REVIEW ON REHABILITATION OF DEGRADED LANDS

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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, overgrazing 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 assessing the cause and direct and indirect impacts of land degradation and its rehabilitation and remediation technologies including bioremediation biocharing and others. 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 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: croplands; desertification; dry forests; land degradation; landscapes; rangelands; restoration

INTRODUCTION

Definition of Land Degradation

Different authors define land degradation differently. Some others also explain the difficulty to define it because of its wider range and scope (Dessalew, 2016). Land degradation can be defined as a natural process or a human activity that causes the land to be unable to provide intended services for an extended time (FAO, 2004) or temporary and/or permanent lowering of the

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productive capacity of land that can take place in the form of deforestation, change in water quality and quantity, and soil degradation (Sil et al., 2014). It also refers to any reduction or loss in the biological or economic productive capacity of the land caused by human activities, exacerbated by natural processes, and often magnified by the impacts of climate change and biodiversity loss (UNCCD, 2013). Land degradation is a long-term loss of ecosystem function and services, caused by disturbances from which the system cannot recover unaided (Dogo, 2014). In this report, land degradation is defined as a negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans.

Millions of hectares of land per year are being degraded in all climatic regions of the world (Assemu & Shigdaf, 2014). Globally, land degradation has been hitting more than 2.6 billion people in more than 100 countries (GEF, 2013). It is increasing in severity and extent in many parts of the world, with more than 20% of all cultivated areas, 30% of forests and 10% of grasslands undergoing degradation (Bai et al., 2008). Land degradation processes have accelerated rapidly in the last century, with an estimated 24 billion tons of fertile soil lost to erosion in the world‘s croplands (FAO, 2011). Global assessments indicate that the percentage of total land area that is highly degraded has increased from 15% in 1991 to 25% by 2011 (UNCCD, 2013). In line with this, (FAO, 2011) indicated that up to 25% of all land is currently highly degraded, 36% is slightly or moderately degraded but in stable condition, while only 10% is improving. IFPRI (2012) report shows that if the current scenario of land degradation continues over the next 25 years, it may reduce global food production by 12% and in turn, this will result in world food prices of 30% higher for some commodities.

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+).

Land degradation is both a natural and human-induced process, which diminish the capacity of land resources to perform essential functions and services. Land degradation is a globalscale, ongoing, and the relentless problem that poses a major long-term challenge to humans in terms of its adverse impact on biomass productivity, food security, biodiversity and environmental sustainability. Land degradation is the broad term that includes soil erosion degradation (acidification, fertility depletion, hard setting), biological degradation (reduction of total biomass and carbon and decline of biodiversity) and ground water depletion (Melkamu et al, 2020).In Ethiopia, land degradation has become a serious problem affecting all spheres of social, economic and political life of the population (Melkamu et al, 2020). It is one of the major challenges to agricultural development and food security of the country. A large portion of the agricultural land, which is mainly located in the highland part of the country, is affected by severe to moderate land degradation (Hurn et al, 2010).

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Combating of land degradation is crucial to ensure the long-term productivity of semi-arid environments. In any environmental policy, an understanding of potential tradeoffs and interactions between carbon sequestration and biodiversity is essential (Reside et al. 2017). Successful restoration includes improvement of land productivity, soil organic C stock, biodiversity and other ecosystem services (Mekuria et al. 2017). About three decades ago, EXs was initiated as one of the restoration practices to avert the degraded semi-arid lands of Ethiopia (Yayneshet 2011). So far, several studies have been conducted in the semi-arid area focused on species composition, soil properties and socio-economic importance of EXs (Yami et al. 2013). The significance for both carbon emission reduction and biodiversity conservation strategy in the arid and semi-arid land of the tropics. Moreover, the existing sole studied on the effectiveness of established EXs to improve soil carbon stock potential was in consistent and variable (Aynekulu et al. 2017).

Artificial Soil Stabilization Techniques

Artificial stabilization of the soil surface has successfully resulted in BSC rehabilitation. There are three primary variations: polyacrilimide application, coarse litter application (such as straw), and use of stabilizing vascular plants. Polyacrilimide application was shown to have either no effect or, in combination with other treatments, a negative effect on chlorophyll fluorescence or nitrogenase activity of transplanted Collema (Collemataceae) lichens (Davidson et al. 2002).

Engineering remediation

Engineering remediation refers to using physical or chemical methods to control heavy metal contamination of soils.

Replacement of contaminated soil, soil removal and soil isolation

Replacement of contaminated soil means adding large amount of clean soil to cover on the surface of the contaminated soil or to blend with the latter. Soil removal refers to remove the contaminated soil and renew it with the clean soil, which is necessary for the seriously contaminated soil with little area. Soil isolation means that to isolate the contaminated soil from the uncontaminated soil, but to completely remedy it still needs other auxiliary engineering measures (Zheng et al., 2002). All of these methods will cost large amount of manpower and material resources, so they can only be applied to small area of soils.

Adsorption

Adsorption method is based on the fact that almost all heavy metal ions can be fixed and adsorbed by clay mineral (bentonite, zeolite, etc.), asteel slag, furnace slag, etc (Wang and Zhou, 2004).

Terracing

Constructing physical soil and water conservation measures, such as terraces, trenches, and soil and stone bunds, can support the regeneration of native plant species. These can improve the quality and quantity of livestock feed that can be harvested from the exclosure. This, in turn,

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improves livestock production and increases the benefits obtained by local communities (Daniel 2020).

