UNITED NATIONS

BES

IPBES/6/INF/1/Rev.1

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

Distr.: General
22 June 2018

English only

Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
Sixth session
Medellin, Colombia, 18–24 March 2018
Agenda item 7

Thematic assessment of land degradation and restoration

Chapters of the thematic assessment of land degradation and restoration

Note by the secretariat

1. In paragraph 2 of section IV of decision IPBES-3/1, the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services approved the undertaking of a thematic assessment of land degradation and restoration in accordance with the procedures for the preparation of the Platform's deliverables set out in annex I to decision IPBES-3/3, based on the scoping report for the assessment set out in annex VIII to decision IPBES-3/1.

2. In response to the decision, a set of eight chapters (IPBES/6/INF/1) and a summary for policymakers (IPBES/6/3) were produced by an expert group in accordance with the procedures for the preparation of the Platform's deliverables for consideration by the Plenary at its sixth session.

3. In paragraph 1 of section V of decision IPBES-6/1, the Plenary approved the summary for policymakers of the thematic assessment of land degradation and restoration (IPBES/6/15/Add.5) and accepted the individual chapters of the assessment, on the understanding that the chapters would be revised following the sixth session as document IPBES/6/INF/1/Rev.1 to correct factual errors and to ensure consistency with the summary for policymakers as approved. The annex to the present note, which is presented without formal editing, sets out the final set of chapters of the thematic assessment of land degradation and restoration including their executive summaries.

4. A laid-out version of the final thematic assessment report on land degradation and restoration (including a foreword, statements from key partners, acknowledgements, a preface, the summary for policymakers, the revised chapters and annexes setting out a glossary and lists of acronyms, authors, review editors and expert reviewers) will be made available on the website of the Platform prior to the seventh session of the Plenary.

Chapter 6

Responses to halt land degradation and to restore degraded land

Coordinating Lead Authors

Ram Pandit (Nepal), John Parrotta (United States of America)

Lead Authors

Yaakov Anker (Israel), Emilie Coudel (France/Brazil), Cristóbal Félix Diaz Morejón (Cuba), Jim Harris (United Kingdom of Great Britain and Northern Ireland), Douglas L. Karlen (United States of America), Ádám Kertész (Hungary), Juana L. Mariño De Posada (Colombia), Phumza Ntshotsho Simelane (South Africa), Noraini M. Tamin (Malaysia), Daniel Luis Mascia Vieira (Brazil)

Fellow

Ruishan Chen (China)

Contributing Authors

Mumuni Abu (Ghana), Sebastian Arnhold (Germany), Simone Athayde (Brazil), Ben Boer (Australia), Violaine Brochier (France), Nicky Broeckhoven (Belgium), An Cliquet (Belgium), Judy Fisher (Australia) , Thomas Hahn (Sweden), Eric S. Higgs (Canada), Anton Imeson (the Netherlands), Amanullah Khan (Pakistan), Ritesh Kumar (India), German Kust (The Russian Federation), Blanche Lormeteau (France), Prasanta Kumar Mishra (India), Joe Morris (United Kingdom of Great Britain and Northern Ireland), Stephen D. Prince (USA), Rajesh K. Rai (Nepal), Alka Sabharwal (India), Giulia Wegner (Italy), Louise Willemen (The Netherlands)

Review Editors

Susan Galatowitsch (United States of America), Florencia Montagnini (Argentina/United States of America)

This chapter should be cited as:

Pandit, R., Parrota, J., Anker, Y., Coudel, E., Diaz Morejón, C. F., Harris, J., Karlen, D. L., Kertész, Á., Mariño De Posada J. L., Ntshotsho Simelane, P., Tamin, N. M., and Vieira, D. L. M. Chapter 6: Responses to halt land degradation and to restore degraded land. In IPBES (2018): The IPBES assessment report on land degradation and restoration. Montanarella, L., Scholes, R., and Brainich, A. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem services, Bonn, Germany, pp. xx-xx.

