Rangeland afforestation is not a natural climate solution

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Large-scale tree planting on global rangelands is promoted as a natural climate solution (NCS), but there is little scientific evidence to support this narrative. The presumed benefits of rangeland afforestation originate from five major misconceptions: (1) conflation between reforestation and afforestation, (2) overestimation of carbon (C) sequestration potential, (3) insufficient recognition of rangeland ecosystem services, (4) potential for adverse ecological outcomes, and (5) neocolonial tendencies of afforestation programs. Rangeland afforestation possesses minimal potential for additional C storage, but it has high potential to reduce vital rangeland ecosystem services that benefit rangeland residents and non-residents alike. Conservation of existing C—most of which is stored belowground, where it is less vulnerable to loss—may prove to be the most appropriate NCS for extensively managed rangelands. Stewardship strategies promoting rangeland multifunctionality will not only contribute to climate-change mitigation but also support biodiversity conservation and sustainable production of high-protein foods for marginalized populations.

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Natural climate solutions (NCSs) are attracting substantial attention as a means of mitigating climate change (Griscom *et al*. 2017). They involve the implementation of conservation, restoration, and improved land management to increase carbon (C) storage and minimize greenhouse-gas emissions. Reforestation and forest conservation have been identified as the solutions possessing the greatest potential for global climate-change mitigation (Griscom *et al*. 2017; Bastin *et al*. 2019). Consequently, large-scale tree planting, along with forest restoration and protection, are being promoted by multiple organizations and programs (Table 1). Both nonprofit and commercial initiatives are involved and the boundaries between them are often difficult to discern.

In a nutshell:

- Global rangelands are being targeted for large-scale tree planting as part of efforts to remove and store atmospheric carbon (C)
- This narrative is founded on several misconceptions that are unsupported by scientific evidence
- Rangeland tree planting has not only limited potential for additional C storage but also a high potential to reduce vital ecosystem services, thereby adversely affecting the lives and livelihoods of local people
- Conservation of existing C and associated ecosystem services may be the most appropriate natural climate solution for extensively managed rangelands

Afforestation goals are varied but primarily emphasize climate-change mitigation, along with ecosystem conservation and restoration (Seddon *et al*. 2021). C credits from afforestation projects have increased rapidly, accounting for more than one-third of global credits issued in 2022 (Haya *et al*. 2023). However, information describing progress toward targeted goals, including both ecological and social outcomes, remains limited (Turner *et al*. 2021).

Increasing emphasis on NCSs has perpetuated the narrative of tree planting as a panacea for climate-change mitigation, regardless of tree species, existing vegetation, or climatic conditions (Holl and Brancalion 2020; Seddon *et al*. 2021). However, this narrative is founded on multiple misconceptions, which create the potential for adverse ecosystem service trade-offs, especially for rangelands (Fleischman *et al*. 2020; Vetter 2020). Rangeland loss through afforestation has been identified as a priority concern by the Global Coordinating Group of the International Year of Rangelands and Pastoralists (https://iyrp.info), which has been designated for 2026 by the UN General Assembly.

Rangelands are dominated by native vegetation—primarily grasses, forbs, shrubs, and scattered trees—and are managed as social–ecological systems to supply diverse ecosystem services that enhance human well-being (Briske and Coppock 2023). Rangelands encompass several diverse biomes, including deserts, grasslands, shrub steppe, savannas, and open woodlands, that collectively cover approximately 50% of the world's terrestrial land area (www.rangelandsdata.org/atlas) and

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Table 1. Targets and time frames of major tree-planting programs

Program	Magnitude	Time frame
Bonn Challenge	350 million hectares	2011–2030
Grain for Green (China)	29 million hectares	1999–2018
Green Legacy (Ethiopia)	20 billion trees	2019–2023
EU Biodiversity Strategy	3 billion trees	2020–2030
1 Billion Trees (New Zealand)	1 billion trees	2017–2028
New York Declaration on Forests	150 million hectares	2014–2020
Trillion Tree Campaign (formerly Billion Tree Campaign)	1 trillion trees (1 billion trees)	2018–ongoing (2006–2018)
Trillion Trees Platform (World Economic Forum)	1 trillion trees	2020–ongoing
UN Decade on Restoration	1 billion hectares, including forests	2020–2030
Multiple corporate pledges (many contribute to above programs)	>200 million trees	2020–2030

Notes: data from Seddon *et al*. (2021); area values include varying proportions of afforestation pledges.

represent a vital but often overlooked component of planetary stewardship (Briske and Coppock 2023). Of the ecosystem services provided by rangelands, the provisioning services of forage and livestock production are the most widely recognized, but critical regulating, supporting, and cultural ecosystem services also exist. Despite their ecological and economic importance, rangelands continue to be undervalued (Vetter 2020; Tittonell 2021) and are frequently converted to alternative land uses with little regard for the associated loss of ecosystem services (Briske and Coppock 2023). This makes rangelands especially vulnerable to afforestation initiatives financed by C offset markets and supported by restoration and conservation programs.

