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Rehabilitation of degraded dryland ecosystems – review

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Highlights

- The prospect of restoring degraded drylands is technically promising.
- The forest landscape restoration concept can be used as the overarching rehabilitation framework.
- Development of process-based models that forecast rehabilitation outcomes is needed.
- Rehabilitation methodologies developed for moist areas are not necessarily suitable for drylands.
- More data is needed on cost-benefit analysis of rehabilitation interventions.

Abstract

Land degradation is widespread and a serious threat affecting the livelihoods of 1.5 billion people worldwide of which one sixth or 250 million people reside in drylands. Globally, it is estimated that 10–20% of drylands are already degraded and about 12 million ha are degraded each year. Driven by unsustainable land use practices, adverse climatic conditions and population increase, land degradation has led to decline in provision of ecosystem services, food insecurity, social and political instability and reduction in the ecosystem's resilience to natural climate variability. Several global initiatives have been launched to combat land degradation, including rehabilitation of degraded drylands. This review aimed at collating the current state-of-knowledge about rehabilitation of degraded drylands. It was found that the prospect of restoring degraded drylands is technically promising using a suite of passive (e.g. area exclosure, assisted natural regeneration, rotational grazing) and active (e.g. mixed-species planting, framework species, maximum diversity, and use of nurse tree) rehabilitation measures. Advances in soil reclamation using biological, chemical and physical measures have been made. Despite technical advances, the scale of rehabilitation intervention is small and lacks holistic approach. Development of process-based models that forecast outcomes of the various rehabilitation activities will be useful tools for researchers and practitioners. The concept of forest landscape restoration approach, which operates at landscape-level, could also be adopted as the overarching framework for rehabilitation of degraded dryland ecosystems. The review identified a data gap in cost-benefit analysis of rehabilitation interventions. However, the cost of rehabilitation and sustainable management of drylands is opined to be lower than the losses that accrue from inaction, depending on the degree of degradation. Thus, local communities' participation, incorporation of traditional ecological knowledge, clear division of tasks and benefits, strengthening local institutions are crucial not only for cost-sharing, but also for the long-term success of rehabilitation activities.

Keywords land degradation; desertification; rangelands; croplands; dry forests; landscapes; restoration
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1 Introduction

Deforestation and land degradation has been well recognized as major threat to human-wellbeing and environment due to the resulting loss in biodiversity, soil degradation and significant contribution to greenhouse gas emission (UNCCD 1994). The interest on land degradation in relation to food security issues and ecosystems services has increased in the last 10 years (ELD Initiative 2013) after the realization that rehabilitated environments will play a central role in the provision of ecosystem services and the realization of the UN's Sustainable Development Goals. As a result, various global initiatives have been launched to restore degraded landscapes, notably "the Bonne Challenge" and "Convention on Biological Diversity". Restoration of degraded landscape is conceived as a triple win solution to regain ecological integrity, enhance human well-being in deforested or degraded landscapes and resilience to climate change (Pfund and Stadtmüller 2005). Global Partnership on Forest Landscape Restoration has been launched to address the "Bonne Challenge" (http://www.forestlandscaperestoration.org/) that envisages to restoring 150 million hectares of lost forests and degraded lands worldwide by 2020. It is estimated that restoring 150 million hectares will bring an annual net benefit to the national and local economy worth of more than 80 billion US dollar, sequester 1 billion metric tons of CO_2 equivalent per year and reduce the current greenhouse gas emission level by 20%.

The Aichi Biodiversity Target 15 of the Strategic Plan for Biodiversity 2011–2020 adopted under the Convention on Biological Diversity also envisages to enhance "ecosystem resilience and the contribution of biodiversity to carbon stocks by year 2020, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification" (https://www.cbd.int/sp/targets/). The rehabilitation of degraded dryland ecosystems also plays an important role in achieving several commitments, such as the Land Degradation Neutrality initiative by UNCCD, the Rome Promise for Drylands, the New York Declaration on Forest, the African Forest Landscape Restoration Initiative (AFR100), the African Resilient Landscapes Initiative (ARLI), and Reducing Emissions from Deforestation and Forest Degradation (REDD+). The Rome Promise, emerged from the Drylands Monitoring Week hosted by the Food and Agriculture Organization of the United Nations (FAO) also "recognized the importance of natural capital for national development and human well-being in drylands; concerned by the slow progress in addressing the continuing degradation of drylands; and convinced that an appropriate and sustainable monitoring and assessment system, including a comprehensive baseline and participatory approaches, is necessary for effective management and restoration of natural capital in drylands" (FAO 2016).

The implementation of these initiatives requires not only active restoration of degraded dryland ecosystems, but also targeting some of the drivers behind the degradation process, such as negative incentives, unsustainable practices, market failures, weak legal frames, lack of incentives and others. The aim of this review was to describe the proximate and underlying processes of dryland degradation and collate the current state-of-knowledge about rehabilitation of degraded drylands. The review mainly focused on the three main land uses, i.e. rangelands, dry forests, and croplands in arid, semi-arid and dry sub-humid drylands where numerous rehabilitation efforts have been made. The review was organized in seven sections that include assessment of dryland degradation processes, an overview of rehabilitation of degraded drylands, and specific rehabilitation measures for rangelands, croplands and dry forests. The economic cost of rehabilitation and the role of community participation were highlighted and finally relevant conclusions and recommendations are provided.

2 Dryland systems

Drylands are land areas, which receive relatively low amounts of precipitation, and areas in which annual mean potential evapotranspiration is at least 1.5 times greater than annual mean precipitation (Safriel et al. 2005). They are broadly defined as land areas with an aridity index (AI) value of between 0.05 and 0.65 (UNCCD 1994). Drylands can be divided into four subtypes based on AI: dry sub-humid (AI = 0.50–0.65), semi-arid (AI = 0.20–0.50), arid (AI = 0.05–0.20) and hyper-arid deserts (AI < 0.05). Drylands occur in all continents between 63°N and 55°S, covering 41.3% (6 billion ha) of the earth's land surface (Millennium Ecosystem Assessment 2005a). However, 72% of the global dryland areas are confined to developing countries while only 28% are within industrial nations.

The dominant land uses in drylands are rangelands and croplands, jointly accounting for 90% of dryland areas; while forests and woodlands account for 10% of the drylands. These land uses, in turn, support integrated agro-pastoral and silvo-pastoral livelihoods of more than 2 billion people, about one third of world population (Millennium Ecosystem Assessment 2005a). Apart from provisioning services (e.g. food, forage, fibre, biochemical), dryland ecosystems provide different regulatory services, including water, pollination and seed dispersal, and climate regulation by sequestering and storing vast amounts of carbon in the soil. Plant biomass per unit area of drylands is lower than many terrestrial ecosystems, but total soil organic and inorganic carbon reserves in drylands comprise 27% and 97%, respectively of the global soil organic and inorganic carbon reserves. Furthermore, dryland ecosystems play an important role in shaping cultural identity and diversity of their inhabitants as well as development of unique dryland farming systems.

3 Dryland degradation: extent, drivers and impacts

Dryland degradation is a widespread and a serious threat to the livelihoods of millions of people particularly in developing countries and the global environment (Millennium Ecosystem Assessment 2005a). The United Nations Convention to Combat Desertification (UNCCD 1994) defines land degradation as a reduction or loss, in arid, semi-arid, and dry sub-humid areas, of the biological or economic productivity and complexity of rain-fed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a single process or combination of processes including processes arising from human activities and habitation patterns. Alternatively, according to Land degradation assessment in drylands (LADA) (2013), land degradation is defined as the reduction in the capacity of the land to provide ecosystem goods and services, over a period of time, for its beneficiaries.

There is a widespread use of the term desertification either interchangeably or in conjunction with land degradation; the definitions of land degradation in drylands and desertification are highly controversial. Desertification is defined by the UNCCD (1994) as "land degradation in the arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities". Desertification is also defined as the process of ecological degradation by which economically productive land becomes less productive and, in extreme cases, develops a desert-like landscape incapable of sustaining the communities that once depended on it (Kassas 1988; Westing 1994).

Desertification constitutes a subset of land degradation – it is land degradation, but confined to three climatic regions – the dry sub-humid, the semiarid and the arid areas (Safriel 2009; UNCCD 1994). Thus, since desertification is land degradation in the drylands, then when desertification is used, land degradation is implicit (Safriel 2009). Based on the definitions of UNCCD, the terms

Table 1. Degraded area and population distribution in dryland zones.

Dryland zones	Size of the area (million km^2)*	Share of global area (%)*	Degraded area (million km^2)**	Degraded area (%)	Population***	Population density (km^2)
Dry sub-humid	12.96	8.7	2.5	19.5	909 972 000	70
Semi-arid	22.67	15.2	4.8	21.2	855 333 000	38
Arid	15.17	10.6	4.5	28.7	242 780 000	16
Total dryland	50.80	34.5	11.8	23.1	2 008 085 000	40

Sources:
* UN 2011. Global drylands: a UN system-wide response. Environment Management Group. 132 p. http://www.unccd.int/Lists/SiteDocumentLibrary/Publications/Global-Drylands-ENG.pdf.
** Zika and Erb 2009.
*** UNCCD 2011. Desertification: a visual synthesis. Bonn. 50 p.

desertification and dryland degradation are by and large synonymous. In the literature, there are considerable discrepancies and inconsistencies in figures related to desertification and land degradation in drylands, which may be a result of different definitions used by different authors. In order to avoid the confusion, in this article mainly the term dryland degradation is used instead of desertification, which is lacking clear and distinct definition (cf. Vogt et al. 2011).

Globally, it is estimated that 10–20% of drylands are already degraded (Millennium Ecosystem Assessment 2005a) and about 12 million hectares are degraded each year (Brauch and Spring 2009; James et al. 2013). Particularly, it is the semiarid zone which has the largest degraded area, adversely affecting the livelihoods of considerable number of people (Table 1). Worldwide, land degradation affects about 1.5 billion people, out of these 250 million people reside in drylands and about one billion people in over 100 countries are at risk (Reynolds et al. 2007; Lean 2009; UNCCD 2014).The number of people affected by dryland degradation may increase substantially as the estimated number of people living in dryland environments will increase from 2 to about 3 billion by 2020 (Fischer and Heilig 1997; Millennium Ecosystem Assessment 2005a; Biazin and Sterk 2013). These people include many of the world's poorest, most marginalized, and politically weak citizens (WMO 2005). In addition to the areas and people directly affected, dryland degradation has adverse impacts on non-dryland areas, often many thousands of kilometers away. For instance, dust storms resulting from reduced vegetation cover may lead to air quality problems, both locally and far away (Millennium Ecosystem Assessment 2005b).

Land degradation which is a serious threat to dryland ecosystems is a complex environmental problem that combines a natural and social cause-effect cycle (Warren 1993; Darkoh 2003; Reynolds et al. 2007; Whitfield et al. 2011; Torres et al. 2015; Wiesmeier 2015). Land degradation is driven by human activities (unsustainable land use), adverse climatic conditions such as extended or recurrent droughts, and population increase (UNEP 1997; Msangi 2004; Mganga et al. 2015; Oudenhoven et al. 2015). Natural forces, through periodic stresses of extreme and persistent climatic events, and human mismanagement (inappropriate use of irrigation, overgrazing, deforestation, urban sprawl) of sensitive and vulnerable dryland ecosystems, often act in unison, creating feedback processes (WMO 2005; Marques et al. 2016). Occasional droughts and long-term severe droughts can both be aggravated by the influence of humans on the environment. To better understand the drivers of land degradation, von Braun et al. (2013) make a distinction between proximate and underlying causes. Proximate causes are those with a direct effect on terrestrial ecosystems and can be further divided into natural (biophysical) and anthropogenic (management) causes; underlying causes (e.g. policies, population density, markets, poverty) will be those directly affecting the proximate causes. However, there is disagreement on the role of underlying drivers of land degradation due to the complex interactions between them (Nkonya et al. 2016) which in most cases are context specific.

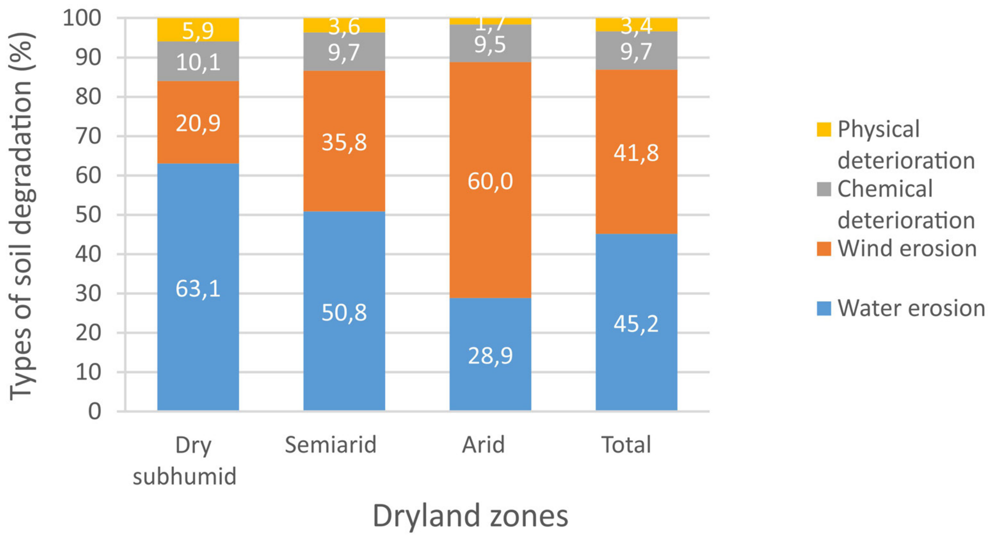

Fig. 1. Types of soil degradation (million km^2) in dryland zones. (Data source: UNEP1997).

Drylands are sensitive to degradation (Reynolds et al. 2007) and principal processes of land degradation include erosion by water and wind, chemical degradation (comprising acidification, salinization, fertility depletion, and decrease in cation retention capacity), physical degradation (comprising crusting, compaction, hard-setting, etc.) and biological degradation (reduction in total and biomass carbon, and decline in land biodiversity) (Sivakumar 2007). Although, globally, water is the main agent of soil erosion, in many arid and semi-arid areas, wind erosion can be substantial and even the dominant agent of soil erosion (Ravi et al. 2010; Belnap et al. 2011; Fig. 1) or even desertification (Brown and Nickling 2003). Dryland ecosystems that are most vulnerable to degradation are those that have low rainfall, long dry seasons, recurrent droughts, mobile surface deposits, skeletal soils, and sparse vegetation cover (Dregne 1983; Le Houérou 1986; Kassas 1995). Drought reduces the phytomass, abundance and ground cover of plants and hence reduces the protection of the soil against erosion (Grainger 1992). Eventually, if all of the top soil layers are removed, the situation becomes irreversible (Bullock and Le Houérou 1996; Ravi et al. 2010).

Key environmental indicators of dryland degradation include net primary productivity, presence/absence of indicative plant species, soil organic matter, and several chemical and physical soil properties (Wiesmeier 2015). In addition, the vegetation/bare soil pattern is an effective indicator of degradation/desertification that can be used to detect it, particularly at the early stages (Wiesmeier 2015).

Climate change may exacerbate and itself is exacerbated by land degradation (Cowie et al. 2011). Climate change may aggravate dryland degradation through alteration of spatial and temporal patterns in temperature, rainfall, solar radiation and winds (WMO 2005; Sivakumar 2007; D'Odorico et al. 2013; Huang et al. 2016). In addition, dryland degradation is associated with biodiversity loss (Bisaro et al. 2014) and contributes to global climate change through loss of carbon sequestration capacity (Lal 2001; Lal 2002; Plaza-Bonilla et al. 2015; Huang et al. 2016) and an increase in land-surface albedo (Millennium Ecosystem Assessment 2005a). Moreover, dryland degradation may release a major fraction of this carbon to the global atmosphere, with significant feedback consequences to the global climate system. It is estimated that 300 million tons of carbon are lost to the atmosphere each year from drylands as a result of degradation (Millennium Ecosystem Assessment 2005a), with an accumulated loss of up to 30 Pg of carbon (Lal 2004).

The impact of climate change on drylands should also be considered in parallel with the effects of existing pressures caused by unsustainable land management – because it is often impossible to separate the effects of these impacts and because their cumulative impact on soils often is greater than a simple summation (Bullock and Le Houérou 1996). As a consequence of climate change, the extent of global dryland area is projected to increase particularly in developing countries, which further increases the proportion of the population affected by water scarcity and land degradation (Huang et al. 2016).

According to UNCCD, the consequences of land degradation include decline in soil productivity and food production, famine, increased social costs, decrease in the quantity and quality of fresh water supplies, increased poverty, political instability, and reduction in the land's resilience to natural climate variability. Dryland degradation leads to regressive succession of ecosystems and converts them to more open vegetation formations with less structural complexity, productivity and resilience (Vieira and Scariot 2006; Zika and Erb 2009; Oudenhoven et al. 2015; Fig 2). Overall, it is estimated that between 4% and 10% of the dryland potential net primary production are lost each year due to dryland degradation (Zika and Erb 2009). Long-term food productivity is threatened by soil degradation, which is now severe enough to reduce yields on approximately 16% of the agricultural land, especially cropland in Africa and Central America, and pastures in Africa. Sub-Saharan Africa has the highest rate of land degradation affecting about 46% of the region.

Damaged land is costly to reclaim and, if severely degraded, its inability to provide ecosystem functions and services leads to a loss of environmental, social, economic and non-material benefits that are critical for society and development. Dryland degradation causes income losses to farmers and pastoralists due to loss of land productivity and also entails costs for land rehabilitation (Zika and Erb 2009). At the global level, it is estimated that the annual income foregone in the areas immediately affected by dryland degradation amounts to approximately 42 billion USD each year (WMO 2005; James et al. 2013). Much of this degradation is undermining economic development and part of the severe degradation may be irreversible. In addition to the economic loss, land degradation is one of the main causes of human migration and can adversely affect local, regional, and even global political and economic stability (Millennium Ecosystem Assessment

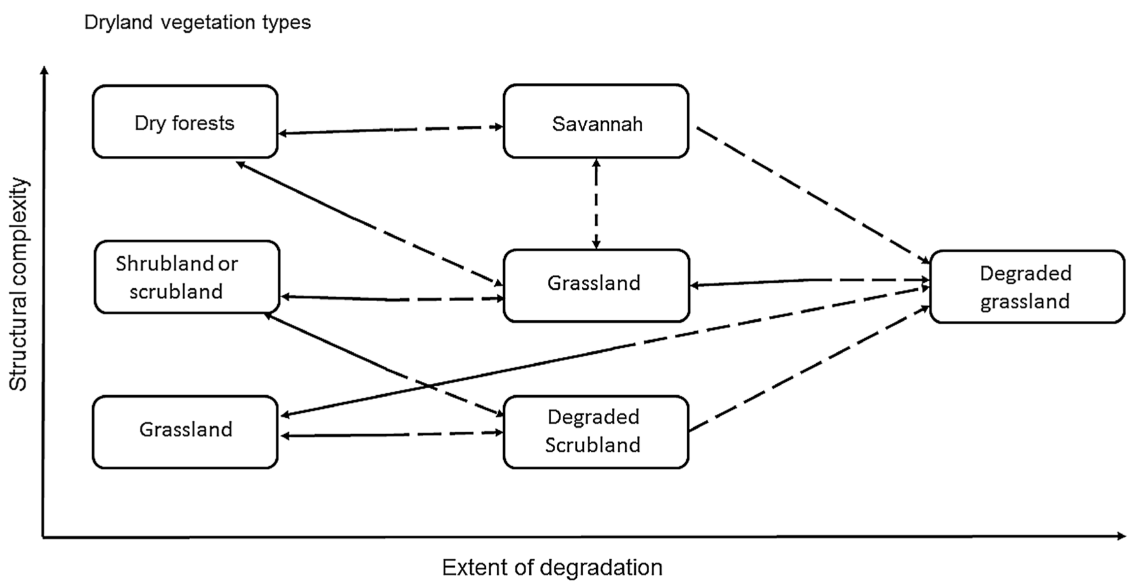

Fig. 2. The extent of dryland disturbance and consequently degradation affects vegetation structural complexity. Dryland vegetation succession (———►) and regression (— — ►).

2005a; Zika and Erb 2009). Hence, the development and adoption of sustainable land management practices is one of the major solutions to combat the problem over the vast drylands around the world (WMO 2005).

4 Rehabilitation of degraded dryland ecosystems

Rehabilitation is defined as the reparation of ecosystem processes, productivity and services without necessarily achieving a return to 'pre-disturbance' conditions (Mansourian 2005; CBD 2012). Although the terms restoration and rehabilitation are used interchangeably in the literature, the term rehabilitation describes best by far most of the remedial activities that have been conducted in degraded dryland ecosystems and hence mainly the term rehabilitation is used in this article.

Rehabilitation of degraded dryland ecosystems is quintessential for the conservation of the threatened and unique dryland biodiversity. Generally, ecological restoration is regarded as an effective way for the enhancement of both biodiversity and the provision of ecosystem services and consequently contributes also to sustainable development (Puigdefábregas 1998; Schiappacasse et al. 2012). Moreover, rehabilitation of drylands enhances the adaptive capacity and resilience of local communities (by improved and diversified livelihood sources) and ecosystems (Lake 2013; United Nations University 2014).

The first step in the rehabilitation of degraded dryland ecosystems should be identifying and addressing the drivers of the degradation; that lays the foundation for the sustainability of rehabilitation endeavours (Le Houérou 2000; Reynolds et al. 2007). In general, the rehabilitation strategy for dryland ecosystems depends, amongst other factors, on the intensity, duration, frequency, and scale of the perturbation and the availability of propagules. However, recovery is difficult if the degradation is severe and the ecosystem has crossed an ecological threshold and reached a new steady state (Le Houérou 2000; ITTO/IUCN 2005; Lamb et al. 2005). In general, due to limited water availability the rehabilitation of degraded drylands is more challenging (Constantini et al. 2016) and slower than ecosystem recovery in moist sites.

Dryland rehabilitation activities have been usually focused on a specific site or on a plot scale. However, environmental factors and processes which include soils, climate, topography, hydrology, land management, water management, and ecological systems operate at much larger scales and are interlinked (Cost Action 2016). Hence, landscape-level planning is rather pertinent for the rehabilitation of degraded drylands, where the multiple functions of the different land uses are taken into account. Moreover, it is easier to make the trade-off between rehabilitation and livelihoods requirements at a landscape-level than at the site-level (Lamb et al. 2005; ITTO/IUCN 2005; Yirdaw et al. 2014).

Landscape level analysis of the drivers of degradation or evaluation of the consequences of dryland degradation are not difficult to find (e.g. Drescher 1995; Yang et al. 2006; Sternberg 2008; Le Polain de Waroux and Lambin 2012; Zhang et al. 2014); however, dryland rehabilitation experiences or research at landscape level are much less common (e.g. Dorrough and Moxham 2005; Ravi et al. 2009) and often refer to protected landscapes (e.g. Dudley et al. 2014; Sankey and Draut 2014). This is not surprising considering the complex socioeconomic, ecological, cultural and sociopolitical interactions present in most dryland landscapes; in many cases, the geographical extent of these landscapes made them fall under several national or even international jurisdictions which makes landscape rehabilitation efforts difficult to implement. Similarly, experiences from dry landscapes are less common in the grey literature compared to the humid landscapes, but some studies are available from South America (Newton and Tejedor 2011); from Africa to Asia (Heshmati and Squires 2013), and the Mediterranean region (Bautista et al. 2009). As a consequence of the lack of studies at the landscape level there is also a paucity of scientific information on the dynamics of landscape processes. Understanding of the landscape processes, which includes both natural and social processes is key in forecasting landscape level changes, which are initiated as a result of dryland rehabilitation.

4.1 Rangeland rehabilitation

Rangelands are the largest terrestrial ecosystems, comprised of natural grasslands, savannas, shrub lands, deserts, tundra, alpine communities, coastal marshes, and wet meadows (United States Department of the Interior Bureau of Land Management 2007). They are estimated to cover around 40% of the Earth's terrestrial surface, of which more than 80% are located in arid and semi-arid areas (Branson et al. 1981). Rangeland ecosystems are major providers of critical ecosystem goods and services, including food, water, and livelihoods for many of the world's poor. According to FAO (2008), pastoral livestock production is a major livelihood system for the majority of 1.2 billion people living on less than $1 per day (FAO 2008).

Rangelands had undergone rapid transformations for centuries as a result of unsustainable land use practices and the associated impact on hydrology, soil processes, and vegetation composition. Still today, substantial parts of the degraded rangelands are used in unsustainable manner worldwide; exacerbating the risk of losing their social-ecological resilience (Asner et al. 2004). Degraded rangelands are generally characterized by sustained reduction of biological and economic productivity. The causes and drivers of rangeland degradation are spatially and temporally complex. The proximate causes of rangeland degradation are overgrazing, deforestation, mining, bush encroachment, invasion by non-native plant species, and plowing of rangelands with subsequent loss of soil productivity, while policies, socio-economic changes, or interactions of socio-economic and governance factors with climatic conditions (e.g. drought) are ultimate drivers of rangeland degradation (Wilcox and Thurow 2006; Bedunah and Angerer 2012).

Livestock grazing is often perceived as the prime cause of rangeland degradation (Müller et al. 2007; Wessels et al. 2007); having a critical impact on rangeland biodiversity (Angassa 2014), soil erosion (Tadesse and Penden 2002; Mekuria and Aynekulu 2013), soil nutrient cycling (Fernandez et al. 2008), and hydrological processes (Savadogo et al. 2007). Others argue that livestock grazing coupled with rainfall variability, and soil and vegetation types have a long-term effect on productivity of rangeland vegetation (Savadogo et al. 2009; Cheng et al. 2011; Kgosikoma et al. 2015; Yayneshet and Treydte 2015). Available evidence also shows that it is the management of the grazing system, not grazing per se, which is the main cause of rangeland degradation in arid and semi-arid environments (Gulelat 2002; Savadogo et al. 2008). For instance, the traditional free-grazing system by pastoralists can lead to overgrazing, depending on livestock stocking density and frequency of grazing. Whereas improved grazing management system (e.g., cut and carry system, rotational grazing in alternating exclosures) enables better control of stock density and frequency of grazing; thereby minimizing the risk of overgrazing and rangeland degradation (Danano 2011; Lindeque 2011).

Bush encroachment and invasion by non-native woody species are serious threats to rangeland ecosystems, particularly by reducing grass cover, which in turn affects the pastoral livestock production system (Dalle et al. 2006; Eldridge et al. 2011; Angassa 2014). However, a 2000-year record of vegetation change in the Dara range of the Mago National Park, southwestern Ethiopia proves that the system is resilient, with alternating open and encroached phases, as along as the encroachment is within the thresholds conditioning the system's resilience (Gil-Romera et al. 2010). The aggressive invasion of rangelands by alien woody species (e.g. *Prosopis juliflora* (Sw.) DC.) has resulted in a decline in livestock productivity, an increase in incidence of health problems for both livestock and humans, and exacerbated biodiversity loss as a result of displacement of indigenous flora, loss of habitat for wild fauna, and blockage of water sources (Yirdaw et al. 2014). However, alien invasive plants are not always a menace, as observed in the Cerrado vegetation where the alien grass *Melinis minutiflora* P. Beauv. increased the above-ground biomass by 40% within two-year period (Martins et al. 2011). The degradation of rangelands is further complicated by rangeland conservation policies, such as exclusion of grazing, the ban on traditional burning practice and tenure insecurity, which are proven to bring no additional benefits but rather exacerbate the degradation process (Solomon et al. 2007; Savadogo et al. 2008; Angassa et al. 2012).

Generally, there are two approaches to rehabilitation of degraded rangelands: passive and active. In moderately degraded rangelands where there is still some vegetation cover to serve as succession primer, passive restoration techniques, such as livestock exclosure for a certain period of time, can be effective ways. For instance, exclosures established in northern Ethiopia have been effective in restoring plant species composition, diversity, biomass, cover, and structure of both herbaceous and woody components (Yayneshet et al. 2009), and improving soil nutrient status, and reducing erosion (Mekuria et al. 2007). In a savanna rangeland of southern Ethiopia, Angassa et al. (2012) have also found higher herbaceous biomass, grass basal cover, herbaceous species richness and diversity in traditional exclosures than in open grazed areas. In Jebel Samhan protectorate of the sultanate of Oman, Said et al. (2013) have found that plant coverage ranged from 36–98%, while vegetation productivity was 63% higher in the Tawi Atier exclosure than adjacent sites.

Rangeland resting or grazing-exclusion for a period of time is also another passive restoration technique to rehabilitate moderately degraded rangelands. In Tunisia, for instance, rangeland resting has been practiced traditionally, where increases in fodder production, soil organic matter, and biodiversity, while reduction in soil erosion have been observed (Ouled-Belgacem 2012). In Niger, distribution of points of available water, building water harvesting structures, and facilitating passageways for herds have been adopted as passive restoration techniques to rehabilitate rangelands, which is proved to be successful to reduce overgrazing problem by 30–40% (Marques et al. 2016).

Severely degraded rangelands can be restored through active restoration approaches, such as reseeding grass species, control and reduction of woody species encroachment, planting fodder trees, improved grazing system and assisting natural regeneration of native species. Grass reseeding technology has been used successfully as a means of rehabilitating degraded rangelands in East Africa (Musimba et al. 2004; Tebeje et al. 2014). Planting of *Pennisetum pedicellatum* Trin. (a local grass species) in combination with legumes and fodder tree seeds was tested in the overgrazed highlands of Ethiopia, and yield a variety of benefits, including fodder and wood production, soil protection, increased fertility, and biodiversity enhancement (Danano 2011). The rehabilitation success, however, is dependent not only on the type of species planted but also on improvement of site conditions by soil scarification, mulching and manure additions. It should be noted that severely degraded rangelands are often poor in soil nutrients to support rapid establishment and growth following seeding or planting.

Prescribed annual early fire, moderate level of grazing and tree cutting have been tested as management regimes in the Sudanian savanna-woodland in Burkina Faso to favor the growth of both herbaceous and woody components. It was observed that grazing tended to favor the diversity of perennial grasses; fire tended to influence the richness of annual grasses and abundance and diversity of perennial grasses, selective tree cutting had no effect on any of the vegetation attributes assessed, and the combined effect of grazing, fire and selective cutting tended to increase the diversity of forbs (Savadogo et al. 2008). The response of herbaceous vegetation to these management regimes exhibited spatial and temporal variabilities, which could be related with soil conditions and inter-annual variation in rainfall (Savadogo et al. 2009). A diverse approach for restoring degraded rangelands in the Succulent Karoo Biome has been implemented and tested in four major ecological regions of Namaqualand (i.e. Richtersveld, Coastal plain, Namakwaland Klipkoppe, Knersvlakte), South Africa (Hanke and Schmiedel 2010). The active restoration measures included brush-packing (covering the soil surface with branches of indigenous shrub species), dung mulching, creating micro-catchments, planting of key species of the natural plant community (functional plants), and stone cover to minimize drought-induced plant mortality, reduce wind speed and to catch wind-blown seeds, soil particles, and organic material. Useful insights about physical and biological process, with implications for restoration practice, were drawn.

Different restoration techniques have been tested to control or suppress woody species encroachment. In the Savanna of Southern Ethiopia, effects of tree cutting, fire, grazing and their combinations on herbaceous vegetation were evaluated, and the results show that tree cutting and fire treatments yielded higher herbaceous biomass, while herbaceous species diversity was improved more by the traditional method of fire and grazing, as well as cutting of woody species (Angassa et al. 2012). Tree cutting and fire combined with grazing were also more effective in suppressing the regeneration of encroaching species (Angassa and Oba 2009). In a similar area, Negasa et al. (2014) evaluated different cutting techniques of two encroaching species (*Acacia drepanolobium* Harms ex Sjostedt and *Acacia mellifera* (Vahl) Benth.), and the results show that cutting at 0.5 m above ground and either debarking the stumps down into the soil surface or dissecting the stumps were effective in controlling *A. drepanolobium* and *A. mellifera*, respectively. The bush encroachment control techniques have also improved herbaceous biomass and plant biodiversity while reducing the population of tick. Ticks are ecto-parasites (external parasites) that live by feeding on the blood of animals and are vectors of a number of diseases that affect both humans and other animals. Heavy infestation by ticks often damages and closes off cow teats; thereby reducing milk yield.

Suppression of shrub encroachment can be contentious in the face of climate change. Recruitment of shrubs in rangelands could be a natural secondary succession process (a form of passive restoration) that will eventually transform the savanna into woodland (Fig. 2); which in turn ameliorate the microclimate, store more carbon in the woody biomass and contribute to abatement of desertification. As a whole, restoration of degraded rangelands is challenging due to the complex nature of degradation causes and drivers. However, the cases presented here give some useful insights to design local-specific restoration approaches

4.2 Rehabilitation of degraded croplands

Rainfed and irrigated agriculture (which is the most intensive and productive form of cropland system in drylands) are the main cropping systems practiced in drylands all over the world. Croplands are the second most extensive land use after rangelands accounting for about 25% of the land use in drylands (UN 2011). Croplands have been extensively degraded in drylands among others due to population pressure, policy and market failures, and inappropriate land use practices and farming techniques (FAO 2008). Degradation of farmlands is further amplified when it coincides with drought, which temporarily but drastically reduces soil and plant productivity (Millennium Ecosystem Assessment 2005b). It is estimated that crop cultivation has caused soil degradation on 235 million ha of land area and accounts for 23% of total soil degradation in drylands (UNEP 1997).

As a result of intensive agricultural activity and overgrazing dryland soils resources are affected by various types of soil degradation, such as water and wind erosion, salinization, soil compaction, crusting, declining soil organic matter, top soil losses and nutrient depletion (Zika and Erb 2009). As a consequence of extensive water and wind erosion there is a reduction in soil depth and ability to store water and nutrients (Mainguet 1986; Grainger 1992). Eventually, dryland soil degradation leads to declines in agricultural productivity, environmental degradation, and food insecurity (Scherr 1999; Dregne 2002; Mganga et al. 2015).

Land degradation and the challenges related to soil fertility and agricultural production are particularly acute in Sub-Saharan Africa, where crop yield gains are greatly needed. It is estimated that losses in productivity of cropping land in sub-Saharan Africa are in the order of 0.5–1% annually, suggesting productivity losses of at least 20% over the last 40 years (WMO 2005). In addition to reduced productivity, land degradation leads to socio-economic problems such as food insecurity and migration. With the forecasted world population increase, there will be mounting pressure on the soil to produce more food; thus, future trends in land productivity and food production are of particular interest.

The scarcity of water in arid areas promotes natural soil salinization process, which is one of the main causes of dryland degradation and desertification. Salt accumulation at or near soil surface hinders the growth of most plants, and hence reduces agricultural crop productivity (Tejdor et al. 2007). Human-induced salinization is a major contributor to desertification, which is often associated with irrigation schemes or with rising groundwater levels due to conversion of natural vegetation to annual crops as in the case of Southern Australia and parts of Central Asia. Globally, nearly 50% of the irrigated land in arid and semi-arid region has some degree of salinization problem and every year about 1.5 million ha of irrigated land loses 25–50% of their productivity due to salinity (Thomas and Middleton 1993; Rubio and Calvo 2005). In addition to salinization, irrigation has also resulted in waterlogging, water pollution, eutrophication, and unsustainable exploitation of groundwater aquifers (Millennium Ecosystem Assessment 2005a).

Soil reclamation in dryland agricultural areas can be achieved by biological, chemical and physical measures. Tree planting considerably reduces soil erosion, dust storms and siltation of streams and water bodies (Sterk et al. 2001). Generally, vegetation cover and plant litter on soil surface protect the soil from erosion by reducing runoff and increasing water infiltration into the soil matrix (Zuazo and Pleguezuelo 2008; Liao et al. 2014). Afforestation or reforestation with salt-tolerant species can ameliorate saline soils to productive lands by the production of litter which increases the soil organic matter and nitrogen content (Mishra et al. 2003) and by lowering the water table. Dryland agroforestry system can be a valuable tool to replenish soil fertility; thereby enhancing land productivity and food security, particularly for smallholder farmers (Erdmann 2005). Leguminous trees within dryland agroforestry systems contribute to soil fertility by fixing atmospheric nitrogen and inputting into the soil, retrieving of nutrients from below the rooting zone of crops, and reducing nutrient losses from leaching and erosion (Buresh and Tian 1998). The woody component can also provide possibilities for the use of green and animal manure for the amelioration of the soil (Pinho et al. 2012; Marques et al. 2016). In addition, agroforests enhance the diversity and abundance of the soil biota and nutrient cycling.

However, the reforestation or afforestation of watersheds previously covered by native grasslands can reduce stream flow due to higher water use by trees (Matyas et al. 2013). Moreover, the conversion of land fully covered by grasses to one covered by scattered bushes creates greater bare soil surfaces alongside 'fertility islands' around shrub canopy patches, which encourages increased runoff and reduce resources redistribution, resulting in higher soil erosion (Millennium Ecosystem Assessment 2005a; Field et al. 2012). Hence complementing tree planting with physical soil conservation measures (terraces, bunds, and check dams) may reduce soil erosion more effectively (Nyssen et al. 2010). As a whole, reforestation as dryland soil reclamation measures should be site specific and take into considerations, among others, site history, the type and degree of soil degradation, microclimatic conditions and broad socio-economic factors.

Another biological soil reclamation measure is the use of cynobacteria, which are capable of photosynthesis and N_2-fixation; thereby contributing substantial quantities of carbon and nitrogen to dryland soils (Mager and Thomas 2011). They form the major component of microbiotic soil crusts in arid environments (Issa et al. 2007). Cyanobacterial soil crusts improve soil biochemical and physical properties – they have a higher organic matter and nitrogen content, and maintain surface stability and decrease the effect of erosion on dryland soils (Thomas and Dougill 2007). Moreover, cyanobacterial crusts increase water retention capacity of soils (Issa et al. 2007). Because of their soil ameliorative effects, cyanobacteria have been used as inoculants for improving soil structure, increasing soil fertility and recovering of damaged soil crusts (Pandey et al. 2005; Issa et al. 2007). In spite of their apparent importance, there is only a limited understanding of their possible applications in dryland soil environments (Mager and Thomas 2011). Termites, well-known ecosystem engineers, are key component of drylands that can modify soil physico-chemical and microbial properties through foraging and processing of plant materials, thereby creating "nutrient reservoirs" compared with the surrounding savanna soil ecosystems. By enhancing termite activity using mulching, Mando et al. (1996) observed good regeneration of woody plants on structurally crusted soils. Traoré et al. (2008) also observed that termite mounds are safe sites for the recruitments of woody plants in tree savanna.

The use of volcanic material (basaltic pyroclasts) as mulch was shown to decrease the salinity of the soil by as much as 86% (Tejdor et al. 2007). The decrease in salinity was attributed to the leaching of soluble salts beyond the root zone and to less evaporation, which prevents the salt from rising to the root zone (Tejdor et al. 2007). In recent years, biochar, produced by pyrolysis of different organic wastes, has gained increasing attention as soil conditioner for improving soil quality, plant growth and yield (Laghari et al. 2016). Another technical solution is the use of synthetic polymers and biopolymers to improve soil physical properties, such as water retention and infiltration capacity of the soil. Moreover, by increasing the soil structural stability polymers reduce soil erosion and run-off (Ben-Hur 2006; Maghchiche et al. 2010). As polymers differ in their molecular weight, molecular conformation, type of charge, and charge density, the selection of the right polymer depends on the structure and composition of the degraded soil and the environmental conditions it is exposed (Ben-Hur 2006).

In the case of croplands, rehabilitation activities should include sustainable land use practices that promote: increase land productivity, provision of ecosystem services, and improvement of livelihoods (USAID 2014). Successful cases have included income generation and livelihoods diversification as a shared objective alongside the rehabilitation of soils and ecological processes. (Thomas et al. 2014)

4.3 Rehabilitation of degraded dry forest landscapes

Dry forests are widespread in tropical and sub-tropical parts of the world, in areas where there is pronounced seasonality in rainfall distribution with several months of drought (Mooney et al. 1995; FAO 2012). Dry forests are one of the major forest types and account for about 42% of the tropical and subtropical forests (Murphy and Lugo 1995). However, it is difficult to determine the original extent of dry forests because some of the present day savannas and woodlands may have been derived from dry forests (Murphy and Lugo 1986; Bianchi and Haig 2013). The largest areas of dry forests are found in South America, sub-Saharan Africa and northeast India (FAO 2012; Blackie et al. 2014). Sizeable concentrations of dry forests are also found in Southeast Asia, Central America and the Caribbean, Sri Lanka, northern Australia and the Pacific islands (Miles et al. 2006; Blackie et al. 2014).

What constitutes a dry forest has been a contentious issue (Blackie et al. 2014) since the publication of Yangambi's phytogeography classification in 1956, as several definitions have been suggested and in some areas the difference between dry forest and other wooded land is not very clear (FAO 2010). Using the FAO definition for forest (area larger than 0.5 ha, min. height of 5 meters and canopy cover larger than 10%), several transition formations between shrub savanna and savanna woodland, some steppe formations with enough woody vegetation, areas with disperse tree cover (e.g. parts of the Cerrado in Brazil), and some spiny thickets (e.g. Madagascar) can be classified as a dry forest. More recently, the definition of forest as part of the United Nations Framework Convention on Climate Change (UNFCCC) and the Global Land Cover Facility (GLCF) have lowered the minimum height of woody vegetation to 2 meters, thus potentially expanding the area covered by 'dry forests'.

Tropical and subtropical dry forests are the most threatened forest types in all regions and they are less protected than other ecosystems (Miles et al. 2006; Portillo-Quintero and Sánchez-Azofeifa 2010) particularly in the pacific islands (Gillespie et al. 2012). Anthropogenic disturbance is the primary cause of dry forest degradation, deforestation and fragmentation and consequently disruption of ecosystem function and services on which local people depend for their livelihoods. The large-scale conversion of dry forests to other land use types is due to the ease of using fires to clear dry forests, relatively rich soil fertility, mineral deposits and the suitability for cattle rearing and human settlement (Aronson et al. 2005; Powers et al. 2009). It is estimated that about 48.5% and 66% of the tropical dry forests in the world and the Americas, respectively have been converted to other land uses (Hoekstra et al. 2005; Portillo-Quintero and Sánchez-Azofeifa 2010). The main drivers of dry forest conversion to other land uses may vary from one region to the other. Although not specifically about dry forests, Hosonuma et al. (2012) indicates that for Latin America the main driver of forest conversion is commercial agriculture while in Africa and Asia commercial agriculture and subsistence agriculture are both the key drivers; in terms of forest degradation, logging is the main driver in Latin America and Asia while fuel wood collection is the main driver in Africa.

For the sustainable rehabilitation of the denuded and degraded dry forests, a landscape-approach that takes into account the heterogeneous mosaic of different land uses with their interactions and local peoples' livelihoods is a judicious method. Furthermore, in this approach key ecological functions that operate at a landscape scale, and multiple objectives can be addressed (Lamb 2005; Sabogal et al. 2015). In this regard, the forest landscape restoration concept which is defined as "a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded landscapes" (Dudley et al. 2005) can be applied as the overarching framework in the rehabilitation of degraded dry forests. The application of forest landscape restoration has to incorporate other commonly found land uses in dry forest landscapes, such as livestock husbandry, small scale farming, agroforestry, etc. in order to balance the trade-offs among the different land users (ITTO 2002).

At the habitat level, the rehabilitation strategy for degraded dry forests should be site-specific and depends among others on proximity to seed sources, soil seed banks, populations of seed dispersing animals, and site history particularly past disturbance regimes (Elliot et al. 2013; Hoffman et al. 2015). The rehabilitation strategies developed for moister forests may not be the best suited for dry forests. The high proportion of small-seeded wind dispersed species, the high sprouting ability and the relatively simple structure and low diversity of dry forests in comparison to moist forests (Vieira and Scariot 2006) should be taken into considerations in the selection of rehabilitation strategies (Marques et al. 2016). Depending on the site conditions and the objectives either a single or combinations of different rehabilitation methods can be utilized.

Due to its resilient nature, passive restoration is usually suggested for dry forest when the level of degradation still allows for natural regeneration (ITTO 2002; Aronson 2005), this implies in most cases regulating key drivers of degradation such as fires, overgrazing, firewood collection, and shifting cultivation. Area exclosures and assisted natural regeneration (ANR) are passive forms of restoration, which are relatively simple and low-cost to implement in rehabilitation of degraded dry forests. They aim at protecting rehabilitation sites from human and animal disturbances enhance natural regeneration, plant and animal diversity, vegetation biomass, and improve soil physical and chemical properties (Mengistu et al. 2005; Mekuria et al. 2007; Shono 2007; Yirdaw et al. 2014).

For highly degraded dry forests active restoration approaches, such as multi-species planting (afforestation and reforestation), framework species, maximum diversity, and nurse tree methods may be more appropriate than passive restoration methods (Marques et al. 2016). Passive restoration methods have been shown to be ineffective in rehabilitating severely degraded sites (Laycock 1995). However, active restoration methods are costly, potentially risky and may require sufficient ecological knowledge (Lamb 2005; Marques et al. 2016), which seems to be lacking particularly in the tropical and subtropical dry forests (Blackie et al. 2014). In sites with a heavily degraded seed bank, enrichment planting of late-successional or rare species is necessary in order to speed up the recovery process (Mengistu et al. 2005; Marques et al. 2016). In active restoration of dry forests, native species and provenances which are adapted to local environmental conditions (particularly high water use efficiency and fire tolerance) should be favoured, whenever possible (Vallejo et al. 2012); ITTO (2002) offers a list of promising species that can be used for restoration of tropical dry forests.

In active restoration of degraded drylands the survival and early establishment (which is the most vulnerable developmental stage) of planted seedlings are among the major challenges faced by practitioners. Studies indicate that inoculation with mycorrhizal fungi in the nursery and the use of tree shelters were the most effective treatments for enhancing both the survival and early growth of planted seedlings in drylands (Piñero et al. 2013).

Because of the relatively slow growth rate of dry forests active restoration strategies may require long-term commitment to be successful (McIver and Starr 2001). On the other hand, although the growth rate and succession of dry forests is slower than moist forests, but because they are less complex floristically and structurally they can recover more quickly than moist forests (Kennard 2002; Vieria and Scariot 2006).

There are frequent occurrences of fire in dry forests. A high incidence of fire, through regressive succession can convert dry forests to more open formations and subsequently to savannas (Menaut et al. 1995; Viera and Scariot 2006). In general, grasses benefit from recurrent fires and out-compete tree seedlings (D'Antonio and Vitousek 1992; Viera and Scariot 2006). Frequent fires may also simplify the species composition of a site by eliminating fire-intolerant species and favouring fire-resistant species (Gillespie et al. 2000). On the other hand, protection of savannas from fire may initiate natural succession and turn them into dense woody formations derived from both seeds and vegetative sprouts (Swaine 1992; Menaut et al. 1995). Hence, the management of recurrent fires is essential in order to rehabilitate degraded dry forests, abandoned dryland farms and fire-maintained savannas. In general, the development of process-based models that allow manipulation of specific ecological processes and forecast rehabilitation outcomes (James et al. 2013) particularly the natural succession of dryland vegetation will be useful tools for researchers and practitioners.

Degradation of dry forests is often accompanied by depletion of soil fertility to support good growth of trees. Thus, site amendment measure would benefit successful rehabilitation of forest on severely degraded sites. A recent meta-analysis of studies on responses of woody plants to biochar addition reveals considerable increase in tree growth (Thomas and Gale 2015). In the context of forest restoration on degraded sites, Sovu et al. (2012) have also reported 1.2-fold increase in growth of trees planted on biochar-amended fallows.

5 Economic costs of dryland rehabilitation

Despite having rough estimations about the economic losses associated to dryland degradation, either globally (LADA 2013), regionally (PNUMA 2003, 2010; ELD Initiative and UNEP 2015) or even for specific countries (Kirby and Blyth 1987; Berry et al. 2003), it is surprisingly difficult to obtain information about the potential cost of avoiding degradation or the cost of restoring degraded drylands. In the lack of accurate information about the cost of rehabilitation, a common argument in favor of action is to add together the so-called 'damaged costs' or forgone revenues, including losses of products and services due to degradation, and approximated cost of rehabilitating a particular area. This will generate a large monetary value and the argument that any rehabilitation effort (including transaction costs) with a price tag below the previous total is worth to implement (Nkonya et al. 2016). As part of the cost-benefit analysis in environmental economics this is known as 'total economic value' (TEV), which basically compares the value of having and maintaining something (e.g. environmental services) against the missing value of not having it. The assumption is that the cost of an alternative (e.g. changing productive activity or moving to a non-degraded area) is too high to bear. This strategy requires the quantification of all benefits (use value + non-use value) and all the costs (direct costs + indirect costs). The estimation of use values and direct costs are relatively easy to compile while non-use values and indirect costs are more difficult to estimate as they usually involve values and costs without markets and that might occur far away from the degraded or restored site.

There are several economic tools to deal with some of the problems associated with TEV as well as international efforts to mainstream the valuation of ecosystem services which are useful for the economic evaluation of dryland rehabilitation actions. The Economics of Ecosystems and Biodiversity (TEEB) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) are two of the international initiatives working on improving the accuracy of economic valuations. During the Second Scientific Conference by the UNCCD two White Papers were prepared to review the economic and social implications of desertification and land degradation (Low 2013) as well as the costs and benefits of dealing with these issues (Poulsen 2013). Unfortunately, the information needed to assess the economic cost of rehabilitating drylands or even assess the total cost of dryland degradation is largely missing in the scientific literature (Low 2013; Nkonya et al. 2016). The best documented case came from Australia where the degradation of drylands due to soil salinization was driven by conversion of perennial native vegetation to agriculture and other land uses in southwest and southeast Australia.

All the aforementioned studies do not deal directly with economics aspects, but the different pieces of information offer a more complete view of the overall cost of inaction. Some of the relevant issues presented in the literature which could be used in a cost-benefit analysis include: the economic case for government intervention and the overall cost of dryland degradation (Kirby and Blyth 1987; Beresford 2001; Roberts and Pannell 2009; Graham et al. 2010), spatial modelling and identification of dryland areas affected by salinization (Graham 1992; Ive et al. 1992; Horwood 1994; Kirkby 1996; Furby et al. 2010), farmers and community perceptions about the salinization problem and the proposed alternatives (Greiner 1997; Hartley et al. 1998; Kington et al. 2003; Khan et al. 2008; Kingwell et al. 2008); landscape or river basin management options (Greiner 1998; Callow 2011, 2012), modelling of on-farm management alternatives and economic trade-offs (John et al. 2005; Cheng et al. 2009; Finlayson et al. 2010; Graham et al. 2010), reintroduction of native trees and shrubs (Schofield 1992; Dorrough and Moxham 2005; Thrall et al. 2005), and the effects of dryland degradation on human health (Jardine et al. 2007, 2008a, 2008b, 2011; Speldewinde et al. 2009, 2011).

The overall economic losses due to salinization are estimated at around 344 million AUD per year (Dept. of Agric. and Food) for southwest Australia and around 305 million AUD per year for southeast Australia (Wilson 2004). These estimations include agriculture, water and infrastructure losses, but exclude other environmental and social costs. It is still difficult to estimate the cost-benefit ratio of investments in control and rehabilitation. For example, some studies suggest that early action by the government is cost efficient (Kirby and Blyth 1987) while delaying action is more cost efficient for farmers if the effects of salinization are expected in the mid- or long-term (Graham et al. 2010); others suggest that funds need to be allocated to support communities to create awareness and that more has to be done to support the development of skills and innovative solutions (Kingwell et al. 2008). In the case of Australia, there have been financial commitments and several programs since the 1980's to control salinization and restore degraded drylands and many millions more have been invested by individual farmers with the same aims. Efforts have been made to ensure that control and restoration investments are cost efficient and programs are under regular monitoring (George et al. 2005; Sparks et al. 2006); nonetheless, the problems continue and the costs are expected to increase in the following decades.

Assessing the cost of dryland rehabilitation in other areas such as Africa is even more complicated. Most of the published research deals with the drivers (human induced and natural) behind dryland degradation, how dryland degradation affects local people, the role of communities as part of rehabilitation efforts, and climate change. Many of these articles include general suggestions about how to reduce degradation or how to enhance the rehabilitation of degraded drylands (grazing management -including exclusion-, reduced tillage, water harvesting, fires management, ethnographic studies, conservation agriculture, scenario planning, managing conflicts, etc.), sometimes under very specific settings (a village or set of villages, a particular livelihood activity, or even an specific crop). Seldom, monetary information about the cost of degradation or the benefits of rehabilitation is given; this is not surprising as much of Africa's drylands productivity is subsistence based and do not reach markets.

There is a lack of knowledge about livelihood dynamics in the tropical drylands (Blackie et al. 2014), and socioeconomic benefits have not been well linked to restoration efforts (Aronson et al. 2010). Restoration practitioners have failed at selling the idea of restoration as a 'worthwhile investment for society', despite the evidence in favor of taking action (ELD Initiative 2013; ELD Initiative and UNEP 2015; Nkonya et al. 2016). Fortunately, there has been a sizable effort in recent years to link ecological restoration with ecosystem services (Alexander et al. 2016; Turner et al. 2016), livelihoods (Reed et al. 2015) and the sustainable development goals (Nkonya et al. 2016). The Economics of Land Degradation Initiative (ELD), tries to bring together the dispersed knowledge available about the overall cost of land degradation and the overall benefits of land restoration in order to quantify the real TEV of land degradation action and inaction; the initiative also aims at closing the gap between economic estimations at the local level and macro-economic estimates, particularly in Africa.

6 The role of community participation in dryland rehabilitation

The rampant degradation of drylands calls for concerted rehabilitation efforts, involving several stakeholders, both governmental and non-governmental. All stakeholders should have clear division of tasks, rights, costs and benefits in order to avoid confusion and replication of efforts (Chokkalingam 2005). Particularly, local communities who are affected most by restoration projects should participate from project conceptualization to implementation and management (Marques et al. 2016). Their participation is crucial for the long-term success of a restoration endeavor

(Chokkalingam 2005). Furthermore, strengthening of local organizations will enable local people to implement and sustain rehabilitation activities. Rehabilitation operations should also consider local peoples' short and long-term needs and value systems in order to sustain their participation and interest (Sayer et al. 2004). A trade-off may be required to meet the needs of local communities and the ecological objectives. This implies that local socio-economic needs should be taken into account when choosing rehabilitation approaches. Cases from Kenya (Mganga et al. 2015), China (Dai 2010; Sjögersten et al. 2013), Ethiopia (Reubens et al. 2011; Yirdaw et al 2014), Jordan (Ajlouni et al. 2010), Tunisia (Visser et al. 2011), and Australia (Bell et al. 2001; Kingwell et al. 2008; Roberts and Pannell 2009) show that communities seek co-benefits in order to adapt and implement sustainable management activities or for the introduction of new technologies. Another policy-related factor that influences the success of dryland rehabilitation is well-defined land tenure and/or secure property rights for land and trees (Muys et al. 2006).

The contribution of Traditional Ecological Knowledge (TEK) in management and conservation of natural resources has been well recognized and utilized over the past few decades (Gadgil et al. 2003). TEK is defined as a "cumulative body of knowledge, practice, and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment" (Gadgil et al. 2003). TEK has been shown to play an important role in filling crucial gaps in our ecological understanding and can contribute in all aspects of ecological restoration (Uprety et al. 2012). The underlying premise for the role of TEK in restoration lies on the fact that local people often interact with a landscape over longer periods of time. Thus, TEK can provide valuable information relevant to restoration ecology, such as construction of the reference ecosystem, traditional land management practices, species selection for restoration planting, and monitoring and assessment of restoration outcomes, in less time and at a lower cost. For instance, traditional range exclosures have been widely practiced by pastoralists in East Africa for dry season grazing, which can be used as a rangeland restoration method to allow the herbaceous vegetation diversity to recover (Angassa et al. 2010). In Western Kenya, Ouma et al. (2016) have documented various beliefs, practices and norms applied by local communities to regulate use of Kakamega Forest. As a whole, the participation of local communities in dryland rehabilitation is essential to achieve the Aichi Biodiversity Target 18 on traditional knowledge, innovations and practices of indigenous and local communities and their customary use of biological resources (https://www.cbd.int/sp/targets/).

7 Conclusions and recommendations

Land degradation is rampant and a serious threat affecting the livelihoods of 1.5 billion people worldwide of which 250 million people reside in drylands. The extent of degradation is estimated at 12 million ha each year, which is expected to increase with projected increasing in human population inhabiting the drylands. Land degradation is driven by human activities, adverse climatic conditions (such as recurrent droughts) and population increase. Land degradation in drylands has already taken its toll in reducing provision of environmental services, food insecurity, social and political instability and reduction in the ecosystem's resilience to natural climate variability. Technically, the prospect of restoring degraded drylands is promising, as evidenced from successes achieved in restoring rangelands, croplands and dry forests using a suite of passive and active restoration measures. Soil degradation is a critical biophysical process affecting ecosystem functions and sustainability of all land uses. Soil reclamation in drylands in general and croplands in particular can be achieved using biological, chemical and physical measures. Recently, new and promising soil remediation techniques have emerged. The review also identified the data gap in cost-benefit analysis following restoration intervention. However, there is a general understanding that the cost of rehabilitation and sustainable management of drylands is lower than the losses that accrue from inaction, depending on the degree of degradation.

Most restoration efforts in drylands have concentrated on research scale, addressing the intricate issues pertaining to restoration separately. Thus, a landscape-level approach, which is an integrated and multidisciplinary approach, would be a promising tool to address the various and often contradictory environmental and societal needs. Addressing dryland rehabilitation at the landscape-level takes into considerations the biophysical and socioeconomic linkages and trade-offs existing between the different land uses and provides a comprehensive and long-lasting measures to reverse land degradation (Dudley et al. 2005; Lamb et al. 2005; Lamb 2014). With respect to dry forest restoration, the forest landscape restoration approach, which operates at a landscape-level, can be used as the overarching framework. The rehabilitation of degraded areas in Australia is a good example of an effort to incorporate different natural (e.g. hydrology, geomorphology, and weathering) and social (e.g. environmental change, agriculture, health, pollution) processes in order to understand and modify the drivers of degradation.

Above all, local communities' participation, incorporation of traditional ecological knowledge and practices, consideration of local peoples' short and long-term needs and value systems, clear division of tasks and benefits, strengthening of local organizations are crucial not only for cost-sharing, but also for the long-term success of dryland rehabilitation endeavors.

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