Table of Contents

Executive summary

The most cost-effective approach to reduce land degradation in the long run is to follow the adage "prevention is better than cure" ***(well established)*** **{6.3.1, 6.3.2, 6.4.2}.** The economic consequences of land degradation are significant. For example, a study of fourteen Latin American countries estimated annual losses due to desertification at 8-14% of agricultural gross domestic products (AGDP), while another study estimated the global cost of desertification at 1-10% of annual AGDP. Across all biomes, estimates of the ecosystem service values lost due to land degradation and conversion range from $4.3 to $20.2 trillion per year. In a global study that considered values of forests for wood, non-wood products, carbon sequestration, recreation and passive uses, it was estimated that the projected degradation and land-use change would reduce the value of these forest ecosystem services by $1,180 trillion over a 50-year period, between 2000 to 2050 {6.4.2.3}. However, a broad range of sustainable land management, soil and water conservation practices, and nature-based solutions, have been effective in avoiding land degradation in many parts of the world *(well established)* {6.3.1, 6.3.2}. For example, agroecology, conservation agriculture, agroforestry and sustainable forest management can successfully avoid land degradation, while enhancing the provision of a range of ecosystem services *(well established)* {6.3.1.1, 6.3.2.3}. Many of these same techniques and measures can also be used to restore degraded lands, but may be more costly than their use for avoiding land degradation *(well established)* {6.3.1, 6.3.2}.

There are no "one-size-fits-all" biophysical and technical responses for avoiding and reducing land degradation, nor for restoring degraded lands ***(well established)*** **{6.3.1, 6.3.2, 6.4.2}.** Actions to avoid or reverse land degradation (of croplands, forests, rangeland, urban land, wetlands) – or to deal with the adverse impacts of invasive species, mineral extraction activities, deterioration of soil health and water quality and climate change – are more effective when they are designed to fit local environmental, social, cultural and economic conditions *(well established)* {6.3.1}. Key considerations for response actions include: the types and severity of degradation drivers and processes affecting the land {6.3.2}; past and present land uses and their socio-economic contexts; and institutional, policy and governance environments {6.4.2} (*well established*). Further, the effectiveness of these actions is often enhanced by the integration of indigenous and local knowledge and practices (*well established*) {6.4.2.2, 6.4.2.4}.

Direct biophysical and technical responses, and their effectiveness to address land degradation drivers and processes, depend on the nature and severity of drivers and the prevailing enabling environment ***(well established)*** **{6.3.2}**. Responses to land degradation due to invasive species include identifying and monitoring invasion pathways and adopting quarantine and eradication (mechanical, cultural, biological and chemical) measures *(well established)* {6.3.2.1}. Responses to land degradation from mineral resource extraction include: on-site management of mining wastes (soils and water); reclamation of mine site topography; conservation and early replacement of topsoil; and passive and active restoration measures to recreate functioning grassland, forest and wetland ecosystems *(well established)* {6.3.2.2}. The responses to invasive species and mineral extraction-related degradation are successful where restoration plans are fully implemented and monitored following an adaptive management approach.

Conservation agriculture, agroecology, agroforestry and traditional practices are effective ways to use and manage soil and land resources sustainably ***(well established)*** **{6.3.1.1}.** These management practices can be effective in reducing soil loss and improving soil quality, as well as other biogeochemical functions and processes in soils including: biological productivity; hydrological processes; filtering; buffering and nutrient cycling; and habitat quality for soil and above-ground organisms and communities {6.3.1.1, 6.3.2.4}. A strong commitment to continuously monitor the quality of soil resources is needed to improve management decisions that consider not only short-term economic gains, but also off-site and long-term consequences.

Effective responses to rangeland degradation include land capability and condition assessment and monitoring, grazing pressure management, pasture and forage crop improvement, silvopastoral management, and weed and pest management *(well established)* {6.3.1.3}. These biophysical responses are generally effective in halting rangeland degradation, but the effectiveness can be enhanced by aligning these responses with social and economic instruments (*well established*) {6.3.1.3}. For example, historic nomadic pastoral grazing practiced on the Egypt-Israel border has been found to be more effective for maintaining rangeland resources than year-round livestock husbandry in pastoral farm and village settings. Shepherd communities of the Jordan Valley have avoided the degradation of pasture land through restrictions on their herds' mobility, with the establishment of new national boundaries throughout the 20th century. The ability of the stationary pastoral rural communities to maintain systematic or semi-systematic grazing and rangeland development regimes also improve their resilience to climate change {6.3.1.3}.