Terraces are constructed for soil and water conservation purposes on sloppy areas in Ethiopia. Besides soil and water conservation, terraces are also used for the control of termites in western Wallaga. Nowadays, farmers in western Wallaga highly appreciate the role of terraces in controlling termites and agricultural and rural development offices also encourage farmers to practice the method. The water stored in the terraces during rainy seasons enters termite tunnels and reaches the nests, suffocates and kills termites found in the nests and in the tunnels. The effect of terracing on termites can be evidenced as a large number of dead termites are seen floating on the surface of water in the terraces Mulatu and Emana (2016).

Chemical process of land degradation and rehabilitation measures

Soil organic carbon stock

Soil organic carbon stocks (Mg C ha/$^{-1}$) within 0–30 cm soil layer was significantly different between the EXs and adjacent DOGL ($p < 0.05$) (Table 6). The top layer (0–15 cm) contributed 57% of the total soil organic C stock for the EXs and 55% for DOGL. As expected, EXs with enrichment by selected tree species enhanced the soil organic carbon stocks by 38% over the adjacent open grazing land. The impact of EXs on SOC was higher in the surface layer (31%) than subsurface soil layer (24%) over DOGL.

Biochar for land rehabilitation and improvement

The United Nations Convention to Combat Desertification (UNCCD) supports biochar as a means for combating land degradation, improving farmland and combating climate change. There are two ways in which biochar use can counter land degradation: 1) biochar use to make farming more sustainable and productive with less harmful pollution. This would reduce pressure for clearing new land (Abend, 2008, 4 pp.). 2). Biochar use to rehabilitate degraded or naturally poor land e also helping to reduce pressures for clearing new areas. One vital question is 'how will degraded or desertified regions produce sufficient biochar to effectively support land rehabilitation when there is only sparse plant cover to use for feedstock?' One solution would be to grow hardy vegetation on unfarmed biodiversity-poor areas, perhaps irrigated with water too poor to be used for food crops. Other possibilities are algae, reeds, grasses or aquatic weeds grown in lagoons using poor quality water and effluents. Waste from towns and agricultural waste might also be used if transport were cost effective. Strezov et al. (2008) suggest un-irrigated elephant grass grown on degraded (non-agricultural) land could produce biofuel and biochar even where growth rates are less than 40 metric tons of dry biomass per hectare per annum.

Land ruined by salts or soda, might produce biomass for biochar production and the removal of the contaminants with the biomass could help rehabilitate it as well as providing material to improve soils elsewhere. This has been explored in Australia (Bartle, Olsen, Cooper, & Hobbs, 2007), where dry, salty land yields coppiced eucalyptus feedstock and in Sumatra where forestry and paper pulp waste has been used (Ogawa, Okimori, & Takahashi, 2006). Urban brown field sites could be rehabilitated with biochar to support amenity planting or biofuel and timber production and at the same time lock-up carbon. Linked to refuse treatment biochar could have much potential. Modern urban refuse in developed countries (and increasingly in developing nations) is likely to be contaminated with toxic materials and so may not make good compost; however, some studies suggest uncontaminated biochar may be produced with it.

The contribution of biochar to mitigating global warming

Biochar production does emit carbon dioxide and other greenhouse gases but combined with waste disposal or biofuel production it appears to offer a practical way to mitigate global warming. Understandably biochar potential is attracting much attention as a safe, practical, technically simple, and affordable methods of sequestration, which has a chance of spreading fast enough to have real effect. If enough farmers, larger agricultural enterprises, biofuel producers, and waste treatment plants a are established it could become an important means of carbon sequestration. This potential is a little better researched than biochar agricultural value; although, there is insufficient data on biochar-burial soil carbon mean residence times (Barrow, 2011).

Remediation of Heavy Metal Contaminated Soils

Fertilizers, pesticides and mulch are important agricultural inputs for agricultural production (Zhang et al., 2011). Nevertheless, the long-term excessive application has resulted in the heavy metal contamination of soils. The vast majority of pesticides are organic compounds, and a few are organic - inorganic compound or pure mineral, and some pesticides contain Hg, As, Cu, Zn and other heavy metals (Arao et al., 2010).

Heavy metals are the most reported pollutants in fertilizers. Heavy metal content is relatively low in nitrogen and potash fertilizers, while phosphoric fertilizers usually contain considerable toxic heavy metals. Heavy metals in the compound fertilizers are mainly from master materials and manufacturing processes. The content of heavy metals in fertilizers is generally as follows: phosphoric fertilizer> compound fertilizer> potash fertilizer> nitrogen fertilizer (Boyd, 2010). Cd is an important heavy metal contaminant in the soil. Cd is brought to soils with the application of phosphoric fertilizers. Many studies showed that, with the application of a large amount of phosphate fertilizers and compound fertilizers, the available content of Cd in soils increases constantly, and Cd taken by plants increases accordingly. In recent years, the mulch has been promoted and used in large areas, which results in white pollution of soils, because the heat stabilizers, which contain Cd and Pb, are always added in the production process of mulch. This increases heavy metal contamination of soils (Satarug et al., 2003).

Electrokinetic remediation

Soil electrokinetic remediation is a new economically effective technology. The report by Hanson et al., (1992) indicated that, the DC-voltage is applied to form the electric field gradient on both sides of the electrolytic tank which contains the contaminated soil; contaminants in the soil is taken to the processing chamber, which is located at the two polar sides of electrolytic cell, through the way of electro-migration, electric seepage or electrophoresis, and thus reduce the contamination (LiQin, et al. 2014).

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Phytoremediation Grow specific plants in the soil contaminated by heavy metals. These plants have the certain hyper-accumulation ability for the contaminants in the soil (accumulated mainly in the root or above the root). When the plants are ripe or reach certain enrichment level of heavy metals, remove heavy metals in the contaminated soil layer thoroughly by harvesting, burning and curing plants. Using plants and their coexisting microbial system to remove heavy metals is a new technology. The key of the method is to find the suitable plants with strong ability for heavy metal accumulation and tolerance. Now more than 400 species of such plants have been found in the world, and most of them belong to Cruciferae, including the genus Brassica, Alyssums, and Thlaspi (Xing et al., 2003).