Our objective here is to challenge the narrative that rangeland afforestation represents a viable NCS. We specifically focus on the conversion of native rangeland to large-scale tree plantations to highlight five major misconceptions used to promote rangeland afforestation: (1) conflation of reforestation and afforestation, (2) overestimation of C sequestration potential, (3) insufficient recognition of rangeland ecosystem services, (4) potential for adverse ecological outcomes, and (5) the neocolonial tendencies of afforestation programs as climate mitigation strategies. Each is discussed in greater detail in the sections below.

Afforestation should not be conflated with reforestation

The popularity of rangeland afforestation is supported by the myth that rangelands represent degraded forests, rather than natural biomes supported by specific climatic conditions and natural disturbance regimes (Davis and Robbins 2018; Bond 2019). This erroneous, but widely held, interpretation is believed to have originated with Western European scholars in the 18th century and then became more widespread during colonial expansion (Ratnam *et al.* 2011; Joshi *et al.* 2018). The institutional dominance of forestry services during this period promoted forested landscapes as environmentally beneficial and rangelands as degraded—a perception that remains to this day. However, tropical savannas and grasslands as well as other biomes that comprise rangelands precede human evolution by millions of years (Veldman 2016; Bond 2019).

Insufficient distinctions between savannas and forests also contribute to the justification of rangeland afforestation (Ratnam *et al.* 2011; Veldman *et al.* 2015). The UN Food and Agricultural Organization defines a forest as any area greater than 0.5 ha in size with trees exceeding 5 m in height and a tree canopy that extends over more than 10% of the ground area (FAO 2020). According to this definition, most savannas and open woodlands are considered "forests". Consequently, many of the proposed tree planting targets have been designated in areas that historically were savannas or only sparsely wooded (Veldman *et al.* 2015; Veldman 2016). However, forests and savannas have fundamentally different ecological dynamics, in addition to ecosystem structures, that are reflected in distinct species assemblages with different functional traits and responses to fire (Ratnam *et al.* 2011; Veldman *et al.* 2015). Tree planting in historically non-forested areas therefore represents afforestation rather than reforestation (Veldman 2016).

Large-scale tree planting programs target sites that may potentially support "forests" based on climate modeling projections (Bastin *et al.* 2019). However, the hydroclimatic conditions required to support tree growth often receive insufficient emphasis. It has been estimated that approximately 50% of the global land area considered suitable for tree planting is unable to provide sufficient water for tree growth from precipitation alone, especially in Africa, Oceania, and portions of South Asia (Ricciardi *et al.* 2022). Tree planting in water-scarce regions has resulted in extensive failures and ineffective resource investments (Holl and Brancalion 2020). For example, in afforestation projects conducted in China between 1952 and 2005, an estimated 24% of planted trees survived (Cao *et al.* 2011); tree planting programs conducted in India from 2016 to 2019, in the Himalayan state of Himachal Pradesh, were similarly ineffective (Rana *et al.* 2022). Moreover, by 2050, climate change is projected to further contract the global area suitable for tree planting by 25% (Bastin *et al.* 2019).

Afforestation has limited carbon storage potential

Forest-based climate solutions assume that additional C storage is a reliable outcome of tree planting regardless of forest type, tree species, environmental characteristics, and the C sequestration potential of existing ecosystems (Lewis *et al.* 2019). These assumptions may have been derived from mature, native forests that store large amounts of C, both aboveground and belowground (Cook-Patton *et al.* 2020). The conservation and reforestation of previously forested regions in the humid tropics and subtropics, where environmental conditions are conducive to high C sequestration, is therefore critical for climate-change mitigation (Lewis *et al*. 2019; Di Sacco *et al*. 2021).