The effectiveness of responses to wetland degradation and water quality degradation depend on the adoption of integrated soil and water management techniques and their implementation *(well established)* {6.3.1.5, 6.3.2.4}. The effective responses to avoid or reverse wetland degradation include controlling point and non-point pollution sources by adopting integrated land and water management strategies and restoring wetland hydrology, biodiversity and ecosystem functions through passive and active restoration measures such as constructed wetlands *(well established)* {6.3.1.5}. Similarly, effective responses to improve water quality include soil and water conservation practices, controlling pollution sources and desalination of wastewater *(established but incomplete)* {6.3.2.4}.

Responses to halt urban land degradation and to improve the liveability in urban areas include improved planning, green infrastructure development, amelioration of contaminated soils and sealed soils, sewage and wastewater treatment, and river channel restoration *(well established)* {6.3.1.4}. The effectiveness of these responses to minimize urban land degradation depends on the context as well as effective implementation. In developed countries, where large urban populations are concentrated, catchment-level natural capital and/or ecosystem service approaches have been proven to be effective in reducing flood risk and improving water quality through the restoration of biodiversity and use of sustainable land management techniques *(established but incomplete)* {6.4.2.3, 6.4.2.4}.

Enabling and instrumental responses address indirect drivers of land degradation and create conditions to enhance effectiveness of direct biophysical and technical responses (*well established*) {6.4.1, 6.4.2, 6.4.5}. A range of enabling and instrumental responses are available to avoid, reduce and reverse land degradation, and address its indirect drivers (e.g., economic and socio-political). These include a variety of legal and regulatory, rights-based, economic and financial, and social and cultural policy instruments such as: customary norms and support for indigenous and local knowledge; strengthening of anthropogenic assets such as research and technology development, skills and knowledge development; and institutional reform (*well established*) {6.4.2}. For example, the application of appropriate legal and regulatory instruments - and the establishment of appropriate governance structures and the devolution of power - have enabled successful restoration or rehabilitation of degraded forest lands and watersheds, in many parts of the world {6.4.2.1, 6.4.2.4, 6.4.5}.

The benefits of taking action (restoring degraded land) are higher than the costs of inaction (continuing degradation) *(well established)* {6.4.2.3}. For example, a study of large-scale landscape restoration in Mali found that adapting agroforestry is economically beneficial, providing direct local benefits to farmers of $5.2-5.9 for every dollar invested over a time horizon of 25 years. Investments in restoration can also stimulate job creation and economic growth. In the USA for example, the average number of jobs created per $1 million invested in restoration programmes has been estimated to be 6.8 for local-level wetland restoration, 33.3 for invasive species removal, and 39.7 for national-level forest, land and watershed restoration. The direct employment of 126,000 workers in restoration projects in the USA generates $9.5 billion in economic output annually - which indirectly creates an additional 95,000 jobs and $15 billion in annual economic output. The employment multiplier for restoration activities in the USA ranges from 1.5 to 2.9, comparable to that of other sectors, including the oil and gas industry (3.0), agriculture (2.3), livestock (3.3) and outdoor recreation (2.0) {6.4.2.3}.

More inclusive analyses of the short-, medium- and long-term costs and benefits of avoiding and reversing land degradation can support sound decision-making by landowners, communities, governments and private investors *(established but incomplete)* {6.4.2.3}. Economic analyses that consider only financial or private benefits and utilize high discount rates favour less investment in sustainable land uses and management practices, while undervaluing biodiversity, ecosystem services, public values and intergenerational benefits. The incorporation of a broader set of non-marketed values in cost-benefit calculations - such as the provision of wildlife habitat, climate change mitigation and other ecosystem services - would encourage greater public and private investment in restoration projects *(established but incomplete)* {6.4.2.3}. Fulfilling land degradation neutrality objectives and large-scale restoration goals requires creating (economic) incentives that encourage landowners, land managers and investors to recognize and capture the public value of restoring degraded land, particularly in severely degraded landscapes.

The effectiveness of policy instruments depends on the local context, as well as the institutional and governance systems in place *(well established)* {6.2.2, 6.4.2}. A variety of instruments have been used to promote the adoption of sustainable land management practices and these have been generally effective {6.4.2}. Establishment of protected areas, as a legal/regulatory response, has been instrumental in avoiding land degradation across the world *(established but incomplete)*, but their effectiveness varies with context *(established but incomplete)* {6.4.2.5}. The area of production forestry under forest certification (eco-labelling) schemes such as the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification (PEFC) standards has increased in recent years {6.4.2.4}. Customary norms (local and indigenous practices) adopted by local communities have avoided land degradation and contributed to sustainable land management, for centuries {6.4.2.2}. While such practices are generally heterogenous and context specific, they are nearly always based on long-term experience and innovation, and in tune with local needs {6.4.2.4}.