Microbial remediation

Microbial remediation refers to using some microorganisms to perform the absorption, precipitation, oxidation and reduction of heavy metals in the soil. Siegel et al. (1986) found that fungi could secrete amino acids, organic acids and other metabolites to dissolve heavy metals and the mineral containing heavy metals. Fred et al. (2001) reported that the fungi, Gomus intraradices, may improve the tolerance and absorption of sunflower to Cr.

Animal remediation

Some animals living in the soil (maggots, earthworms, etc.) can take heavy metals in the soil. Wang et al. (2007) proved that when the concentration of Cu was low in the soil, the activities and secretion of earthworms could promote the absorption of Cu by ryegrass.

Rehabilitative effects of reforestation on soil microbial activity: Comparisons between the *Acacia* plantation soil and the other soils show the restorative effects of *Acacia* planting (Doi and Ranamukhaarachchi 2007). The microbial function has also been restored, as shown by the fact that there were no significant differences in soil dehydrogenase activity between the *Acacia* plantation and the evergreen forest soils. The Acacia trees had been growing for 18 or 19 years, during which time many native species have returned to the plantation plots (Kamo *et al*. 2002). Thus, planting of *A. auriculiformis* and the return of native species may restore the original soil microbial activity. Increasing the diversity of plant community can help enrich soil fertility (El-Keblawy and Ksiksi 2005) and establish the soils ecological structure (Beare *et al*. 1995). In an adjacent paddy field, the soil fertility, expressed as a soil fertility index (Moran *et al*. 2000), was comparable to the bare ground soil, and the value of dehydrogenase activity was poor (1.81 ±0.74 µmol formazan/h/g dry soil [n=40]). This shows the difficulty of restoring microbial activity when the land is burdened with agricultural production.

Soil dehydrogenase as a single measure of land degradation and rehabilitation: In this study, changes in soil dehydrogenase activity correlated very well with the land degradation/ rehabilitation. The dehydrogenase assay offers a continuous measure of soil microbial activity as a result of the total redox sequences (Smith *et al*. 1983). Moreover, various soil microbes have redox sequences in their cells. The dehydrogenase assay is superior to techniques that involve threshold-manner observation such as most-probable-number counting of a target soil microbial group (e.g., Soares *et al*. 2006).

Thus, it appears that the assay of soil dehydrogenase activity can serve as an integrative measure of soil quality. A soil ecosystem includes many microbes that can reduce tetrazolium compounds (Sollod *et al*. 1992) Dehydrogenase isozymes of a microbial species respond to different environmental impacts in different ways (Berchet *et al*. 2000), while the formation of formazan can be measured as a single variable. This value has a highly significant correlation with the first principal component, suggesting its integrative nature; i.e., formazan formation represents various redox reactions. This study shows that soil dehydrogenase activity is a useful single criterion for measuring the status of land degradation and the rehabilitation in the field. This technique will minimize labor, expense and time spent for monitoring soil quality in land rehabilitation efforts in savanna regions.

Biological process of land degradation and rehabilitation measures

Arbuscular mycorrhizal fungi and the Mycorrhizosphere Ecology

The rhizosphere may be a narrow zone of soil suffering from the presence of plant roots (Hrynkiewicz and Baum, 2011). It is extremely important and active area for root activity and metabolism (Saharan and Nehra, 2011). Roots release a mess of organic compounds (e.g., exudates and mucilage) derived from photosynthesis and other plant processes making the rhizosphere a hot spot of microbial activities mainly that of fungi and bacteria (Hrynkiewicz and Baum, 2011). The physical, chemical and biological environment of the rhizosphere is hence, clearly distinct from the majority soil (Barea et al., 2002).

Similarly, the rhizosphere of the mycorrhizal plant are often mentioned because the mycorrhizosphere (Barea et al., 2002). Mycorrhizosphere comprises both the basis and hyphae influence zones or the rhizosphere and hyphosphere (Timonen and Marschner, 2006). Mycorrhizal hyphal growth in soils is extensive, with mycelial lengths reaching 111 m cm-3 or 0.5 mg g-1 or 900 kg ha-1 of soil (Simard and Austin, 2010). Hence, the mycorrhizosphere provide a critical link between plants, other microorganisms and therefore the soil (Hrynkiewicz and Baum, 2011).

AMF Improve Soil Aggregation; Hence Increase Soil Organic Matter and Soil Water Relation

Fungi and most significantly AMF could also be the foremost effective soil organisms in stabilizing soil structure (Augé, 2004). AMF hyphae grow into the soil matrix to make the structure that holds primary soil particles together to make soil aggregates (Augé, 2004; Al-Karaki, 2013). AMF also improve soil aggregation by influencing bacterial communities, which will improve soil aggregate formation (Rilling, 2004). Furthermore, the dead AMF hyphae produce glomalin, which is hydrophobic stable aggregate former (Barea et al., 2002; Simard and Austin, 2010). Hence, AMF increase both soil aggregation and stability. AMF may stabilize soils up to five months after their host’s death (Soka and Ritchie, 2014).

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Meanwhile, as a results of the many amount of mycorrhiza derived soil carbon (Rilling, 2004) and improved soil aggregation and stability, AMF increase soil organic matter content and stability (Rilling, 2004; Leifheit et al., 2014). Improved soil aggregation also increases soil water relation.