However, the high C sequestration potential of mature, native forests does not directly translate to forest plantations, which have a much lower potential for C storage (Noormets *et al*. 2015; Lewis *et al*. 2019). Consequently, soil C storage is consistently reduced by conversion of native forest to tree plantations (Panel 1; Guo and Gifford 2002; Paul *et al*. 2002). Although trees in natural forests and forest plantations exhibit similar productivity, commercially selected tree species allocate greater amounts of C to aboveground biomass at the expense of roots and root symbionts (Noormets *et al*. 2015). Proportionally greater C storage in aboveground biomass, especially in the form of fine fuels that accumulate early in plantation establishment, increases vulnerability to loss by fire, as well as to drought and pathogens—disturbances that are anticipated to increase under future climates (Bond 2019). Insufficient recognition of the vulnerability of C storage conveys major risks for forest-based climate solutions that could indeed worsen climate-change impacts (Dass *et al*. 2018; Anderegg *et al*. 2020).

Conversion of native rangeland to tree plantations, especially in regions that are more mesic, decreases the C storage capacity of the soil (Panel 1) to a similar degree as conversion of native forests to tree plantations. Large-scale tree planting projections often erroneously overestimate the potential for C sequestration because estimates of existing C storage in rangeland biomass and soils are typically excluded (Griscom *et al*. 2017; Rohatyn *et al*. 2022). This may occur in part because rangeland C is primarily stored belowground in root biomass and soil organic C, which makes it less obvious and more difficult to measure than in forests (Veldman 2016; Sanderson *et al*. 2020). Rangelands are estimated to represent approximately 30% of the total terrestrial C pool and large interannual variation in C sequestration by intertropical rangelands is thought to be a major contributor to interannual variation in the global C cycle (Ahlström *et al*. 2015).

Rangelands provide critical ecosystem services

Rangelands are primarily managed as extensive pastoral systems that supply diverse ecosystem services, including biodiversity, biomass production, water and nutrient cycling, C storage, and rich cultural services (Briske and Coppock 2023). Of the Earth's land area, an estimated 35% is identified as "ecologically intact" and 15% is currently located within protected areas (Kuempel *et al*. 2020). Rangelands represent approximately 50% of these ecologically intact areas, which support high biodiversity and supply diverse ecosystem services (Garnett *et al*. 2018). Rangelands have been estimated to provide 35% of the total value of terrestrial ecosystem services, a level comparable to that of temperate forests (de Groot *et al*. 2012). Sustainably managed pastoral systems are vital for biodiversity conservation because they prevent rangeland conversion to alternative uses (Briske and Coppock 2023). In addition, diverse flora and fauna are exclusively dependent on habitats provided by rangeland ecosystems, with many of these species now listed as imperiled (www.rangelandsdata.org/atlas).

Increasing woody plant cover has been found to reduce rangeland biodiversity at a global scale (Wieczorkowski and Lehmann 2022). The diversity of forb species (ie herbaceous dicots) is most sensitive to woody plant expansion, followed by C_4 and C_3 grass species. Biodiversity loss is primarily a consequence of decreasing herbaceous production in response to increasing tree cover, which in turn suppresses the natural disturbance regimes of fire and grazing (Panel 2; Figure 1). Equally problematic is the difficulty encountered with restoring biodiversity after conversion of grasslands to tree plantations.

Panel 1. Soil carbon dynamics after conversion of rangelands to tree plantations

The tremendous heterogeneity of rangelands and their associated natural disturbance regimes and anthropogenic management practices greatly complicate the ecological dynamics of soil carbon (C) storage and its persistence. This complexity, coupled with limited empirical evidence, has been a major contributor to the uncertainty associated with natural climate solutions on rangelands (Li *et al*. 2018; Reinhart *et al*. 2021). However, the following broad generalizations emerged from an analysis of several major assessments of soil C dynamics after conversion of rangelands to tree plantations (Guo and Gifford 2002; Paul *et al*. 2002; Griscom *et al*. 2017; Lewis *et al*. 2019).

- Soil C decreases when ecologically intact rangelands are converted to tree plantations, primarily in response to physical soil disturbance. Conversely, soil C increases when rangelands that have been ecologically degraded by cropland conversion, severe overgrazing, or soil erosion are converted to tree plantations. However, few studies have established experimental controls to determine whether improved management (eg reduced grazing or erosion prevention) or tree planting is the primary mechanism contributing to greater soil C.
- Plantations of pine and other softwood species are associated with greater reductions in soil C than plantations of deciduous hardwood species. However, although seldom used in plantations, when native deciduous species are used they are often located in regions where natural forest regeneration may have proceeded.
- Soil C accumulates slowly after tree planting and values may decline immediately after planting. Soil C begins to increase after approximately 10 years, but maximum soil C storage may not be reached for decades (≥45 years) after tree planting. Consequently, plantations harvested on short rotation intervals or subject to frequent, severe disturbances will have minimal potential to store additional C.