The economic and financial instruments to avoid land degradation and to restore degraded land in order to provide ecosystem services and goods include: policy-induced price changes (i.e., taxes, subsidies); payments for ecosystem services; biodiversity offsets; improved land tenure security (establishing property rights); and the adoption of natural capital accounting to reflect the flow and stock value of natural assets in national accounts *(established but incomplete)* {6.4.2.3}. Tax measures which restrict land degrading behaviour and subsidies to promote land restoration activities have been mostly successful *(well established)* {6.4.2.3}. Effectiveness of emerging incentive schemes such as payments for ecosystem services (e.g., REDD+)

and biodiversity offsets are context dependent and hence are also sometimes in conflict with local norms and land management practices - requiring more evidence before upscaling these approaches *(established but incomplete)* {6.4.2.3}. Secure property rights are an essential and effective way to avoid land degradation in situations where these rights are not well defined *(well established*) {6.4.2.3}. Natural capital accounting as a response to land degradation is in its infancy, but is a promising tool for avoiding land (flow and stock) degradation by bringing the true value of land - including non-monetary societal values - into land management decision-making *(unresolved)* {6.4.2.3}.

Integrated landscape planning to address land degradation problems that involves both the private and public sector can successfully create synergies across relevant sectoral development policies while minimizing trade-offs *(established but incomplete)* {6.4.3}. This would typically involve: (i) the promotion of sustainable land management practices (arable and urban lands); (ii) community-based management and decision-making - including traditional and local practices; (iii) climate change adaptation planning; and (iv) enhancing effective corporate social responsibility approaches from private sectors in an integrated way (i.e., aligning with other sectoral development priorities) (*established but incomplete*) {6.4.2.4, 6.4.2.6, 6.4.3}.

Anthropogenic assets required to address land degradation and restoration needs (knowledge, capacities and resources) are unevenly distributed within, and especially between, countries and regions *(established but incomplete)* {6.4.4}. Gaps or inadequacies in knowledge and skills, capacity and resources among countries need to be addressed to halt land degradation and restore degraded lands {6.5}. Particularly, there is a need for capacity-building in sustainable land management, including efficient land information systems in many developing countries that are prone to and affected by land degradation {6.4.4}. However, while labour-intensive restoration approaches may be more feasible in countries with lower labour costs (such as in Asia and the Pacific), their application may be limited by the training or extension gaps required by local communities to implement such practices.

Institutional reform that enables community-based natural resource management and the utilization of both Western scientific and indigenous and local knowledge or practices have been proven effective for conserving forests, soils, wildlife (biodiversity) and water quality in developing countries *(well established)* {6.3.1.1, 6.3.1.2, 6.4.2.4, 6.4.5}. In Nepal, for example, the establishment of local Community Forest Users Groups have been highly successful in avoiding deforestation and forest degradations as well as restoring previously degraded forest landscapes {6.4.5}. In other countries and contexts, legal instruments and compliance mechanisms adopted by local authorities have been the preferred approach to avoid land degradation and to restore degraded lands, as for example in the case of the restoration of degraded watersheds in China’s Loess Plateau region {6.3.1.1}.

6.1 Introduction

The design and application of effective, preventive as well as mitigation responses to land degradation requires a thorough understanding of its drivers (Chapter 3), processes (Chapter 4) and impacts on human well-being (Chapter 5). Human responses to land degradation and restoration can be broadly grouped into enabling and instrumental responses (i.e., legislation, policy, institutions and governance systems) and direct biophysical and technical responses (i.e., on the ground actions). Because of complexity and site-specificity of land degradation and restoration responses, any type of human action must be based on the best available knowledge from all sources (i.e., natural and social science, indigenous and local knowledge) (Reed *et al.*, 2011; SRC, 2016a; SRC, 2016b). For responses to be effective in bringing desirable changes, they must be technically and environmentally sound, economically viable, socially acceptable and politically feasible (Hessel *et al.*, 2014).