AMF Improve Plant Nutrition and Nutrients Cycling Index

The most important role of AMF is their role in phosphorous nutrition (Skujins and Allen, 1986). There are also data indicating that AMF can transfer nitrogen from one plant to another (e.g., Requena et al., 2001), increase the utilization of different forms of nitrogen by plants and may also take up nitrogen directly and transfer it to host roots (Govindarajulu et al., 2005). However, there's considerable doubt on the cost-benefit of AMF in plant N nutrition (Smith and Smith, 2011). Although few data exist, AMF were observed to enhance potassium nutrition in plants (Dag et al., 2009; Garcia and Zimmermann, 2014). AMF also can increase the uptake of other macro and micro nutrients by plants (Birhane et al., 2012). Generally, the external mycelium of AMF establishes an underground network that links the various plants and hence sequester carbon, nitrogen, and phosphorous and also allow the transfer of those nutrients among plants (Rodriguez-Echeverria et al., 2007). These important roles of AMF therefore play great role in nutrient cycling where the necessity for further nutrient inputs is significantly reduced (Gianinazzi et al., 2010; Al-Karaki, 2013).

Arbuscular mycorrhizal fungi not only improve nutrient cycling but also reduce nutrient leaching from the soil (Rodriguez-Echeverria et al., 2007). In a comprehensive assessment done by Bender et al. (2015), it was possible to determine the role AMF have in nutrient cycling and leaching. Accordingly, it had been determined that while AMF inoculation increased nutrient uptake by plants it also reduced leaching of dissolved organic N and un-reactive P (Bender et al., 2015).

AMF Increase Tree/Shrub Seedlings Growth, Productivity, Field Survival and Establishment on Degraded Lands

Lekberg and Koide (2005) administered a meta-analysis supported 290 published experiments to work out the role of AMF on plant growth and productivity. The analysis also determined the consequences of three common AMF management methods; inoculation, short fallow, and reduced soil disturbance. The result of the meta-analysis revealed that AMF generally increase individual plant's growth and productivity. Inoculation and short fallow resulted in significantly positive effects on plants' growth and productivity (Lekberg and Koide, 2005). A recent meta-analysis on 304 papers also concluded that AMF inoculation increases the expansion and productivity of plants grown alone (Lin et al., 2015). A similar result was also reported by (Birhane et al. 2014). Huante et al. (2012) also did experiment on six tree species and located out that AMF inoculation has significant effect on seedlings growth and most importantly slow growing tree species.

AMF Biotechnology for the Restoration of Degraded Lands

Arbuscular mycorrhizal fungi show no host specificity to forge symbiotic relationship with plants and are very ubiquitous, found almost in every soil (Abbott and Robson, 1991; Brundrett and Abbott, 2002; Barea et al., 2011; Al-Karaki, 2013). Hence, many researchers argue that AMF inoculation is likely to be valuable in only few conditions such as mine fields where indigenous

AMF inoculum is surely little or none available (Brundrett and Abbott, 2002). Koide and Mosse (2004) suggested instead of going for AMF inoculation it would be quite economical and appropriate to focus on managing the indigenous AMF population of a site. According to Renker et al. (2004), inoculation is an important but the last option. However, contrary to having several dispersal agents such as; wind, water, rodents, birds, worms, and ants (Brundrett and Abbott, 2002), AMF were observed to have poor dispersal. Accordingly, Hailemariam et al. (2013) were able to observe that within a single piece of farm land, soil AMF status and infectiveness can vary in short distances indicating poor dispersal. Similarly, Friese and Allen (1991), also indicated that AMF have poor dispersal. Therefore, to overcome the dispersal limitation of AMF, inoculation may be a worthily intervention.

Meanwhile, AMF inoculation has proved to be effective under wide range of soil conditions (Janos, 1980; Brundrett and Abbott, 2002) including on soils with good AMF abundance (e.g., Banerjee et al., 2013). Positive AMF effect is not ensured by the presence of abundant indigenous AMF but by both abundance (quantity) and efficiency (quality) of indigenous fungal populations (Onguene and Kuyper, 2005). Veiga et al. (2011) also demonstrated that AMF inoculation suppressed weeds and, interestingly enough, hypothesized that AMF inoculation could suppress ruderal plants which are known to invade degraded sites (Veiga et al., 2011). This is particularly important in ecological restoration since ruderal plants could invade degraded lands and compete with tree/shrub seedlings planted.

Mine land rehabilitation

Species selection: from scoring single species to community-wide approaches

Revegetation is intended to encourage the development of mine soil, produce pleasing landscapes and increase ecosystem productivity (Skousen, 2010) to increase the sustainability of mining operations. As the rapid growth and closure of vegetation are necessary for erosion control and site stabilization, fast-growing plant species are required from a technical point of view. Propagules for revegetation activities must be adapted to the environmental constraints of the degraded habitats. Furthermore, they should be compatible with the desired post-mining land-use and restoration targets (Skousen, 2010). Plants able to establish symbioses with mycorrhizal fungi and rhizobia (Anderson, 2008; Siqueira et al., 2007; Sprent, 2009) or attractors of pollinators (Garibaldi et al., 2014) are necessary to establish trophic networks and achieve dynamic, process-based restoration goals (Perring et al., 2015). The application of arbuscular mycorrhizal fungi may enhance revegetation and rehabilitation (Cicatelli et al., 2010; SolísDomínguez et al., 2011). Additionally, Giannini et al. (2016) highlight propagation criteria, interactions and services as well as natural geographic distribution to score the restoration potential of different species.