Panel 2. Carbon storage: biodiversity trade-off created by fire suppression

The Brazilian Cerrado is among the most species-rich tropical savannas in the world. However, three decades of government-imposed fire suppression has resulted in a 14-fold increase in tree biomass in the Cerrado, along with increases in both tree density and canopy cover (Figure 1; Abreu *et al.* 2017). Increasing tree growth has increased total carbon (C) storage, including both aboveground and belowground C (0–20 cm soil depth), from 20.8 Mg ha^{-1} in savanna to 83.5 Mg ha^{-1} in forest. However, greater tree biomass and canopy cover has contributed to a large loss of biodiversity in these savannas. Plant species richness and savanna specialist plant species fell by 27% and 67%, respectively, during a 30-year period of increasing canopy closure. Similarly, the total number of ant species and savanna specialist ant species fell by 35% and 86% during this same period. Biodiversity declined rapidly after a threshold tree basal area of 15 m^2 ha^{-1} and a leaf area index of 2.5 (total leaf area per ground area) were attained. Forest-based natural climate solutions must carefully consider potential trade-offs with biodiversity.

Figure 1. Increasing tree density and cover following fire suppression in the Brazilian Cerrado increases carbon storage but substantially decreases savanna biodiversity. Image credit: A Tomaselli Fidelis.

Centuries may be required for tropical and temperate grasslands to recover their original plant diversity, especially when soil disturbance damages or destroys belowground plant organs (Zaloumis and Bond 2011; Nerlekar and Veldman 2020). The large and potentially irreplaceable loss of rangeland biodiversity alone provides a strong justification for limited use of rangeland afforestation as an NCS.

Livestock husbandry provides livelihoods to millions of pastoralists worldwide and is a source of animal products, including fiber, meat, and milk. Ruminant herbivores maintained by pastoralists convert energy from diverse forages that are undigestible by humans to highly desirable and nutritionally valuable meat and dairy products (Adesogan *et al.* 2020). In 2016, meat and dairy products accounted for 25% of the protein and 18% of the calories consumed globally. These high-protein food sources are especially critical to the health of women and children in low-income countries, and they increase food security in rangelands characterized by high resource variability (Adesogan *et al.* 2020). Consequently, rangeland afforestation may serve to diminish the supply of a vital source of dietary protein that supports food security, particularly in the Global South (Cook-Patton *et al.* 2020). Moreover, afforestation of cropland in agro-pastoral regions may further compromise food security by reducing crop production and by increasing the costs of plant-based foods (Kreidenweis *et al.* 2016).

Negative ecological outcomes of rangeland afforestation

Afforestation substantially alters ecosystem processes, including water and energy fluxes between the land and atmosphere. Tree establishment in previously grass-dominated areas increases transpiration, which may lower water tables and reduce stream flows (Jackson *et al.* 2005; Ge *et al.* 2023). Increased transpiration results from the deeper rooting depth, higher leaf area index, and lower albedo—solar radiation reflected from tree canopies—of trees as compared to other vegetation types. Increased transpiration after tree planting may increase regional precipitation, a process known as "atmospheric water recycling" (Hoek van Dijke *et al.* 2022). Although estimated to increase water availability by up to 6% in some regions, expanded tree cover may reduce water availability by as much as 38% in other regions. Overall, increasing global forest cover not only is projected to decrease water availability on most continents but also may exacerbate water limitations in regions where water is already scarce (eg parts of Asia and sub-Saharan Africa) (Naik and Abiodun 2016).

The warming associated with the lower albedo of forest canopies has been demonstrated to partially negate the cooling effect created by photosynthetic removal of atmospheric C. Approximately 448 million hectares of the world's rangelands have been identified as being suitable for afforestation, with a C sequestration potential of 32.2 billion tons by 2100 (Rohatyn *et al*. 2022). However, 22.6 billion tons of C would be required to offset the warming derived from the lower albedo of forest canopies. Consequently, afforestation may provide a minimal offset for the projected C emissions during this period. The extent to which forest cover reduces albedo depends on the background albedo of the existing vegetation and soil surfaces, and characteristics of the forest canopy.