Typical direct responses often include a wide range of conservation measures and land management practices that have been used to avoid or reduce land degradation (Liniger & Critchley, 2007). The effectiveness of these direct responses often depends on enabling and instrumental initiatives and policy instruments designed to halt land degradation and promote restoration (Geist & Lambin, 2002; Hessel *et al.*, 2014; Reed *et al.*, 2011). Those policy instruments include: (i) legal and regulatory rules; (ii) right-based instruments and customary norms; (iii) economic and financial incentives (e.g., taxes, subsidies, grants, or creation of new markets such as payments for ecosystem services); and (iv) social and cultural programmes (e.g., eco-labelling, education/training, corporate social responsibility and voluntary agreements).

Historically, various types of enabling, instrumental and direct responses have been applied to address land degradation drivers and processes under different situations. As stated by Lal *et al.* (2012), these mitigation or restoration responses have been applied individually, or in combination, at micro (e.g., farmer adoption of zero tillage practices) and macro scales (e.g., striving f5or a “land degradation neutral world” by the global community). Despite a growing knowledge base regarding drivers, processes and their interactions on both ecosystem services and quality of human life (i.e., food, feed, fibre, fuel supplies and social stability), progress towards effectively responding to land degradation remains a formidable challenge (Winslow *et al.*, 2011).

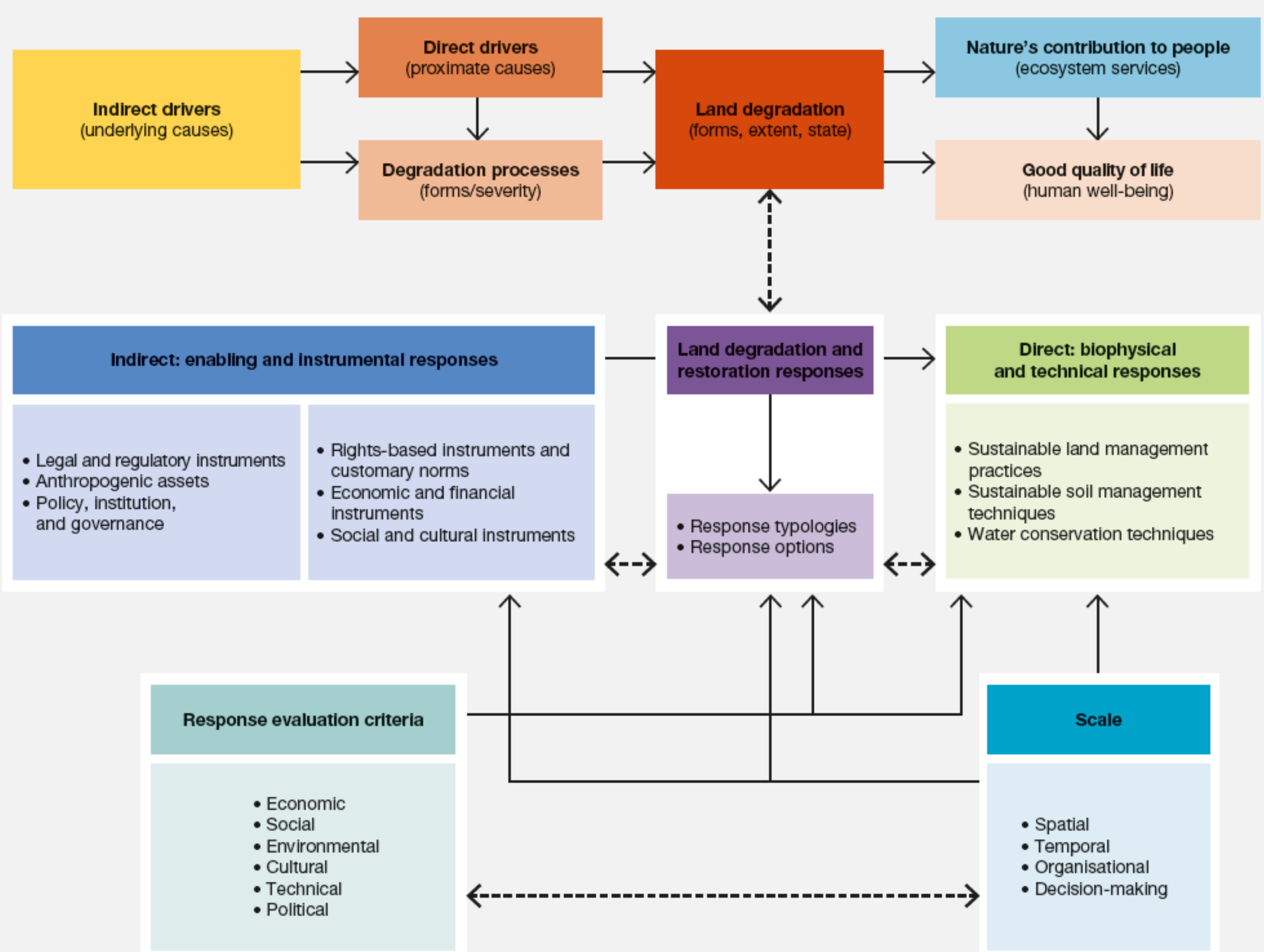
Consistent with the IPBES framework (Díaz *et al.*, 2015), this chapter focuses on critical evaluations of current response strategies; both their effectiveness for avoiding or mitigating land degradation and for restoring previously degraded lands are examined. More specifically, this chapter:

- Develops a chapter-specific framework to assess the effectiveness of existing interventions designed to avoid and reduce land degradation processes and to rehabilitate or restore various types of degraded lands (e.g., croplands, rangelands, forest lands, urban lands and wetlands) through the recovery of biodiversity, ecosystem structure and services. The ultimate goal is to enable the land to provide the essential functions needed to sustain human societies;
- Assesses how responses to land degradation and restoration vary according to site-specific characteristics, including the type and severity of degradation, underlying direct and indirect drivers, and effects on ecosystem services and quality of life;

- Evaluates the effectiveness of various response options to direct drivers (e.g., better land management techniques, access to training) and indirect drivers (e.g., institutions, governance systems) of land degradation;
- Examines the relative success of different institutional, governance and management response options to avoid, reduce and reverse land degradation across a range of economic, social, environmental, cultural, technical and political scenarios; and
- Assesses different institutional, policy and governance responses to research and technology development.

Recognizing that land degradation and restoration responses operate at different temporal, spatial, organizational and decision-making scales, we developed a chapter-specific conceptual framework (Figure 6.1) to evaluate the effectiveness of various response options based on the conceptual frameworks of IPBES (Díaz *et al.*, 2015) and the Economics of Land Degradation (Mirzabaev *et al.*, 2015).

Figure 6 1 Framework to evaluate effectiveness of land degradation and restoration responses, including prevention, mitigation and rehabilitation.



The dashed or two headed arrows in Figure 6.1 represent interdependencies between framework components, while the response criteria per se include: economic (feasibility, efficiency, effectiveness - on-/off- site, direct/indirect, present/future), social (equity - procedural/distributional, inclusivity, participatory, adoption potential), environmental (ecosystem function, ecosystem services, biodiversity, sustainability), cultural (compatibility with customary practice, local norms and values, indigenous and local knowledge and practices), technical (scientific skills and knowledge, technology), and political (acceptability, feasibility, policy, legal provisions and institutional support) considerations.

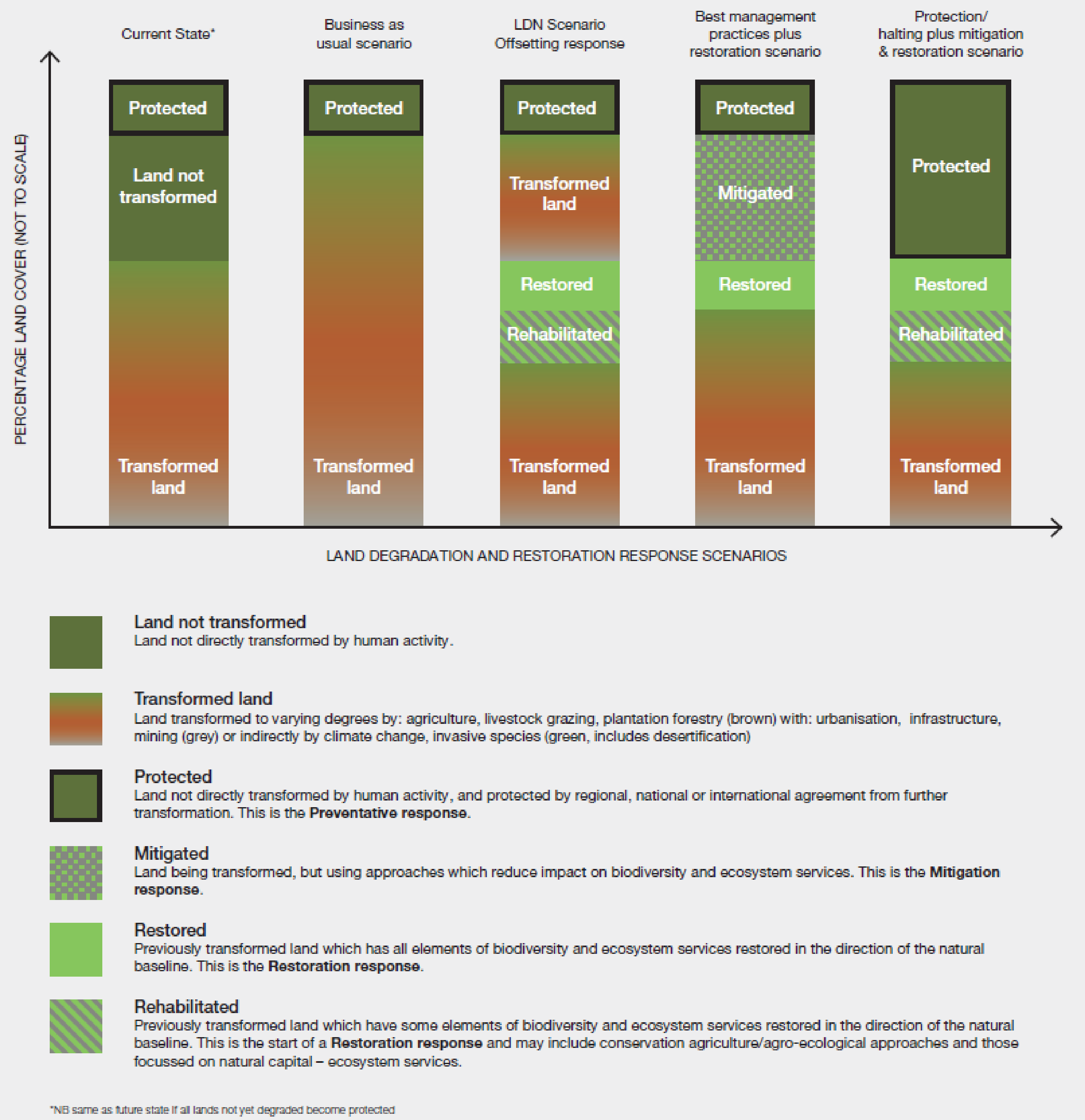
6.2 Response typology, options and evaluation framework

6.2.1 Response typology and options

To achieve land degradation neutrality, as stated in Target 15.3 of the Sustainable Development Goals, any response framework - which addresses biodiversity and ecosystem service impacts of land degradation - must consider the entire response hierarchy (i.e., prevention, mitigation, restoration and offsets). Furthermore, depending on the stage and severity of land degradation, the various drivers, processes and impacts will determine which enabling and instrumental and/or direct responses will be most effective for achieving land degradation neutrality and better scenarios (Figure 6.2, columns 3, 4 and 5).

Land degradation and restoration responses can be grouped into different typologies based on assessment needs. Response typologies can be developed based on: degradation drivers that need to be controlled; degradation processes that need to be halted or reversed; institutions that initiate the responses; types of responses that are applied to the drivers and processes (both direct and indirect); land-use categories that are affected by land degradation and need response actions; and the scale of responses - temporal (past, present), spatial and organizational (local, national, regional, global/international), and decision-making (household, community, private sector, public sector) levels.

Figure 6 2 Land cover type (not to scale) under different land degradation and restoration response scenarios.



Direct responses may seek to either avoid or reduce land degradation. Avoidance or preventive responses refer to conservation measures that maintain land and its environmental and productive functions, whereas reducing or mitigating responses are interventions intended to reduce or halt ongoing degradation and start improving the land and its functions. Reversing or restoration responses focus on the recovery of an ecosystem that has been degraded, damaged or destroyed (SERI, 2004). