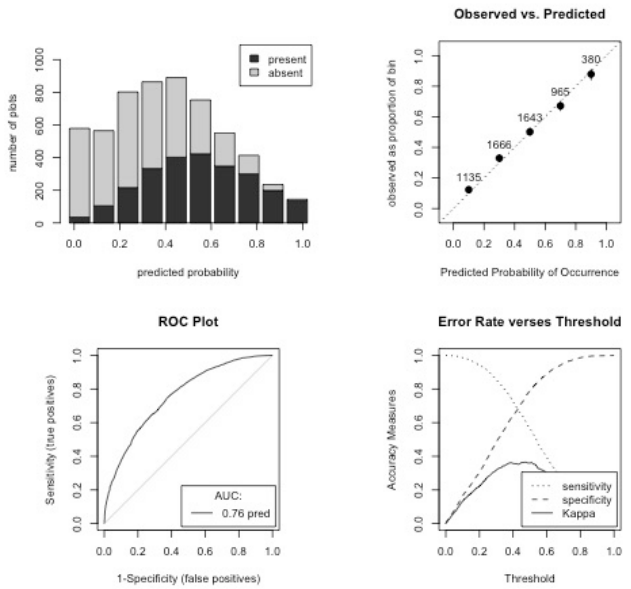


Figure 23. Validation prediction results for root-depth predictions, based on Landsat 8.

Reduced infiltration capacity and increased overland flow can in turn lead to accelerated soil erosion, so from a management perspective, identifying areas that have a high prevalence of root-depth restrictions is important in order to target interventions that reduce soil compaction. In agricultural areas such practices may include reduced- or no-till practices, or the introduction of agroforestry tree species that help increase SOC and break up restrictive layers.

The effect of root-depth restrictions on infiltration capacity is evident in the Mbale study area (Figure 22) where initial infiltration capacity is about two times higher in areas that do not have root-depth restrictions and saturated infiltration capacity is almost four times higher. This has important implications in this area as restrictions occur in about 26% of the two sites combined.

Water infiltration rates are very low in Campo Verde, even if no compaction was recorded in the field. This area is therefore prone to flooding, even with moderate rainfall and is also likely to be sensitive to soil physical degradation even at low levels of disturbance.

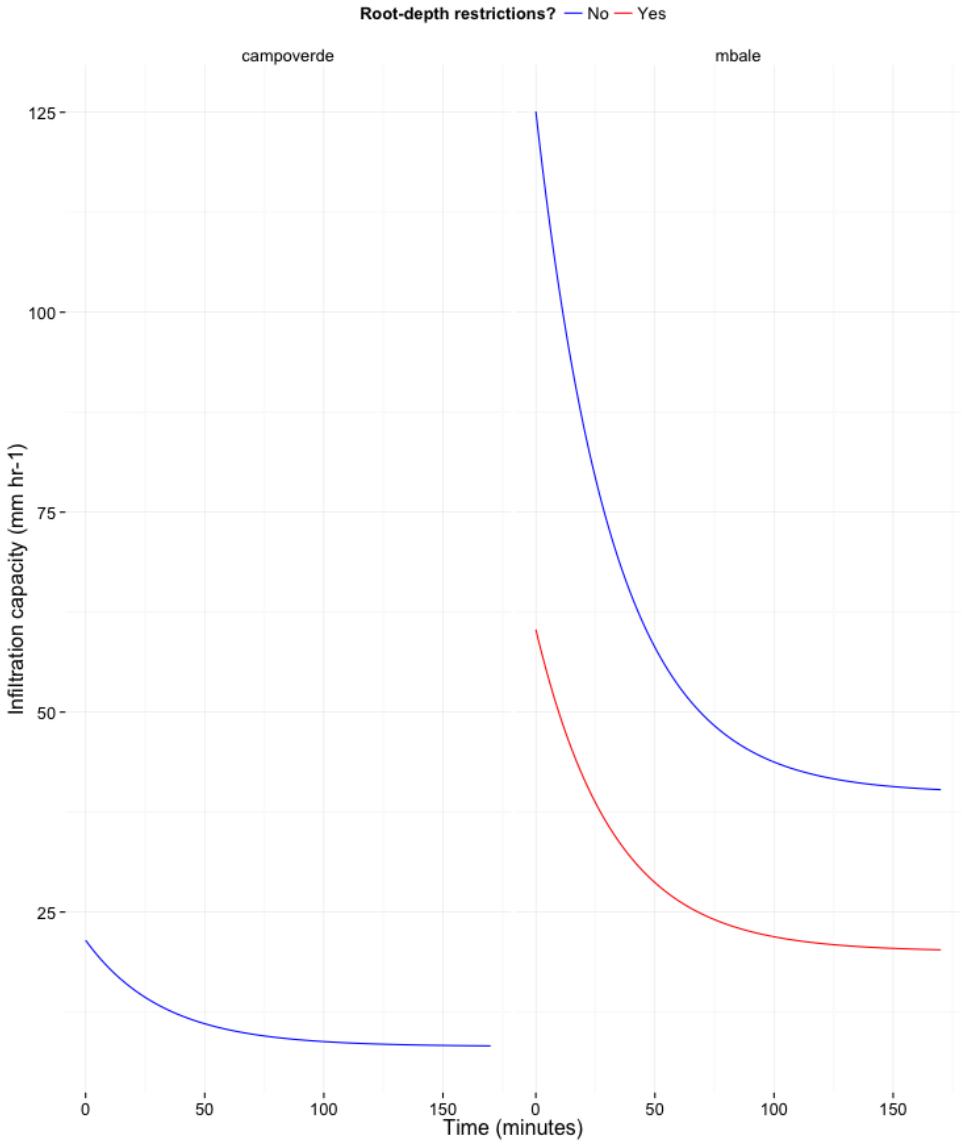


Figure 22. Infiltration curves for plots with and without root-depth restrictions in Mbale and Campo Verde.