Tree plantations, especially those composed of pines (*Pinus* spp) and eucalypts (*Eucalyptus* spp), accumulate large amounts of highly flammable aboveground biomass as compared to most rangeland vegetation types (Nuñez *et al*. 2021). This increases the potential for intense wildfires that may adversely affect human livelihoods, wildlife and wildlife habitat, soil erosion, and existing C storage. In contrast, rangeland fires often burn more frequently but less intensely and therefore are typically less destructive because they are supported by lower fuel loads, composed primarily of fine fuels. In addition, less C is lost in rangeland fires than in forest fires because the majority of rangeland C is stored belowground, and C losses that do occur are rapidly recaptured during subsequent growing seasons (Veldman 2016; Dass *et al*. 2018).

Afforestation represents a neocolonial climate mitigation solution

Rangeland afforestation is promoted as a win–win strategy that increases C sequestration, improves ecosystem function, and subsequently increases socioeconomic resilience of local communities. This portrayal represents an appealing outcome; however, substantial uncertainties and trade-offs exist among these anticipated goals (Turner *et al*. 2021). Afforestation is implemented through top-down initiatives that often reflect international and national agendas. For example, rangelands may be considered as unproductive wastelands and as such are ideal candidates for afforestation to achieve climate-change mitigation goals. Implementation of afforestation initiatives can be influenced by the political goals of governments, which may result in the resettlement of local populations, land privatization, and transformation of livelihoods (Elkin 2022; Turner *et al*. 2023). Even when not overtly coercive, afforestation initiatives are often inconsistent with the interests and livelihood requirements of local peoples and create adverse ecological and socioeconomic impacts (Panel 3; Figure 2; Davis and Robbins 2018; Malkamäki *et al*. 2018).

The win–win strategy associated with afforestation programs is simply assumed to occur with little monitoring of

Panel 3. Adverse consequences of afforestation on local communities

The Great Green Wall Initiative (GGWI), supported by the UN Convention to Combat Desertification in sub-Saharan Africa, has mobilized considerable public interest and investment from a consortium of donors. Afforestation initiatives are diverse, including stabilization of sandy soils in Senegal, rehabilitation of hardpan areas in Burkina Faso and Niger (Figure 2), and improved management of coppiced trees and shelterbelts on cropland in Nigeria. However, afforestation programs have experienced low rates of tree survival and irrigation is often necessary to enhance tree establishment, which diverts water from human and livestock needs (Turner *et al*. 2023). In the Ferlo region of Senegal, a sandy area with a deep water table, increased water demand for GGWI nurseries has forced families to reduce their water use and delayed livestock access to water. Afforestation has blocked critical livestock migration routes, and rangeland enclosures and tree plantations have obstructed rangeland access to local herders (Ndiaye 2016; Turner *et al*. 2021). In Niger, projects have created opportunities for privatization of communal lands by local elites against the interests of the rural poor—an outcome that is consistent with GGWI goals to restore land to make it more attractive for outside investment (eg crop production or peri-urban development; Sarr *et al*. 2021; Turner *et al*. 2021). In the Sudano-Sahelian region, afforestation programs frequently benefit local or extra-local elites while exacerbating the risks to and vulnerability of rural communities that depend on livestock production and wild foods for subsistence.

Figure 2. A forestry guard patrols the Lido Great Green Wall afforestation site in the Dosso region of Niger. Image credit: DAWNING/N Parisse.

project outcomes beyond technical metrics of numbers of trees planted, hectares restored, and people trained (Turner *et al.* 2021). The presumed outcomes—greater C sequestration, tree cover, and direct benefits to local communities—are often slow to develop (Coleman *et al.* 2021). In contrast, the adverse outcomes can be immediate, including water scarcity and reduced access to grazing land, especially on communal and open-access rangelands that do not possess well-defined property rights. Marginalized populations that are highly dependent on natural resources, such as pastoralists and subsistence farmers, are most negatively affected by afforestation (Yeh 2009; Turner *et al.* 2023). Consequently, rather than producing a win–win outcome, afforestation may increase the vulnerability of rural populations to climate change (Panel 3). This contributes to a situation where marginalized populations of the Global South may further bear the negative socioeconomic impacts of climate-change mitigation, in addition to the costs of accelerating climate change originating in the Global North (Malkamäki *et al.* 2018).

More ethical NCSs are required to simultaneously prioritize climate mitigation strategies and human well-being on global rangelands (Fleischman *et al.* 2020). This will necessitate a more comprehensive understanding of the livelihoods and socioeconomics of rural communities, and their vulnerabilities to climate change, in contrast to strategies emerging from international and national agendas. Acquisition of this knowledge will necessarily require direct and persistent engagement with multiple livelihood groups within rural communities to determine both their vulnerabilities and adaptive capacities (Davis and Robbins 2018; Di Sacco *et al.* 2021). Current understanding of the ecological and social impacts of afforestation programs is inadequate, reflecting both the top-down implementation of these programs and the minimal monitoring of their outcomes.

Appropriate natural climate solutions for rangelands

Assessments of rangeland NCSs are constrained by minimal documentation of existing C storage and high uncertainty regarding the potential for additional C storage. C storage projections vary widely because rangelands span multiple biomes, natural disturbance regimes, and legacies of diverse land-use practices, in addition to inconsistencies associated with C assessment methodologies (Li *et al.* 2018; Reinhart *et al.* 2021). Climate change is anticipated to further reduce this potential by exacerbating environmental stresses on plant production (Boone *et al.* 2018). Therefore, conservation of existing C stores may prove to be the most appropriate NCS for extensively managed rangelands (Deng *et al.* 2016; Cook-Patton *et al.* 2021). As compared to forest C storage, C storage belowground, where it is less vulnerable to loss by fire, grazing, and drought, further justifies the importance of rangeland C conservation (Dass *et al.* 2018; Anderegg *et al.* 2020).

Rangelands are of critical importance to planetary stewardship beyond their potential for C storage and climate regulation. They supply diverse ecosystem services not only to rangeland residents but also to society at large (Tittonell 2021; Briske and Coppock 2023). Consequently, stewardship strategies must embrace a multifunctional perspective that recognizes and values the full complement of, and human dependence on, rangeland ecosystem services. Many rangeland systems rely on the natural disturbance regimes of fire, grazing, and drought to maintain multifunctionality, including long-term C storage (Li *et al.* 2018; Zhou *et al.* 2022). Grasses remain the dominant contributor to soil C in African savanna even with increasing woody cover and fire frequency (Coetsee *et al.* 2023). In contrast, rangeland afforestation substantially marginalizes rangeland multifunctionality by minimizing natural disturbance regimes, disturbing soils, and decreasing biodiversity (Di Sacco *et al.* 2021; Seddon *et al.* 2021). The resulting impacts on ecosystem function and long recovery times following substantial modification will have important, and often adverse, consequences for human well-being.

Conclusions

The narrative promoting large-scale tree planting on global rangelands as an NCS cannot be scientifically substantiated. Rangeland afforestation possesses the potential for only minimal storage of additional C but has great potential for increasing the loss of the diverse ecosystem services that support planetary stewardship. This represents a very high cost for C storage that will be of negligible importance in offsetting current global C emissions (Holl and Brancalion 2020; Rohatyn *et al.* 2022). Future assessments of the C sequestration potential of rangelands require greater scrutiny from a multifunctional perspective, including (1) conservation of existing C storage, (2) current and future hydroclimatic constraints on tree growth, (3) potential for adverse trade-offs among ecosystem services, and (4) cultural and socioeconomic impacts on rural communities (Di Sacco *et al.* 2021; Seddon *et al.* 2021). Effective stewardship of extensively managed rangelands will contribute to future climate-change mitigation by supporting biodiversity conservation and the sustainable production of high-protein foods for marginalized populations.

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References

Abreu RCR, Hoffmann WA, Vasconcelos HL, *et al.* 2017. The biodiversity cost of carbon sequestration in tropical savanna. *Sci Adv* **3**: e1701284.

Adesogan AT, Havelaar AH, McKune SL, *et al.* 2020. Animal source foods: sustainability problem or malnutrition and sustainability solution? Perspective matters. *Global Food Security* **25**: 100325.

Ahlström A, Raupach MR, Schurgers G, *et al.* 2015. The dominant role of semi-arid ecosystems in the trend and variability of the land CO_2 sink. *Science* **348**: 895–99.

Anderegg WRL, Trugman AT, Badgley G, *et al.* 2020. Climate-driven risks to the climate mitigation potential of forests. *Science* **368**: eaaz7005.

Bastin J-F, Finegold Y, Garcia C, *et al.* 2019. The global tree restoration potential. *Science* **365**: 76–79.

Bond WJ. 2019. Open ecosystems: ecology and evolution beyond the forest edge. Oxford, UK: Oxford University Press.

Boone RB, Conant RT, Sircely J, *et al.* 2018. Climate change impacts on selected global rangeland ecosystem services. *Glob Change Biol* **24**: 1382–93.

Briske DD and Coppock DL. 2023. Rangeland stewardship envisioned through a planetary lens. *Trends Ecol Evol* **38**: 109–12.

Cao S, Chen L, Shankman D, *et al.* 2011. Excessive reliance on afforestation in China's arid and semi-arid regions: lessons in ecological restoration. *Earth-Sci Rev* **104**: 240–45.

Coetsee C, February EC, Wigley BJ, *et al.* 2023. Soil organic carbon is buffered by grass inputs regardless of woody cover or fire frequency in an African savanna. *J Ecol* **111**: 2483–95.

Coleman EA, Schultz B, Ramprassad V, *et al.* 2021. Limited effects of tree planting on forest canopy cover and rural livelihoods in northern India. *Nat Sustain* **4**: 997–1004.

Cook-Patton SC, Drever CR, Griscom BW, *et al.* 2021. Protect, manage and then restore lands for climate mitigation. *Nat Clim Change* **11**: 1027–34.

Cook-Patton SC, Leavitt SM, Gibbs D, *et al.* 2020. Mapping carbon accumulation potential from global natural forest regrowth. *Nature* **585**: 545–50.

Dass P, Houlton BZ, Wang Y, *et al.* 2018. Grasslands may be more reliable carbon sinks than forests in California. *Environ Res Lett* **13**: 074027.

Davis DK and Robbins P. 2018. Ecologies of the colonial present: pathological forestry from the *taux de boisement* to civilized plantations. *Environ Planning E Nat Space* **1**: 447–69.

de Groot R, Brander L, van der Ploeg S, *et al.* 2012. Global estimates of the value of ecosystems and their services in monetary units. *Ecosyst Serv* **1**: 50–61.

Deng L, Zhu G-Y, Tang Z-S, *et al.* 2016. Global patterns of the effects of land-use changes on soil carbon stocks. *Global Ecol Conserv* **5**: 127–38.

Di Sacco A, Hardwick KA, Blakesley D, *et al.* 2021. Ten golden rules for reforestation to optimize carbon sequestration, biodiversity recovery and livelihood benefits. *Glob Change Biol* **27**: 1328–48.

Elkin RS. 2022. Plant life: the entangled politics of afforestation. Minneapolis, MN: University of Minnesota Press.

FAO (UN Food and Agriculture Organization). 2020. Global forest resources assessment 2020: terms and definitions. Rome, Italy: FAO.

Fleischman F, Basant S, Chhatre A, *et al.* 2020. Pitfalls of tree planting show why we need people-centered natural climate solutions. *BioScience* **70**: 947–50.

Garnett ST, Burgess ND, Fa JE, *et al.* 2018. A spatial overview of the global importance of Indigenous lands for conservation. *Nat Sustain* **1**: 369–74.

Ge F, Xu M, Li B, *et al.* 2023. Afforestation reduced the deep profile soil water sustainability on the semiarid Loess Plateau. *Forest Ecol Manag* **544**: 121240.

Griscom BW, Adams J, Ellis PW, *et al.* 2017. Natural climate solutions. *P Natl Acad Sci USA* **114**: 11645–50.

Guo LB and Gifford RM. 2002. Soil carbon stocks and land use change: a meta analysis. *Glob Change Biol* **8**: 345–60.

Haya BK, Evans S, Brown L, *et al.* 2023. Comprehensive review of carbon quantification by improved forest management offset protocols. *Front Forests Global Change* **6**: 958879.

Hoek van Dijke AJ, Herold M, Mallick K, *et al.* 2022. Shifts in regional water availability due to global tree restoration. *Nat Geosci* **15**: 363–68.

Holl KD and Brancalion PHS. 2020. Tree planting is not that simple. *Science* **368**: 580–81.

Jackson RB, Jobbagy EG, Avissar R, *et al.* 2005. Trading water for carbon with biological carbon sequestration. *Science* **310**: 1944–47.

Joshi AA, Sankaran M, and Ratnam J. 2018. "Foresting" the grassland: historical management legacies in forest–grassland mosaics in southern India, and lessons for the conservation of tropical grassy biomes. *Biol Conserv* **224**: 144–52.

Kreidenweis U, Humpenöder F, Stevanović M, *et al.* 2016. Afforestation to mitigate climate change: impacts on food prices under consideration of albedo effects. *Environ Res Lett* **11**: 085001.

Kuempel CD, Chauvenet ALM, Possingham HP, and Adams VM. 2020. Evidence-based guidelines for prioritizing investments to meet international conservation objectives. *One Earth* **2**: 55–63.

Lewis SL, Wheeler CF, Mitchard ETA, *et al.* 2019. Regenerate natural forests to store carbon. *Nature* **568**: 25–28.

Li W, Ciais P, Guenet B, *et al.* 2018. Temporal response of soil organic carbon after grassland-related land-use change. *Glob Change Biol* **24**: 4731–46.

Malkamäki A, D'Amato D, Hogarth NJ, *et al.* 2018. A systematic review of the socio-economic impacts of large-scale tree plantations, worldwide. *Global Environ Chang* **53**: 90–103.

Naik M and Abiodun BJ. 2016. Potential impacts of forestation on future climate change in southern Africa. *Int J Climatol* **36**: 4560–76.

Ndiaye A. 2016. Practices of the Great Green Wall project in the Ferlo (Senegal): effects on pastoral resilience and development. *World J Soc Sci* **3**: 2.

Nerlekar AN and Veldman JW. 2020. High plant diversity and slow assembly of old-growth grasslands. *P Natl Acad Sci USA* **117**: 18550–56.

Noormets A, Epron D, Domec JC, *et al.* 2015. Effects of forest management on productivity and carbon sequestration: a review and hypothesis. *Forest Ecol Manag* **355**: 124–40.

Nuñez MA, Davis KT, Dimarco RD, *et al*. 2021. Should tree invasions be used in treeless ecosystems to mitigate climate change? *Front Ecol Environ* **19**: 334–41.

Paul KI, Polglase PJ, Nyakuengama JG, *et al*. 2002. Change in soil carbon following afforestation. *Forest Ecol Manag* **168**: 241–57.

Rana P, Fleischman F, Ramprasad V, and Lee K. 2022. Predicting wasteful spending in tree planting programs in Indian Himalaya. *World Dev* **154**: 105864.

Ratnam J, Bond WJ, Fensham RJ, *et al*. 2011. When is a "forest" a savanna, and why does it matter? *Global Ecol Biogeogr* **20**: 653–60.

Reinhart KO, Sanni Worogo HS, and Rinella MJ. 2021. Ruminating on the science of carbon ranching. *J Appl Ecol* **59**: 642–48.

Ricciardi L, D'Odorico P, Galli N, *et al*. 2022. Hydrological implications of large-scale afforestation in tropical biomes for climate change mitigation. *Philos T Roy Soc B* **377**: 20210391.

Rohatyn S, Yakir D, Rotenberg E, *et al*. 2022. Limited climate change mitigation potential through forestation of the vast dryland regions. *Science* **377**: 1436–39.

Sanderson JS, Beutler C, Brown JR, *et al*. 2020. Cattle, conservation, and carbon in the western Great Plains. *J Soil Water Conserv* **75**: 5A–12A.

Sarr MS, Diallo AM, and King-Okumu C. 2021. A review of public versus private reforestation programs in the Senegalese Sahel: taking stock of realities and challenges. *Restor Ecol* **30**: e13582.

Seddon N, Smith A, Smith P, *et al*. 2021. Getting the message right on nature-based solutions to climate change. *Glob Change Biol* **27**: 1518–46.

Tittonell P. 2021. Beyond CO_2: multiple ecosystem services from ecologically intensive grazing landscapes of South America. *Front Sustain Food Syst* **5**: 664103.

Turner MD, Carney T, Lawler L, *et al*. 2021. Environmental rehabilitation and the vulnerability of the poor: the case of the Great Green Wall. *Land Use Policy* **111**: 105750.

Turner MD, Davis DK, Yeh ET, *et al*. 2023. Great Green Walls: hype, myth, and science. *Annu Rev Env Resour* **48**: 1–25.

Veldman JW, Overbeck GE, Negreiros D, *et al*. 2015. Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *BioScience* **65**: 1011–18.

Veldman JW. 2016. Clarifying the confusion: old-growth savannahs and tropical ecosystem degradation. *Philos T Roy Soc B* **371**: 20150306.

Vetter S. 2020. With power comes responsibility – a rangelands perspective on forest landscape restoration. *Front Sustain Food Syst* **4**: 549483.

Wieczorkowski JD and Lehmann CER. 2022. Encroachment diminishes herbaceous plant diversity in grassy ecosystems worldwide. *Glob Change Biol* **28**: 5532–46.

Yeh ET. 2009. Greening western China: a critical view. *Geoforum* **40**: 884–94.

Zaloumis NP and Bond WJ. 2011. Grassland restoration after afforestation: no direction home? *Austral Ecol* **36**: 357–66.

Zhou Y, Singh J, Butnor JR, *et al*. 2022. Limited increases in savanna carbon stocks over decades of fire suppression. *Nature* **603**: 445–49.