Substrates from waste piles and mine pits are generally acidic and depleted in organic matter, nutrients and soil organisms and, depending on the mined ore or metal, bear high concentrations of available (heavy) metal ions (Candeias et al., 2014; Ettler et al., 2014; Pourret et al., 2016). Therefore, species from metalliferous ecosystems are expected to perform better in mining environments (e.g., Stradic et al., 2014), whereas species from further ecosystems may decelerate mine land rehabilitation by reduced germination, survival or reproduction rates (Kirmer et al., 2012; Leps et al., 2007). In particular, metallophytes or metal hyperaccumulators from metalliferous ecosystems might do better than species originating from ecosystems not adapted to high metal concentrations (Komal et al., 2015).

By offering refuge from physical stress, predation and competition or improved resource availability (Michalet and Pugnaire, 2016; Stachowicz, 2001), facilitation, i.e., mutualistic or commensalistic interactions between a nurse and a facilitated individual, may play important roles in recovering ecosystem structure and functioning (Gomez-Aparicio et al., 2005; Ren et al., 2008; Zwiener et al., 2014). Therefore, facilitation is discussed as a driving force of ecological succession (Padilla and Pugnaire, 2006), as nurse plants may facilitate the arrival and establishment of further individuals during revegetation by providing food and shelter for seeddispersing fauna (Giannini et al., 2016). As facilitation expands species' niches (Carrion et al., 2017), nurse plants present the potential to mitigate environmental degradation by mining in the short term as well as the effects of climate changes in the long term (Bulleri et al., 2016). Thus, the unambiguous identification of nurse species and their propagation contributes to greater effectiveness during the restoration processes.

While the reclamation of mine land for agricultural purposes prioritizes the rehabilitation of soil properties to satisfy the resource requirements of the desired crops, the straightforward restoration of self-perpetuating ecosystems that can perform their functions and provide services requires the selection of species that trigger ecosystem development. The species selection criteria depend on the type of ecosystem to be restored, which highlights the importance of defining restoration goals, including detailed specifications for target ecosystems.

Forests or other wildlife habitats should be considered when permanently restoring defunct mine lands, such as waste piles. For the temporal revegetation of active mine pits, the plantation of trees is not recommended, as they may destabilize the scene by falling. In such environments, grasses with fibrous root systems to stabilize soil, large biomass production and adaptations to regrowth after mowing or grazing as well as leguminous or nonleguminous forbs should be preferred. Some legume species can fix atmospheric nitrogen and transfer it to other plants, especially in areas without top soil application (Priest et al., 2016) Legume seeds should be inoculated when native rhizobia are not present in the soil (Giller et al., 2016; Gourion et al., 2015). Furthermore, the application of arbuscular mycorrhizal fungi may enhance revegetation success, especially in heavy metal-contaminated soils (Rodríguez-Caballero et al., 2017).

Because knowledge about functional attributes of native species from megadiverse tropical ecosystems is limited (e.g., Fernandes et al., 2016; Gibson et al., 2012), more simplistic approaches have been proposed for environmental rehabilitation in the past. For the restoration of forest ecosystems, available species are categorized as pioneers and non-pioneer species. Pioneer species are smallseeded, wind-dispersed, fast-growing and light-demanding species commonly observed during initial stages of ecological succession, while non-pioneers are shade-tolerant, medium to large seedsized, slow-growing species (e.g., Gandolfi and Rodrigues, 2009). Although the underlying mechanisms of ecological succession are not fully understood (Boukili and Chazdon,

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2017; Dini-Andreotea et al., 2015), seedling mixtures containing 50% pioneers and 50% non-pioneers arranged in rows or as quincunxes are recommended to reproduce succession during rehabilitation (Soares et al., 2016).

Biological invasions: enduring control through biological interactions

Alien invasive species are plants, fungi, or animals not native to a specific location, i.e., introduced species, which tend to spread to a degree causing damage to biodiversity, ecosystems, economics or human health (Simberloff et al., 2013). Invasion science quantifies and categorizes impacts (Elton, 1958), which range from shifts in community composition and dynamics (e.g., Hendrix et al., 2008; Loo et al., 2009) to multiple effects on ecosystem structure and functions due to alterations of biogeochemical pools and fluxes of matter and energy (Ehrenfeld, 2010; Simberloff, 2015; Vitousek et al., 1997). Biological invasions can drive native flora and fauna to extinction through competition due to more rapid resource acquisition, predation or hybridization (Rhymer and Simberloff, 1996; Bardgett and Wardle, 2010). Increases in stand biomass accelerate flux rates, alter disturbance regimes, especially fire, and impact entire food webs (Abernathy et al., 2016; Caplan et al., 2015; Martina et al., 2014; Rundel et al., 2014; Taylor et al., 2017). Invasion impacts include above- and belowground transformations (Wardle et al., 2004) able to alter selection pressures in macro- and microevolution by changing ecosystem and geomorphic processes (Fei et al., 2014).

Invasion success depends on propagule pressure and on abiotic and biotic characteristics (Colautti and MacIsaac, 2004). Propagule pressure includes dispersal and geographical constraints (Catford et al., 2009). Augmented propagule pressures increase genetic diversity, the probability of introduction to a favorable environment and the possibility of buffering temporarily unfavorable conditions (Lockwood et al., 2005). Abiotic characteristics are key to the availability of resources and conditions to permit or to impede the survival, growth and reproduction of invasive species, i.e., environmental and habitat constraints (Catford et al., 2014), while biotic characteristics include specific community dynamics and assembly as well as positive or negative interactions between invasive and native species. Enemy release (Liu and Stiling, 2006; Zheng et al., 2015), biotic resistance (Parker and Hay, 2005), biotic containment (Levine et al., 2004) and interspecific competition (Burke and Grime, 1996) reduce the invasibility of native communities or ecosystems. Thus, phylogenetic and functional approaches investigating assembly and dynamics in natural and invaded ecosystems are promising tools to advance knowledge about invasion processes (Cadotte et al., 2010; Kennedy et al., 2002; Ramos et al., 2015).

Monitoring revegetation success: the need for interdisciplinary approaches

Monitoring of revegetation projects is intended to measure the success of restoration projects (McDonald et al., 2016; Suding, 2011) and is furthermore a legal requirement in many countries (e.g., IBAMA, 2011; ICMBio, 2014; Barry, 1980). In the interest of more effective monitoring, the comparison of reference systems, i.e., natural control areas, and areas in environmental rehabilitation (ideally along so-called chronosequences allows the evaluation of restoration success (e.g., Almeida and Sanchez, 2005; Reis, 2008; Gandolfi and Rodrigues, 2009). Although response ratios are larger when comparing restored and degraded areas than the comparison of

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restored and reference areas (Benayas et al., 2009), the latter is necessary to identify deviations from restoration goals (Vickers et al., 2012). For further improvement of restoration practices, monitoring based on systematic surveys followed by statistical analysis provides high-value feedback to improve the entire restoration process, including the inference in legal requirements, i.e., the emergence of non-analogous ecosystems (Cadotte et al., 2010; Derhe et al., 2016).

Monitoring should measure the implementation success of all restoration goals. A variety of indicators ranging from structural parameters to diversity indices have been proposed for such monitoring (e.g., Reis, 2008), but no consensus has yet been achieved on which indicators should be measured as well as when and how they should be assessed (Kollmann et al., 2016; Ruiz-Jaen and Aide, 2005).

Termite management options

Termites are essential members of the soil ecosystem and are found throughout the world (Abdel and Skai, 2011). They are the most important fauna in nutrient recycling, improving soil fertility and serving as food sources for other animals. On the other hand, termites are highly destructive and polyphagous pests of crop plants, which damage green foliage, seedlings, forests, pasture, wooden structures, fibers including household cellulose-based materials, and postharvest stored products (Upadhyay 2013). Damage inflicted by termites impedes food security and threatens livelihoods of smallholder farmers (Demissie et al., 2019). Physical, biological and chemical controls of termites have been practiced for decades in western ONRS; unfortunately, none of these practices have been successful in reducing crop damage by termites (Negassa and Sileshi, 2018).

According to Daniel, the complexity of the termite problem justifies the use of various management options in an integrated manner, as no single control method is sufficiently effective against this pest. In the past, termite control campaigns including mound destruction and killing of termite queens were practiced in western Ethiopia. However, these control measures did not effectively reduce the damage due to lack of a holistic approach. Moreover, emphasis was given to the use of chemical insecticides, mainly of chlorinated hydrocarbons, which have detrimental environmental impacts and have since been banned due to their persistent toxicity and effects on non-target beneficial organisms and the environment (Demissie et al., 2019).

Abdurahman et al. (2010) have reviewed a wide range of termite management options practiced in Ethiopia. Accordingly, termite control measures practiced by farmers in western Ethiopia include traditional methods such as flooding mounds, digging mounds and removal of the queen or excavating the top parts of the mounds and burning straw to suffocate and kill the colony, placing the produce of different crops on wooden beds (protects harvested crops raised few centimeters above the ground from termite damage), mound poisoning and seed treatment. Gebreslasie and Meressa, 2018) reported that the application of crop residue and cattle manure reduced the number of termites on crop fields by 21.6 and 29.7% compared to non-treated fields. Demissie et al. (2019) also reported that the application of maize stover as mulch combined with animal manure, maize-soybean intercrops and wood ash, maize-desmodium intercrops, and mulching combined with maize-soybean intercrops consistently reduced termite damage to maize in field experiments conducted in Sibu Sire and GudeyaBilla districts of East Wollega Zone of Oromia National Regional State. The authors claimed that the results were comparable to the standard check, Diazinon 60% EC. In addition, intercropping maize with desmodium, mulching + intercropping maize with soybean, and Diazinon 60% EC reduced the number of lodged plants per plot compared to control (Daniel, 2020).

Planting termite resistant plants

Chomo grass
Chomo grass, Brachiaria humidicola (Rendle) Schweick, is native to Africa, from South Sudan and Ethiopia in the north to South Africa and Namibia in the south. It is strongly stoloniferous and rhizomatous perennial grass, forming a dense ground cover. It grows on a very wide range of soil types from very acid-infertile (pH 3.5), to heavy cracking clays, to high pH coralline sands. It can be sown for permanent pasture for grazing and as ground cover for control of erosion and weeds and good nematode control. Chomo grass is a termite-resistant fodder grass and potential grass species to rehabilitate degraded land. It can also be used for rehabilitation of degraded land in humid, acidic soil and termite infested fields. It can maintain good ground cover under heavy grazing and thus can support high stocking rates (Adie and Duncan, 2013). Chomo grass is one such grass species expected to reclaim the degraded land because of its creeping nature that anchors its lower nodes to the ground and its resistance to intermittent attack by termites (Dereje et al., 2014).

According to a report by Dereje et al. (2014) some volunteer farmers in some villages of Diga district (East Wallaga zone), were increasingly engaged in producing fodder crops such as Rhodes grass (Chloris gayana), Napier grass (Pennisetum purpureum) and Chomo grass (Brachiaria humidicola) on their fields. In addition to their major significance as animal feed, these perennial grasses serve to rehabilitate the degraded soil which has developed as a result of improper land management and termite infestation. The grass is thought to compete well with the other two species in terms of tolerance to termites and for simplicity of its multiplication through its tillering habit (Daniel, 2020).

Vetiver grass
Vetiver grass, Vetiveria zizanioides (L.) Nash, is a densely tufted bunch perennial plant native to India which can be easily established in both tropics and temperate regions of the world. The grass plays a vital role in watershed protection by slowing down and spreading runoff harmlessly on the farmland, recharging ground water, reducing siltation of drainage systems and water bodies, reducing agro-chemicals loading into water bodies and for rehabilitation of degraded soils. Vetiver grass could tolerate extremely high levels of heavy metals (Oshunsanya and Aliku, 2017).

Vetiver seems to tolerate a remarkable array of soil types and there are well-documented examples where vetiver is growing in very adverse soils, like on coastal sand dunes in South Africa, extremely acid soils (with pH as low as 4.0) in Louisiana, highly alkaline soils (pH up to 11) at

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Lucknow in India, black cotton clays (which heave and split and eject most plants) in Central India, barren soils with little fertility or organic matter in South China and other places, waterlogged soils in the black cotton clays, and even swamps, of India, parched land, in the arid state of Rajasthan of India, and Saline soils in Australia as (National Research Council, 1993). reported by (Daniel 2020). Thus, vetiver can be easily grown in the acid soil of Wallaga zones.

The accumulation of experiences worldwide is convincing that vetiver hedges can indeed block the passage of soil, can keep topsoil on site and, over time, retard most surface erosion. In many cases they can also help fill up gullies. Contour hedges of vetiver can slow down and hold back moisture that would otherwise rush off and be lost to the slopes and this ability to hold moisture on the slopes, and thus increase infiltration (National Research Council, 1993).The improvement of agricultural crop yield is one of the resultant benefits of the effects of vetiver grass technology on soil and water conservation. This could be beneficial to farmers, especially those farming on sloppy lands that are usually prone to erosion. Vetiver grass improves crop harvest by reducing crop failure against the dry spell (Oshunsanya and Aliku, 2017). The very long extensive root system of vetiver grass holds soil particles together in place and the aerial part which grows in profuse reduces runoff water and control erosion increasing water infiltration (Daniel, 2020). Besides its other uses and rehabilitation of termite degraded lands, vetiver grass can be used as biological pest control (Oshunsanya and Aliku, 2017).

Vetiver grass is also used as fodder for animal feed, mulch for improving soil moisture and fertility, and fibrous root system for holding soils in place could guarantee food production on a sustainable basis owing to the fact that this grass can withstand adverse environmental and climatic conditions, coupled with quick regeneration after pruning. Thus, when vetiver grass is applied appropriately, it could be a low-cost, simple and easily applicable multi-purpose soil and water conservation tool for sustainable agriculture. It is also a grass of great utility that could provide other means of revenue for local farmers (Oshunsanya and Aliku, 2017).

Planting termite resistant trees

Not all trees are equally susceptible to termites and thus some are more resistant to termite attack. A German citizen named Cruze residing in Chalia town in Guliso district, planted termite resistant coniferous trees on completely termite degraded lands at different sites in Ayira district; the trees are now well established and converted to forest which is used in checking water erosion and important in soil conservation. Besides, the forest has become a habitat for different animals and thus it is important for their conservation; when harvested the termite and decay resistant trees are sold at high price and thus are used as source of income for the community as well as for the project which runs it (Daniel, 2020).

Rehabilitation of degraded dryland ecosystems

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 (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

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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 (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 (Yirdaw et al. 2014).

Rangeland rehabilitation

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.

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.

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

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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.

Rehabilitation of degraded croplands

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.

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

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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).

Rehabilitation of degraded dry forest landscapes

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). In general, the development of process-based models that allow manipulation of specific ecological processes and forecast rehabilitation outcomes (James et al. 2013)

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

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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.

Causes and Impacts of land degradation

Scholars identified different causes of land degradation. For example, according to Berry (2003), the cause of land degradation involves two interlocking complex systems: the natural ecosystem and the human social system. Interactions between the two systems determine the success or failure of resource management. While, WMO (2005) classified the causes of land degradation into biophysical factors such as unsuitable land use (land use for the purpose for which environmentally unsuited for sustainable use), socioeconomic factors such as poor land management practices, land tenure, marketing, institutional support, income and human health, and political factors such as lack of incentives and political instability. In parallel, Mulugeta (2004) argued that land degradation is a biophysical process driven by socioeconomic and political causes in which subsistence agriculture, poverty and illiteracy are important causes of land and environmental degradation in Ethiopia. While, Gebreyesus and Kirubel (2009) reported that the heavy reliance of some 85 percent of Ethiopia's growing population on an exploitative kind of subsistence agriculture is a major reason behind the current state of land degradation. Similarly, studies conducted by Temesgen et al. (2014a, b) in Dera District, Ethiopia exemplified the increased of land degradation which mainly caused by the growing population of the area. Additional study by Fitsum et al. (1999) illustrated that there are multiple interacting forces which have caused and are causing land degradation in Ethiopia. These are the proximate and interacting or root causes. Thus, the above classification indicates that land degradation in Ethiopia is caused by the interaction of many forces.

Direct impacts on land degradation

There are two main levels of uncertainty in assessing the risks of future climate change induced land degradation. The first level, where uncertainties are comparatively low, is the changes of the degrading agent, such as erosive power of precipitation, heat stress from increasing temperature extremes (HÜVE et al. 2011), water stress from droughts, and high surface wind speed. The second level of uncertainties, and where the uncertainties are much larger, relates to the above and belowground ecological changes as a result, changes in climate such as rainfall, temperature, and increasing level of CO_2. Vegetation cover is crucial to protect against erosion (Mullan et al. 2012). Changes in rainfall patterns, such as distribution in time and space, and intensification of rainfall events will increase the risk of land degradation, both in terms of likelihood and consequences (high agreement, medium evidence). Climate induced vegetation changes will increase the risk of land degradation in some areas (where vegetation cover will decline) (medium confidence). Landslides are a form of land degradation that is induced by extreme rainfall events. There is a strong theoretical reason for increasing landslide activity due to intensification of rainfall, but the empirical evidence is so far lacking that climate change has contributed to landslides (Crozier 2010; Huggel et al. 2012; Gariano and Guzzetti 2016), human disturbance may be a more important future trigger than climate change (Froude and Petley 2018).
Erosion of coastal areas as a result of sea level rise will increase worldwide (very high confidence). In cyclone prone areas (such as the Caribbean, Southeast Asia, and the Bay of Bengal) the

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combination of sea level rise and more intense cyclones (Walsh et al. 2016b), and in some areas also land subsidence (Yang et al.2019), will pose a serious risk to people and livelihoods (very high confidence), in some cases even exceeding limits to adaption.

Indirect impacts on land degradation

Large Human Population

Land degradation in Ethiopia is triggered by complex processes and factors (Bezuayehu et al., 2002). Of which, unprecedented population growth has been considered as the leading factor for the search of land for subsistence agriculture (Markos, 1997; Lakew et al., 2000; FAO, 1994) that in turn leads to over exploitation and misuse of land resources. Population pressure with limited land resources and limited secondary (nonfarm) economic activities in Ethiopia is the basis for land shortage, poor land management, and poverty that bring about land degradation. Lakew et al. (2000) identified population pressure, poverty, high costs of and limited access to agricultural inputs and credit, fragmented land holdings and insecure land tenure, and farmers' lack of information about appropriate alternative technologies as the underlying and direct causes of land degradation. Improper resource management and traditional agricultural practices were also found to have great impact on land degradation in Ethiopia (Nyssen et al. 2009).

Forest areas in Ethiopia’s highlands are increasingly threatened (Georg et al., 2014). Growing population pressures have led to expansion of agricultural land and high demands for fuel and construction wood. This overexploitation of forest resources in Ethiopia has left less than three percent of the country’s native forests untouched (World Bank, 2010). Most farmers also opt for expanding cropland through conversion of forests and woodlands when they experience financial strains to access farm inputs like fertilizer and plough (Gray, 2005). The use of firewood and animal dung for household energy sources has been identified as the other important cause of land degradation in Ethiopia (Gebreegziabher et al., 2006). Frequent drought (Maria et al., 2012), deforestation, overgrazing, expansion of cropland and unsustainable use of natural resources (Descheemaeker et al., 2011) has been also contributed to land degradation Ethiopia for long centuries and still going on. Changing patterns of land ownership and policy relating to ethnic groups (Sil et al., 2014; Berry, 2003) and the frequent redistribution of land have been exacerbated tenure insecurity thereby reducing the incentive to engage in land conservation (Maria et al., 2012). Ending future land distributions have positive and significant impact in land investment improvement and reduction of land degradation (Benin, 2002). Furthermore, uncontrolled grazing in the forests is common and endangers the soil and water conservation activities implemented in the adjacent watersheds (Georg et al., 2014).

Deforestation

Deforestation is severe and has a long history in Amhara region where subsistence farming and settlements have been changing landscapes for millennia. Deforestation was always followed by a change in land use and land cover, from forest to grassland and cropland. A particular increase in cropland was observed in the second half of the 20th century, largely at the expense of grassland and forestland (Hans et al., 2010). About 20 thousand hectares of forest are harvested annually in

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Amhara region for fuel wood, logging and construction purpose (Lakew et al., 2000; ILRI, 2000). Forest degradation in Amhara region is dwindling from day-to-day due to population growth, overgrazing pressure, and lack of strong forest policy. There is still no regional forest policy, strategy, and proclamation to control deforestation and illegal forest product movement and encroachments (Mulatie et al., 2015). The increasing demand for pasture, shelter, food crops, urbanization, and the eventual conversion of natural forests to croplands were also contributed to severe deforestation in Amhara region.

Large Livestock size and Uncontrolled grazing

Livestock production is a major component of the economy of Amhara region. The region has been the home to about 35% of the total country's livestock population (BoA, 1999) and based on agricultural sample survey 2012/13, high populations of livestock per km^2 were found in this region (Samson & Frehiwot, 2014). Compared with other regions, Amhara stands first in the number of goats, second in cattle, sheep, asses, horses and poultry (CSA, 1998). Livestock in the region provide about 16.4 million tonnes of manure annually, equivalent to 114 thousand tonnes of nitrogen, which is being used primarily for fuel rather than manure (CEDEP, 1999).
Uncontrolled or free grazing and browsing too many livestock for too long a period on land unable to recover its vegetation (overgrazing) are the dominant grazing systems in the region. Uncontrolled grazing on crop land contributes to soil compaction and the need for frequent tillage to prepare fields for crops, making practices such as reduced tillage less feasible. Grazing concentrated on hillsides fragile areas slopes, on marginal and cultivated land after harvest result in soil compaction, low moisture retention and high runoff, which are the main causes for the formation of gully, excessive vegetation removal, and reductions in crop yields (Lakew et al., 2000). Uncontrolled grazing system also has a negative effect on the conservation efforts, as trampling animals often damage physical conservation structures such as stone terraces and soil bunds. Biological conservation practices such as grass strips and tree plantations are also being destroyed or trampled, reducing the chance for establishment and regeneration. It is more destructive during the rainy season when other sources of feed (e.g., stubble grazing and crop residues) are in short supply (Samuel & John, 2002), the period of high runoff production, and when the soil is very sensitive to be washed by floods.

CONCLUSION

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

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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.

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 landscapelevel, 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.

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