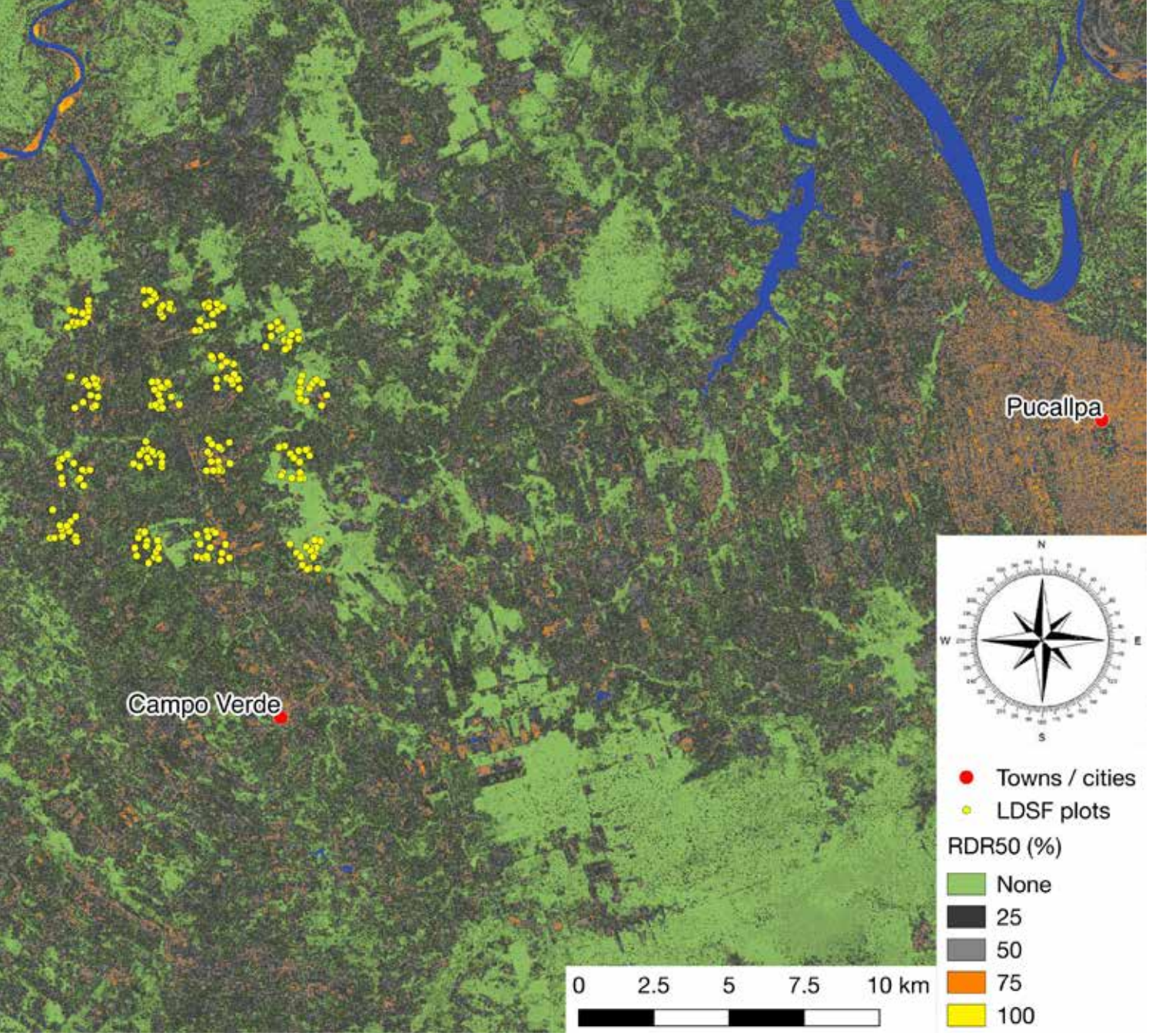


Figure 24. Map showing the predicted probabilities of root-depth restrictions in the Campo Verde study area, based on Landsat 8 from March 2013.

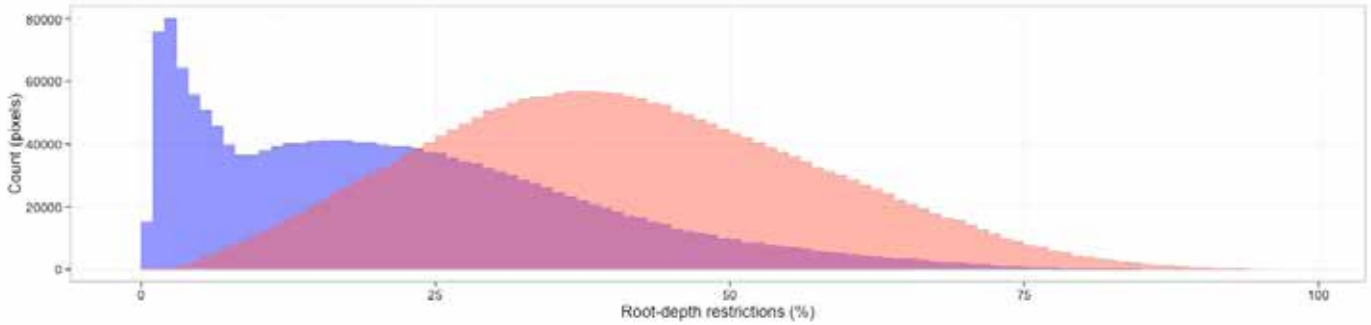


Figure 25. Distribution of root-depth restrictions in the two study landscapes (blue=Campo Verde; red=M-bale), based on Landsat 8 imagery from 2013 and 2014 (see maps in Figure 24 and Figure 26). Note that these distributions represent the broader landscapes, not just the LDSF sites. Root-depth restriction values higher than 50% are considered high, while values higher than 75% indicate severe restrictions.

The prediction performance for root-depth restrictions (Figure 23) is similar to that for erosion shown earlier and satisfactory for predictive purposes. We therefore proceeded to predict and map root-depth restrictions in the two study areas in Figure 24 and Figure 26.

Forested ecosystems have virtually no occurrences of compaction, while some of the pastures, particularly along paths, show higher prevalence of compaction in Campo Verde. Also, areas that have higher prevalence of soil erosion show higher occurrences of root-depth restrictions in both study areas. In the Mbale study area, root-depth restrictions are particularly common in croplands and along ridges with shallow soils (Figure 26). Considering the strong impact on infiltration capacity, reducing such restrictions is critical for restoration efforts in these areas.

Root-depth restrictions also need to be considered when determining which species that are suitable for restoration, particularly when erosion prevalence is also high as these processes are likely to reinforce each-other.

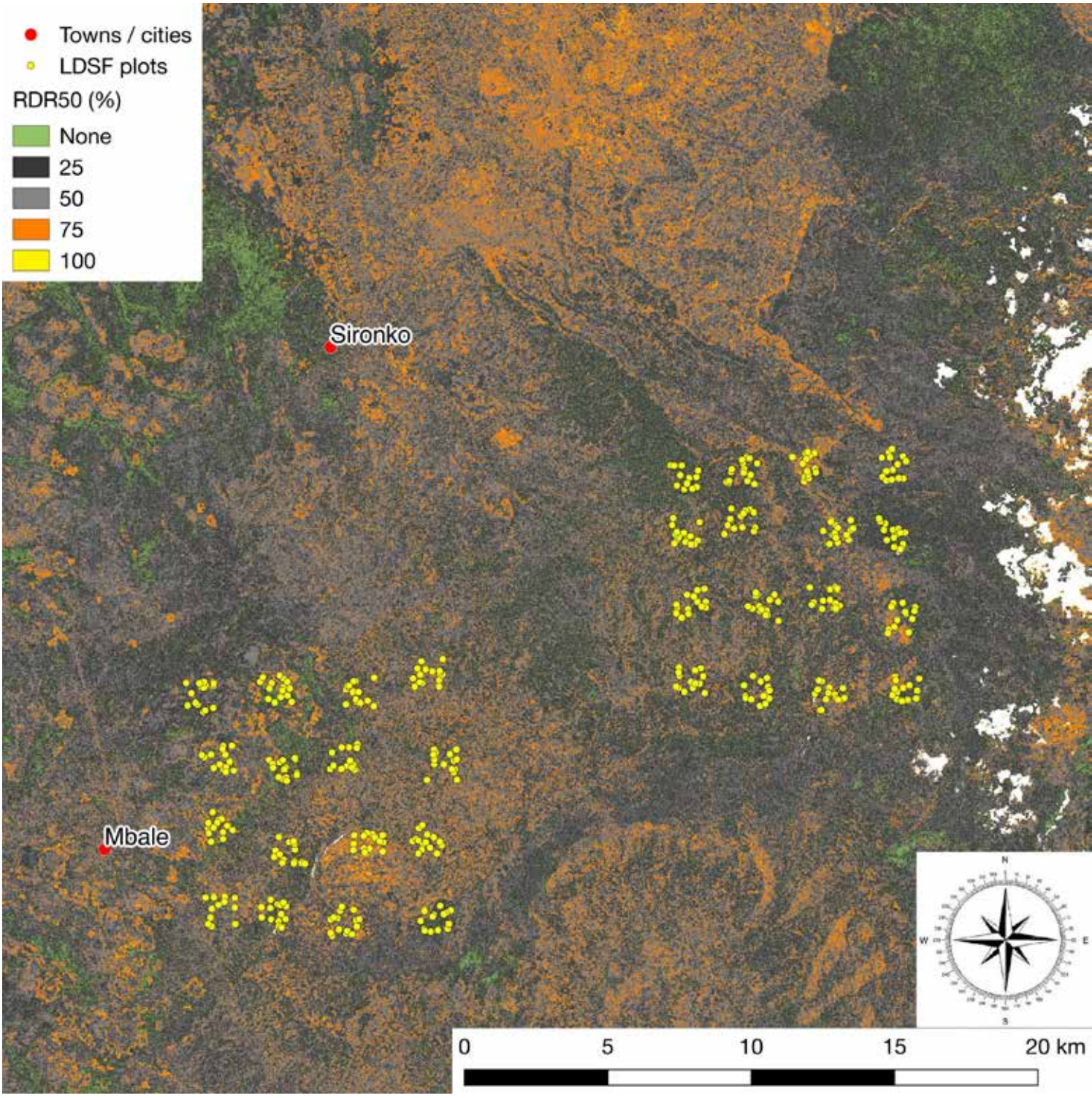


Figure 26. Map showing the predicted probabilities of root-depth restrictions in the Mount Elgon study area, based on Landsat 8 from February 2014.

4.3. Soil health

Soil health is an integrative property that reflects both the productive capacity of soil (e.g. for agricultural production) and the ability of the soil to provide other ecosystem services. We build our use of soil health on the definition proposed by Kibblewhite et al. (2008): "A healthy (agricultural) soil is one that is capable of supporting the production of food and fibre, to a level and with a quality sufficient to meet human requirements, together with continued delivery of other ecosystem services that are essential for maintenance of the quality of life for humans and the conservation of biodiversity".

Factors such as soil texture (clay, silt and sand content) influence soil health by creating constraint en-

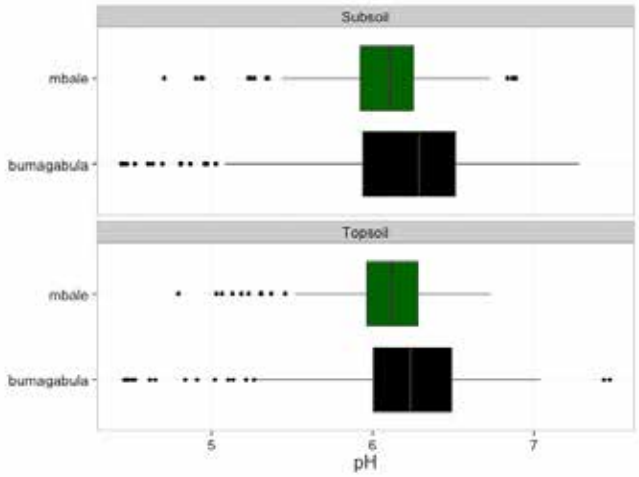


Figure 28. Soil pH in topsoil (0-20 cm) and subsoil (20-50 cm) for Mbale and Bumagabula.

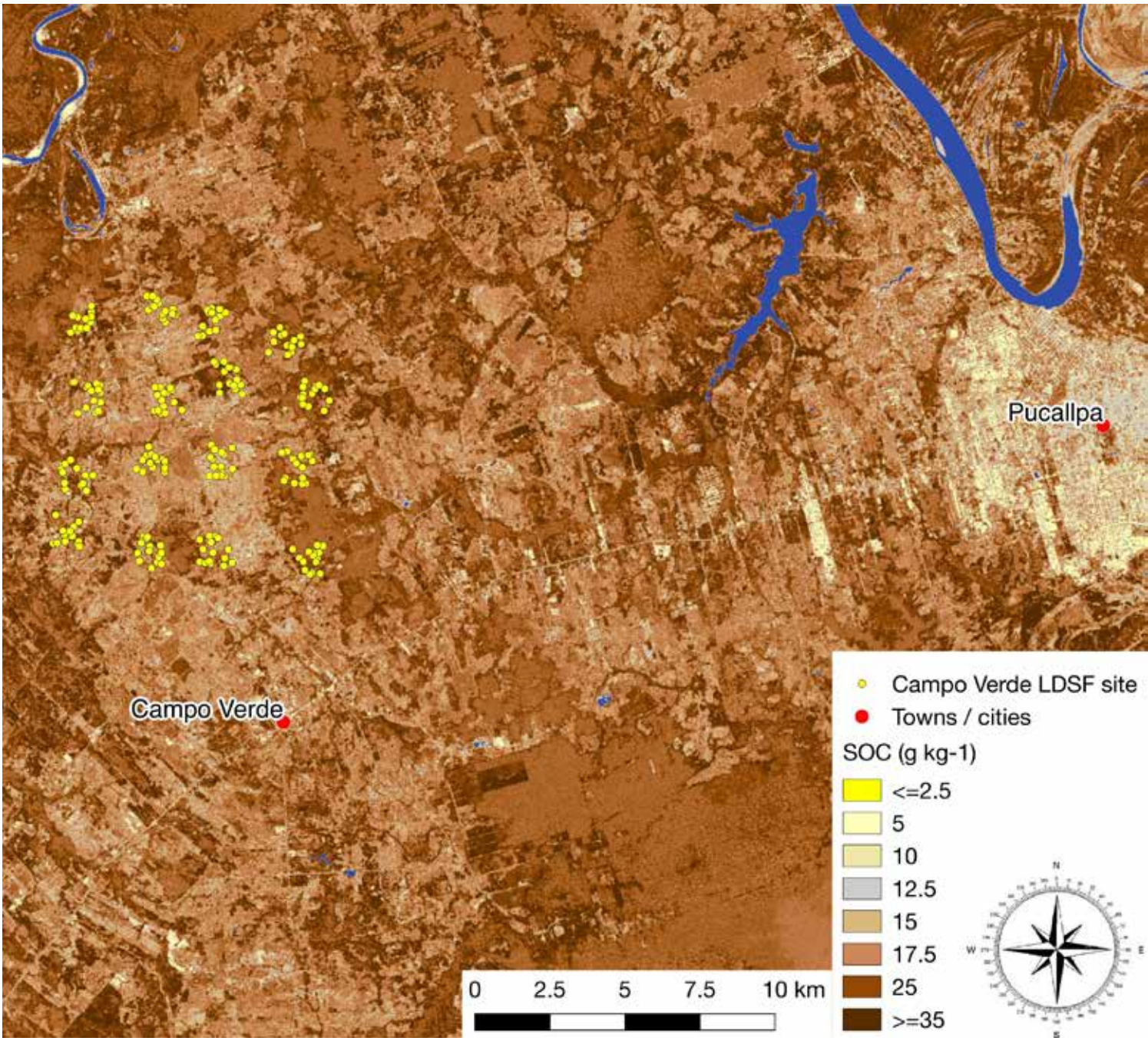


Figure 27. Map of SOC in the Campo Verde study area, based on Landsat 8 from March 2013.

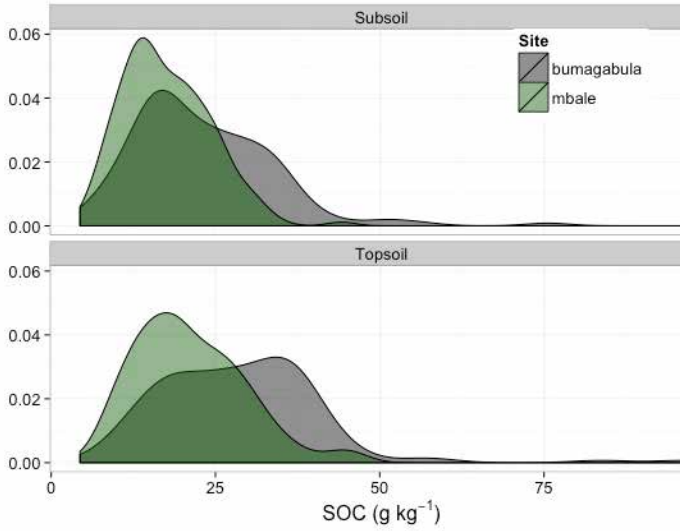


Figure 30. Distributions of laboratory measured topsoil (0-20 cm depth) and subsoil (20-50 cm depth) SOC in the Mbale and Bumagabula LDSF sites, respectively.

velopes for variable or dynamic soil properties such as SOC, pH and bulk density, as well as assemblages of soil organisms. These dynamic properties are often strongly influenced by human land use and hence being able to characterize and map their spatial (and temporal) distribution is critical for the development of land restoration options.

4.3.1. Soil organic carbon (SOC)

Common approaches for C sequestration recognized by international agreements on climate change such as the United Nations Framework Convention on Climate Change (UNFCCC) include afforestation and reforestation, and hence restoration forest landscapes can be an important strategy for C sequestration. Also, management of croplands, grasslands and forests have later been recognized as important for C sequestration. Agroforestry systems have received particular attention in this regard, as they represent a significant potential for increased C sequestration in croplands through the introduction of trees into farming systems, as well as a key component of forest landscape restoration efforts.

The actual potential for C sequestration in a given system depends on its ecological production potential as well as management, specific land use types and species composition. Drylands generally have a lower ecological potential for C sequestration than humid ecosystems, for example.

Sequestration of C in soil is of particular importance as it has the potential to provide a more permanent sink for C than aboveground biomass. However, most

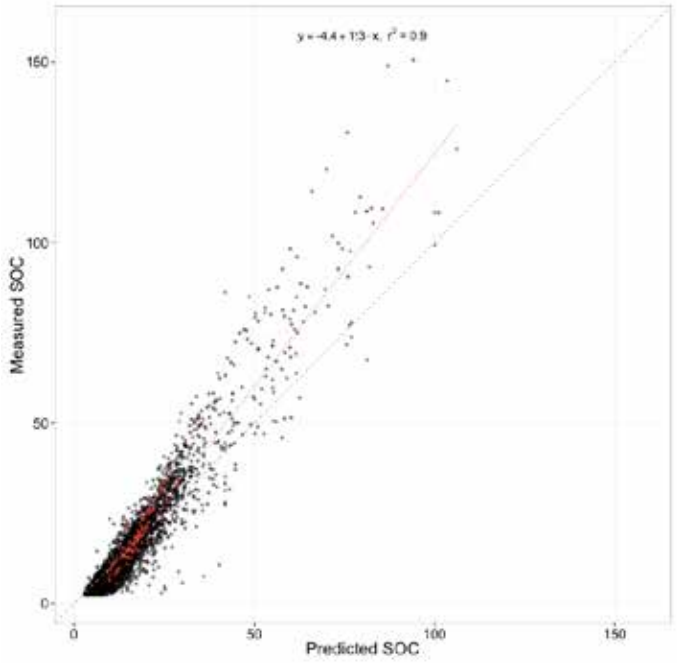


Figure 29. Prediction model performance for the prediction of SOC from Landsat 8 remote sensing data, based on an independent test dataset. Mbale and Bumagabula are shown as red points.

methods for assessing SOC concentrations, and particularly stocks in landscapes are not well suited for understanding spatial variations in SOC sequestration potential. Further, inherent properties of soils such as texture determine their potential for C sequestration, which means that understanding the interactions between SOC and texture is critical for determining the actual potential of a given area for C sequestration.

Given the heterogeneity of landscapes, spatial patterns of SOC concentrations and stocks tend to be complex. Spatial information on the distribution of SOC and other soil properties therefore need to be made at relevant spatial scales (e.g. at the farm level). Most models and estimates of C sequestration potential only allow for coarse-scale assessments of SOC sequestration potential and are not able to predict the possible fate of carbon due to land-use change at scales relevant to management interventions. Alternative approaches have been suggested that enable these types of assessments to be conducted using cumulative soil mass (CM) measurements and direct calculations of SOC stocks (Winowiecki et al., 2015), without the use of bulk density, which is prone to error, limiting its application for landscape level assessments of SOC stocks.

At the time that this report was written, soil mid-infrared (MIR) measurements were available for the Mt Elgon sites, but not for Campo Verde due to delays with the international shipping of soil samples from Peru to Kenya. Hence, the estimated SOC values for

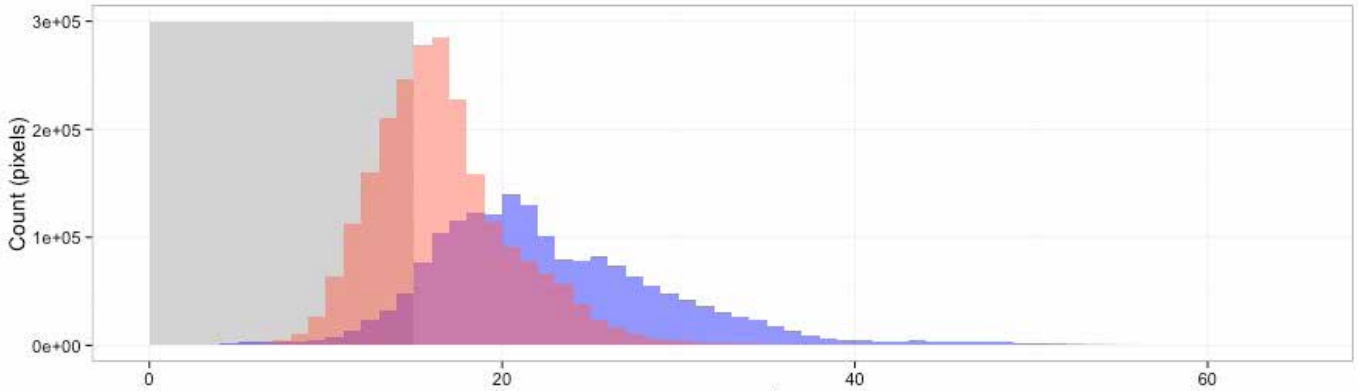


Figure 31. Distribution of SOC in the two study landscapes (blue=Campo Verde; red=Mbale), based on Landsat 8 imagery from 2013 and 2014 (see maps in Figure 27 and Figure 32). Note that these distributions represent the broader landscapes, not just the LDSF sites. SOC values lower than 15 g kg^{-1} (gray area on graph) are considered low.

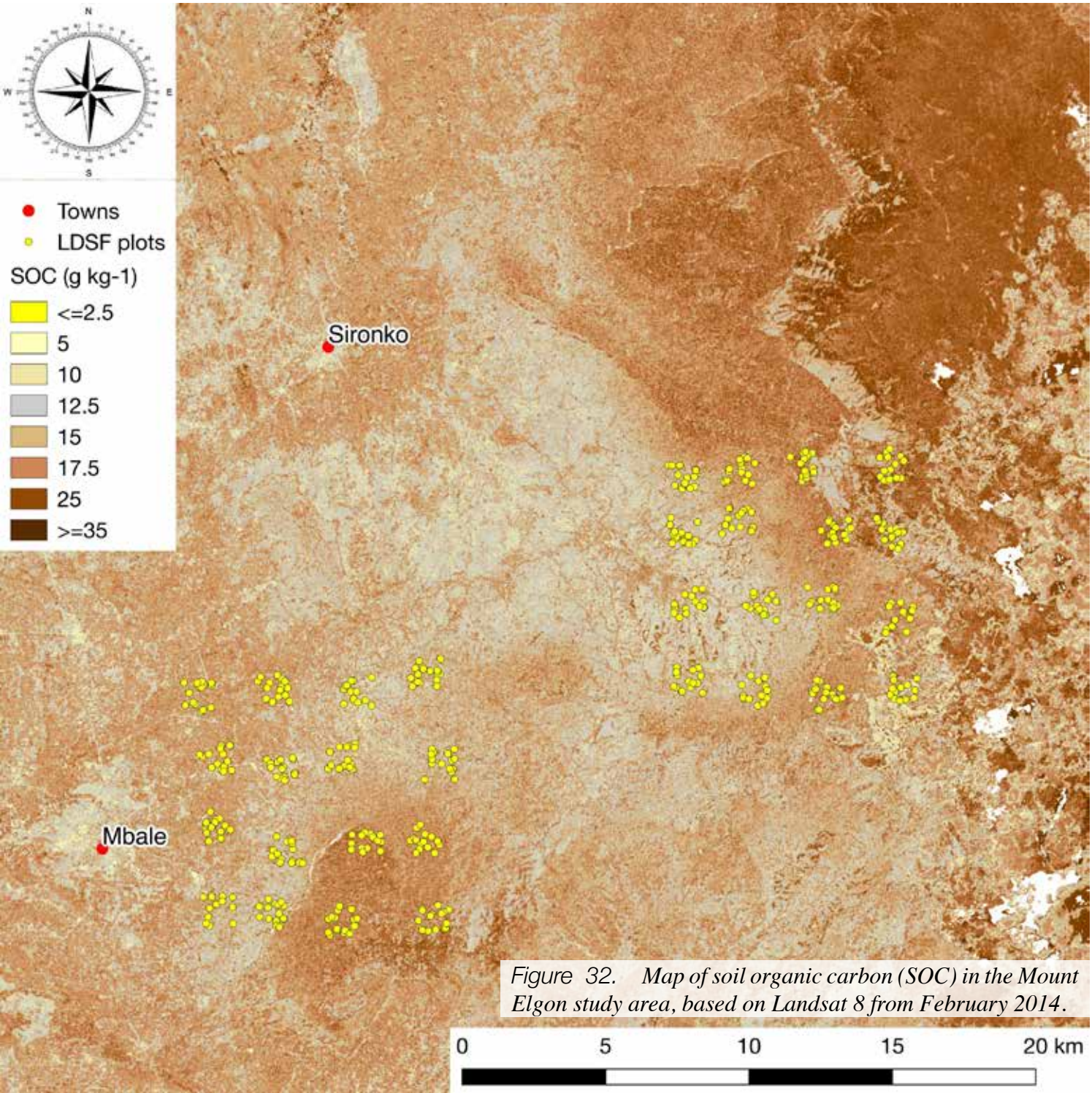


Figure 32. Map of soil organic carbon (SOC) in the Mount Elgon study area, based on Landsat 8 from February 2014.

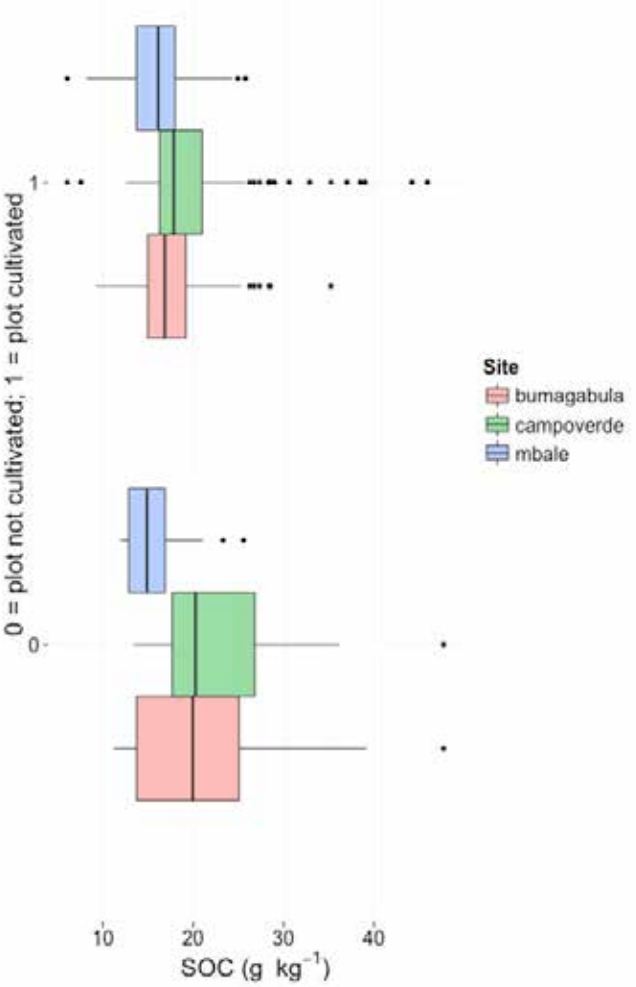


Figure 33. The effect of cultivation on topsoil (0-20 cm) SOC in the three LDSF sites.

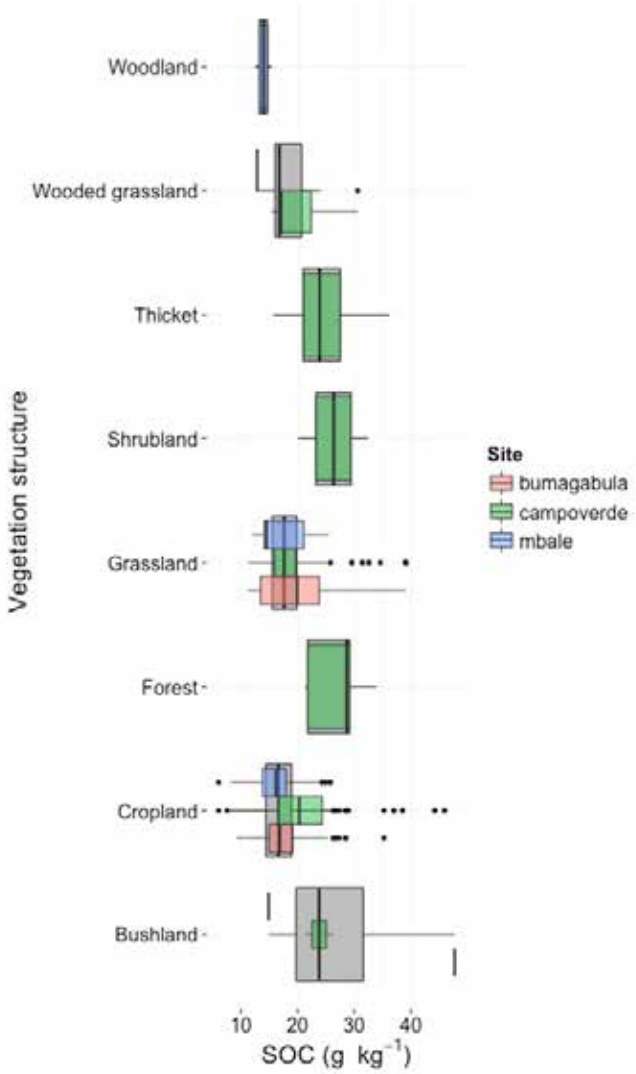


Figure 34. Topsoil (0-20 cm) SOC by vegetation structure in the three LDSF sites. The gray box shows SOC for each vegetation structure class across all sites, while red, green and blue boxes show individual sites.

Campo Verde will need to be validated once these samples have been analyzed.

However, model performance is good overall (using data from 12,600 LDSF sampling plots), as well as for the two sites near Mbale (Figure 29), for the prediction of SOC from Landsat 8. The model applied here for the mapping of SOC explains about 90% (r^2 = 0.90) of the variation in SOC when applying the model to laboratory measured SOC from 4,200 randomly drawn LDSF sites (Figure 29).

The Mbale landscape has lower SOC concentrations overall than Campo Verde (Figure 31), with about 40% of this area having SOC concentrations that are low (<15 g SOC kg^{-1}). About 46% of the surveyed plots in Mbale have SOC values at or below 20 g kg-1, while 24% of the plots in Bumagabula fall below this threshold. The predicted SOC values show lower SOC in cultivated than in uncultivated plots in both Campo Verde and Bumagabula, while SOC is similar for cultivated and uncultivated plots in Mbale. The latter has been cultivated for over 100 years, while the two former LDSF sites have been converted from forest more recently. Interestingly, woodlands in Mbale, which are predominantly Eucalyptus, have very low SOC overall (Figure 34).

Also, SOC varies strongly between vegetation structure classes, as shown in Figure 34. Study plots located within the forest reserve on Mt Elgon have higher SOC than the other plots, indicating significant losses of SOC resulting from forest conversion. There is also a decline in SOC along the altitudinal gradient going down Mount Elgon (east to west on map in Figure 32).

Interactions between SOC and sand content determine the actual potential of a given area for C sequestration. Bumagabula has lower sand content overall than Mbale, which also explains part of the gradient in SOC along the topographic sequence discussed earlier.

5. Site specific restoration options

There is experience of planting of a wide range of timber, fruit and 'service' trees in the Amazon. These include both native and exotics (e.g. *Acacia mangium, Eucalyptus xurograndis, Khaya spp., Gmelina arborea, Pinus caribaea, Tectona grandis* and others.

However, it should be noted that for practical purposes, in any given situation species choice is likely to be constrained by a number of factors, including:

- Farmer preferences, which will reflect market or direct consumption potential of the species and their prior familiarity with it,
- Availability of germplasm from suitable sources,
- Ecological requirements of the species in relation to site and degree of degradation,
- Legal constraints e.g. on use of exotics species in projects categorized as 'restoration'.

The same constraints apply to the Mount Elgon region.

In the Amazon, about 40 species have been used in restoration projects, their ecological requirements, particularly in a restoration context, are not well known. For example, although ICRAF's timber tree planting manual directed at the Peruvian Amazon (Wightman et al. 2005, 'Plantemos Madera') outlines a generic 'technological package', more detailed mini-species guides could only be drawn up for a reduced group of species. The implication is that in practice, restoration activities, in order to minimize risk, should focus on a small group of commercial and service species for which there is sufficient experience (e.g. *C. spruceanum, D. odorata, S. amara, Ochroma spp.* (balsa), *Cecropia spp., I. edulis*, and, on suitable sites or in advanced stages of restoration, the more demanding species such as *G. crinita, S, macrophylla and C. odorata*).

Given the very low infiltration rates in Campo Verde and the flat topography, tree species that tolerate some water-logging and flooding will be important in restoring degraded pastures.

The Mount Elgon study area falls into the category of Afromontane rainforest in terms of the natural potential vegetation of this area. Tree species that characterize this vegetation type include *Chrysophyllum gorungosanum, Cola greenwayi, Cyathea dregei, Cyathea humilis, Cyathea manniana, Cylicomorpha parviflora, Diospyros abyssinica, Entandrophragma excelsum, Ficalhoa laurifolia, Fleroya rubrostipulata, Macaranga capensis, Myrianthus holstii, Ochna holstii, Ocotea usambarensis, Olea capensis, Parinari excelsa, Podocarpus latifolius, Pouteria adolfi-friedericii, Prunus africana, Strombosia scheffleri, Syzygium guineense, Tabernaemontana stapfiana*, and *Xymalos monospora*.

Markhamia lutea, which is the most common tree species found in the Mbale study area is primarily found in evergreen or semi-evergreen forests, as well as in transition zones between evergreen forest and Afromontane rainforest, has been widely adopted as an agroforestry species in this area. It is generally well suited for soil conservation and restoration of degraded soils. *Fleroya rubrostipulata* is a species that is native to this system and suitable for soil restoration in degraded areas, which does not appear in the data from the field surveys, but would be a good candidate for introduction into agroforestry systems in this area.

5.1. Restoration through fallows

Fallows were recorded in less than 1% of the plots surveyed in Mbale and 12% of the surveyed sites in Bumagabula. This mainly reflects the lack of available land in Mbale due to high population densities, which means that virtually no spare land is available, while Bumagabula was more recently converted. The fact that 100% of fallowed plots are in croplands in Mbale (Table 3) confirms this. Also, farmers in this area reported very short fallow periods.

In Campo Verde, there is a wider range of fallow systems in terms of vegetation structure (Table 3), with the majority of fallows (62.5%) classified as thickets consisting of pioneer species. About 17.5% of the areas classified as fallow were forest and hence more long-term fallows. About 25% of the surveyed plots in Campo Verde were in fallow.

Based on these results, fallows will not be a viable option for forest landscape restoration in the Mbale landscape and restoration efforts in this area will need to focus more on agroforestry and the (re)introduction of native species to increase diversity and resilience in these systems.

Future work in Mbale therefore needs to focus on the establishment of nurseries with native species, and processes to effectively disseminate knowledge about the availability and suitability of native species

for the various uses (Figure 37) that farmers are interested in. Also, information about land degradation status and soil health needs to be considered when advising farmers on appropriate agroforestry species for restoration.

For example, in areas that are strongly eroded in Figure 21, assisted restoration will often be necessary using a combination of grasses and trees to stabilize contour ridges or barriers or similar soil conservation measures as trees alone are unlikely to be effective. Trees that provide adequate ground cover should be prioritized for this kind of restoration.

In cases where SOC is low (e.g. SOC < 15 g kg^{-1}) in Figure 32, total nitrogen (N) is also likely to be low and tree species that are nitrogen fixing can be used to restore soils. The planting of trees in such areas can also be combined with conservation agriculture practices such as reduced- or no-tillage.

In Campo Verde, fallows may well be an effective way to restore degraded areas, but further testing of the ecological requirements of candidate species will be needed. Also, increasing the tree diversity in fallow systems such as thickets should be a priority in this landscape.

The main constraint from a biophysical point of view in this area is drainage requirements of different trees (i.e. sensitivity to water-logging). Hence, native species that can tolerate some water-logging should be explored for natural and assisted restoration.

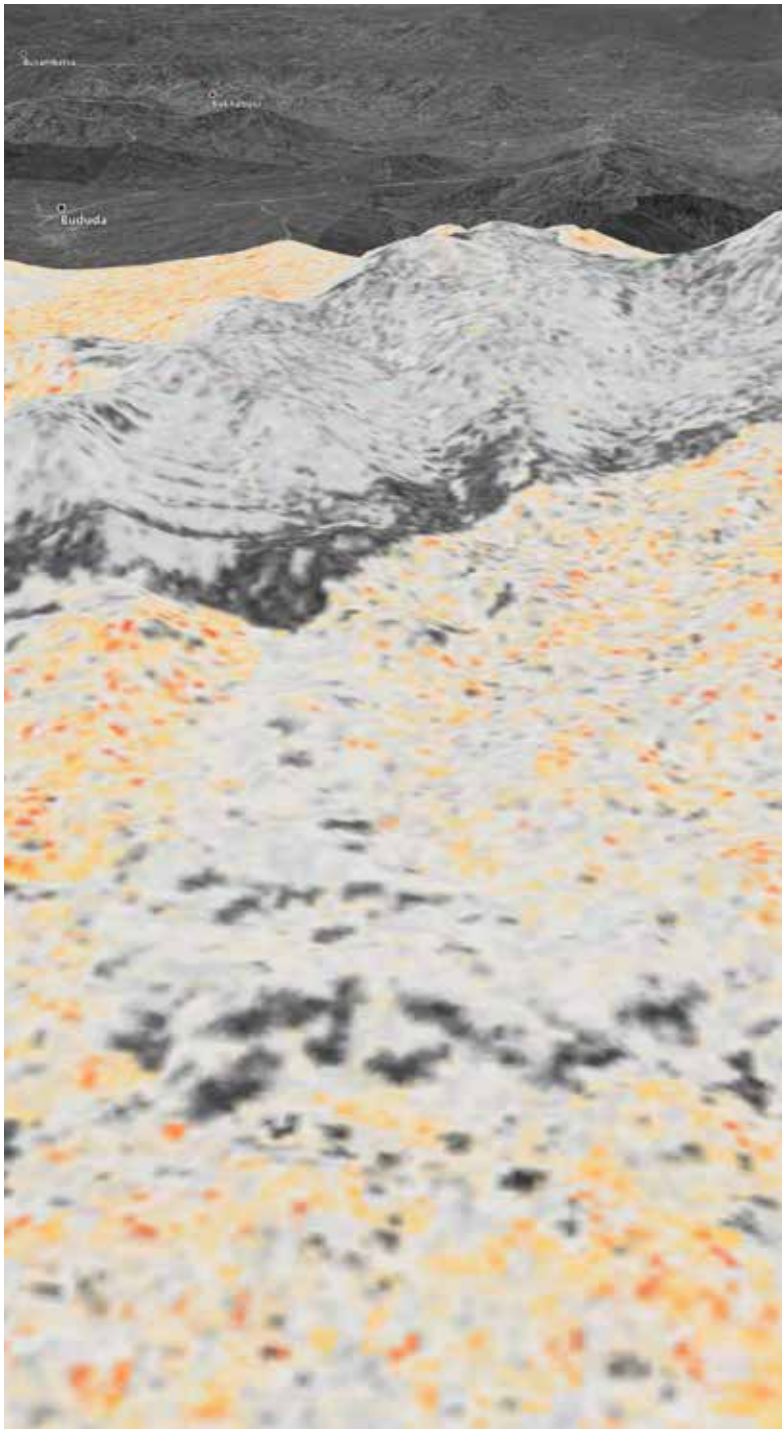


Table 3. The proportion of fallowed plots by vegetation structure class for each site.

Vegetation structure	Site		
	Mbale	Bumagabula	Campo Verde
	---------- % ----------		
Forest			17.5
Woodland			
Bushland			5.0
Shrubland			5.0
Thicket			62.5
Wooded grassland			5.0
Pasture (grassland)		37.5	5.0
Cropland	100.0	62.5	
Other			

Table 4. locally identified options for mosaic restoration presented according to existing CUM classes and results of a rapid assessment of existing plantation and silvo-pastoral systems conducted by ICRAF in 2014 (Esquivel, 2013, ICRAF internal report)

Major land use	Legally defined goal (CUM)	Landscape element	Restoration strategy
Forest	Reduced Impact Logging and/or conservation/ plantations*	Degraded pasture	Plantation of native timber and fruit tree species.
		Fallow	Enrichment planting with native timber species in association with management of natural regeneration.
		Remnant forest	Enrichment planting with valuable timber species.
		Inundated Forest with Aguaje	Plantation of native traditional fruit species.
Pasture land	Sustainable livestock production	Degraded pasture	Silvo pastoral systems
			Pepper in AFS
			Cacao in AFS
		Fallow / crop rotations	Enrichment planting with native timber and fruit species in association with management of natural regeneration
			Enrichment planting with leguminous to improve soil during cropping phase and other leguminous associated with permanent tree component
Protection	Respect of the climatic, edaphic, and topographical limitations or impediments (riverbanks, slopes etc.)		Reforest on riverbanks and headwaters
			Protection against fire

Description
Use of Inga sp (Guaba) to improve and protect soil in association with native species (fast growing and slow). Combination and spatial arrangement to be determined depending on specific site characteristics, level of degradation and species edaphic-climatic requirements. Combination and spatial arrangement to be determined depending on specific site characteristics, level of degradation and species edaphic-climatic requirements. High valuable timber species like mahogany (Swietenia macrophylla), Spanish cedar (Cedrela odorata), shihuahuaco (Dipteryx micrantha), tornillo (Cedrelinga cataeniformis), possibly nursed by fast-growing species such as Simarouba amara (marupá). High valuable local Mauritia flexuosa (e.g 'shambo' variety planted to recover degraded aguajales and protect water courses and sources.
Improvement of pastures with leguminous species (eg. Centrosema sp.) and use of quality fodder species including Leucaena Leucaena . Association with timebr species is possible, depending on specific site characteristics, level of degradation and species edaphic-climatic requirements Planting of pepper climbing on high value timber species or on living stakes of N_fixing species (e.g. G. sepium) with high-value species planted up to shade tolerance of black pepper Planting of guaba/pacae (Inga sp.) and other native fast growing species with cacao under shade. It is possible ot associate medicinal plants (Sangre de grado (Croton leucherii) , Una de gato (Uncaria tomentosa) Combination and spatial arrangement to be determined depending on specific site characteristics, level of degradation and species edaphic-climatic requirements. Keep alternating rotations of annual crops with enriched fallows and a permanent tree component
Management of natural regeneration of riverbank species like bolaina (Guazuma crinita) capirona (Calycophyllum spruceanum), aguaje, and other species Establishment and Management of Fire breaks, Clear fire management prescriptions Collective actions for fire control

One of the outputs of the current projects is a mobile phone application (app) to:

- Provide users with a free app that can run on affordable smart phones in Uganda,
- Determine the user's geographical position (co-ordinates),
- Determine the potential natural vegetation type at the user's location,
- Enable the user to select a subset of suitable tree species that provide desired products and environmental services, and
- Capture geo-tagged species and landscape imagery that document the presence of particular tree species or the current status of a landscape.

A workshop (Figure 35) was held in Mbale in July 2015, where participants were introduced to step-by-step installation of the smart phone application (Tree Finder App). The smart phone app was also field tested during this workshop and further work was done on its development, including a field data capture module after the workshop. App development is now in Beta mode and ready for users to test further in the field.

Figure 36 and Figure 37 illustrate the design and application of the Tree Finder App. The images collected using this mobile application can subsequently be used to monitor progress in restoration projects, analyze the accuracy or shifts of the original vegetation map or develop species and vegetation distribution models and maps.

Figure 35. Workshop participants during Tree Finder workshop in Mbale (June, 2015).

Figure 36. Illustration of the "Tree Finder App" field data entry and upload.

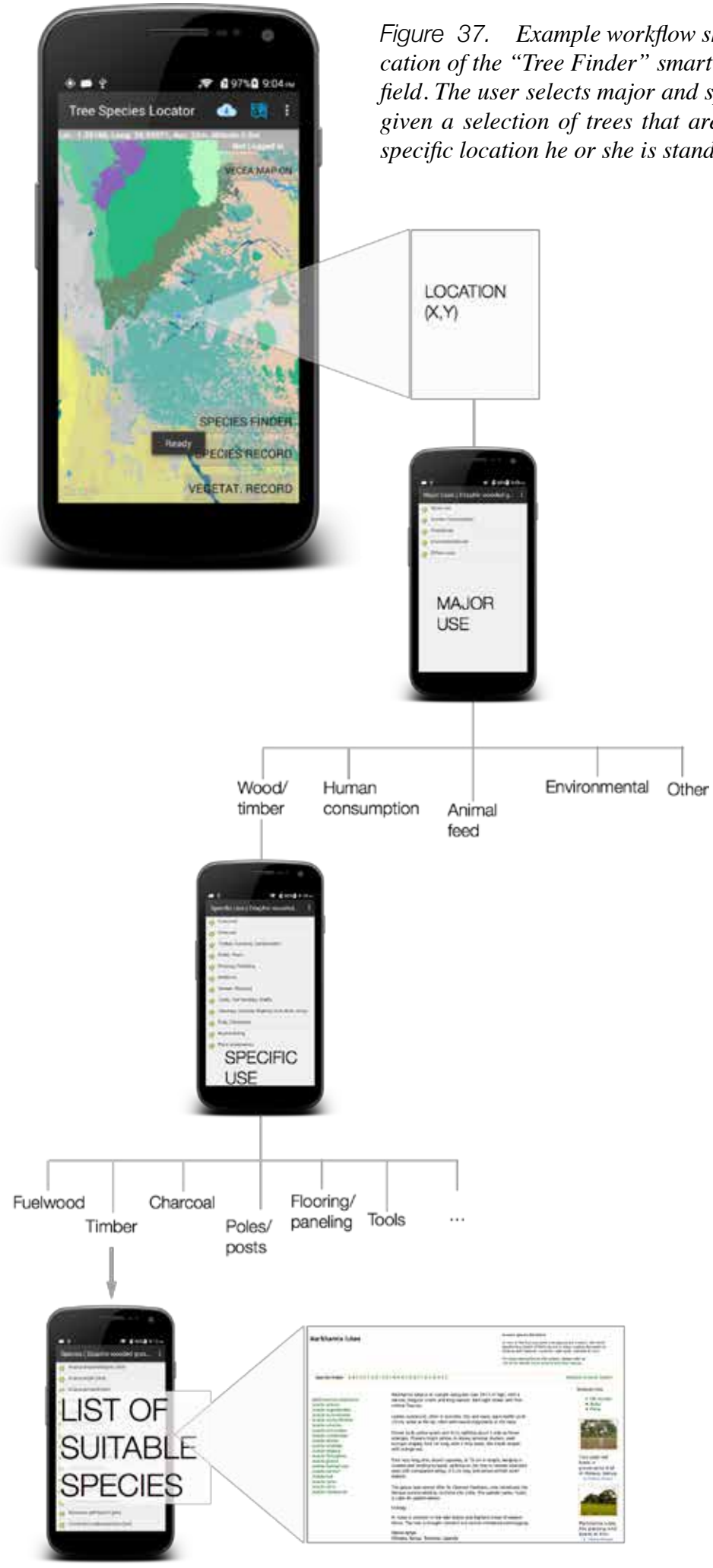


Figure 37. Example workflow showing the application of the "Tree Finder" smart phone app in the field. The user selects major and specific use and is given a selection of trees that are suitable for the specific location he or she is standing in.

References

Aronson, J., Alexander, S., 2013. Ecosystem Restoration is Now a Global Priority: Time to Roll up our Sleeves. Restor. Ecol. 21, 293–296. doi:10.1111/rec.12011

Barbier, E., 1997. The economic determinants of land degradation in developing countries. Philos. Trans. R. Soc. London B 352, 891–899.

Batterman, S.A., Hedin, L.O., van Breugel, M., Ransijn, J., Craven, D.J., Hall, J.S., 2013. Key role of symbiotic dinitrogen fixation in tropical forest secondary succession. Nature 502, 224–227. doi:10.1038/nature12525

Beddington, S.J., 2011. The future of food and farming. Int. J. Agric. Manag. 1, 0–4.

Benayas, J., Newton, A., Diaz, A., Bullock, J., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science (80-.). 325, 1121–1124.

Bongers, F., Chazdon, R., Poorter, L., Pena-Claros, M., 2015. The potential of secondary forests. Science (80-.). 348, 642–643. doi:10.1126/science.348.6235.642-c

Chambers, R., Leach, M., Conroy, C., 1989. Trees as savings and security for the rural poor, World Development.

Chazdon, R., 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. Science (80-.).

Chazdon, R., Coe, F., 1999. Ethnobotany of Woody Species in Second-Growth, Old-Growth, and Selectively Logged Forests of Northeastern Costa Rica. Conserv. Biol.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecol. Indic. 1, 3–10. doi:10.1016/S1470-160X(01)00003-6
Eswaran, H., Lal, R., Reich, P., 2001. Land degradation: an overview. Responses to L. Degrad.

FAO, 2010. GLOBAL FOREST RESOURCES ASSESSMENT 2010, Forest.

FAO, 2011. The State of the World's Land and Water Resources for Food and Agriculture. Rome.

Feldpausch, T., 2004. Carbon and nutrient accumulation in secondary forests regenerating on pastures in central Amazonia. Ecol. ….

Foresight, 2011. The Future of Food and Farming: Challenges and choices for global sustainability. London, UK.

Geist, H.J., Lambin, E.F., 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. Bioscience. doi:10.1641/0006-3568(2002)052[0143:P-CAUDF]2.0.CO;2

Gondard, H., Jauffret, S., Aronson, J., Lavorel, S., 2003. Plant functional types: a promising tool for management and restoration of degraded lands. Appl. Veg. Sci. 6, 223–234. doi:10.1111/j.1654-109X.2003.tb00583.x

Guo, L., Gifford, R., 2002. Soil carbon stocks and land use change: a meta analysis. Glob. Chang. Biol.

Junqueira, A., Harvey Shepard Jr, G., Clement, C., 2010. Secondary forests on anthropogenic soils in Brazilian Amazonia conserve agrobiodiversity. Biodivers. Conserv. 19, 1933–1961.

Kibblewhite, M.., Ritz, K., Swift, M.., 2008. Soil health in agricultural systems. Philos. Trans. R. Soc. B Biol. Sci. 363, 685–701. doi:10.1098/rstb.2007.2178

Laestadius, L., Maginnis, S., Minnemeyer, S., Potapov, P., Saint-Laurent, C., Sizer, N., 2011. Mapping opportunities for forest landscape restoration. Unasylva 62, 47–48.

Lalibertá, E., Wells, J.A., DeClerck, F., Metcalfe, D.J., Catterall, C.P., Queiroz, C., Aubin, I., Bonser, S.P., Ding, Y., Fraterrigo, J.M., McNamara, S., Morgan, J.W., Merlos, D.Sã¡., Vesk, P.A., Mayfield, M.M., 2010. Land-use intensification reduces functional redundancy and response diversity in plant communities. Ecol. Lett. 13, 76–86. doi:10.1111/j.1461-0248.2009.01403.x

Lamb, D., Erskine, P.D., Parrotta, J.A., 2005. Restoration of Degraded Tropical Forest Landscapes. Science (80-.). 310, 1628–1632. doi:10.1126/science.1111773

Laurance, W., Sayer, J., Cassman, K., 2014. Agricultural expansion and its impacts on tropical nature. Trends Ecol. Evol.

Lohbeck, M., Poorter, L., Lebrija-Trejos, E., Martínez-Ramos, M., Meave, J.A., Paz, H., Pérez-García, E.A., Romero-Pérez, I.E., Tauro, A., Bongers, F., 2013. Successional changes in functional composition contrast for dry and wet tropical forest. Ecology 94, 1211–1216. doi:10.1890/12-1850.1

Lohbeck, M., Poorter, L., Martínez-Ramos, M., Bongers, F., 2015. Biomass is the main driver of changes in ecosystem process rates during tropical forest succession. Ecology 96, 1242–1252. doi:10.1890/14-0472.1

Lohbeck, M., Poorter, L., Paz, H., 2012. Functional diversity changes during tropical forest succession. Perspect. plant

Lull, H., 1959. Soil compaction on forest and range lands.

Mace, G., 2014. Whose conservation. Science (80-.). 345, 1558–1560.

Marsett, R.C., Qi, J., Heilman, P., Biedenbender, S.H., Watson, M.C., Amer, S., Weltz, M., Goodrich, D., Marsett, R., 2006. Remote Sensing for Grassland Management in the Arid Southwest. Rangel. Ecol. Manag. doi:10.2111/05-201R.1

Martínez-Garza, C., Bongers, F., Poorter, L., 2013. Are functional traits good predictors of species performance in restoration plantings in tropical abandoned pastures? For. Ecol. Manage. 303, 35–45. doi:10.1016/j.foreco.2013.03.046

Minnemeyer, S., Laestadius, L., Sizer, N., 2011. Bonn challenge on forests, climate change and biodiversity. Washington, DC.

Parrott, L., 2010. Measuring ecological complexity. Ecol. Indic. 10, 1069–1076. doi:10.1016/j.ecolind.2010.03.014

Qi, J., Marsett, R., Heilman, P., Bieden-bender, S., Moran, S., Goodrich, D., Weltz, M., 2002. RANGES improves satellite-based information and land cover assessments in southwest United States. Eos, Trans. Am. Geophys. Union. doi:10.1029/2002EO000411

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., Venter, M., Boedhihartono, A.K., Day, M., Garcia, C., van Oosten, C., Buck, L.E., 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proc. Natl. Acad. Sci. 110, 8349–8356. doi:10.1073/pnas.1210595110

Smith, J., Finegan, B., Sabogal, C., do Socorro Goncalves, Maria Siles, G.G., van de Kop, P., Armando Diaz, B., 2001. Management of secondary forests in colonist swidden agriculture in Peru, Brazil and Nicaragua, in: World Forests, Markets and Policies. Springer, pp. 263–278.

Smith, J., van de Kop, P., Reategui, K., Lombardi, I., Sabogal, C., Diaz, A., 1999. Dynamics of secondary forests in slash-and-burn farming: interactions among land use types in the Peruvian Amazon. Agric. Ecosyst. Environ. 76, 85–98.

Unger, P., Kaspar, T., 1994. Soil compaction and root growth: a review. Agron. J. 86, 759–766.

Vågen, T., Shepherd, K., Walsh, M., Winowiecki, L., Desta, L.T., Tondoh, J.E., 2010. AfSIS technical specifications: soil health surveillance, World Agroforestry Centre, …. Nairobi, Kenya.

Vågen, T.-G., Winowiecki, L. a., Tondoh, J.E., Desta, L.T., Gumbricht, T., 2015a. Mapping of soil properties and land degradation risk in Africa using MODIS reflectance. Geoderma 0–9. doi:10.1016/j.geoderma.2015.06.023

Vågen, T.-G., Winowiecki, L.A., Abegaz, A., Hadgu, K.M., 2013. Landsat-based approaches for mapping of land degradation prevalence and soil functional properties in Ethiopia. Remote Sens. Environ. 134, 266–275.

Vågen, T.-G., Winowiecki, L.A., Tamene Desta, L., Tondoh, J.E., 2015b. The Land Degradation Surveillance Framework (LDSF) - Field Guide v4_1. World Agroforestry Centre, Nairobi, Kenya.

Vågen, T.G., Lal, R., Singh, B.R., 2005. Soil Carbon Sequestration in sub-Saharan Africa: A review. L. Degrad. Dev. 16, 53–71. doi:10.1002/ldr.644

van Oudenhoven, A.P.E., Petz, K., Alkemade, R., Hein, L., de Groot, R.S., 2012. Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol. Indic. 21, 110–122. doi:10.1016/j.ecolind.2012.01.012

Voeks, R., 1996. Tropical forest healers and habitat preference. Econ. Bot.

Wilson, J., Strittholt, J.R., Slosser, N.C., Dellasala, D., 1999. Global Forest Restoration: A Review. Corvallies, OR.

Winowiecki, L., Vågen, T.-G., Huising, J., 2015. Effects of land cover on ecosystem services in Tanzania: A spatial assessment of soil organic carbon. Geoderma. doi:10.1016/j.geoderma.2015.03.010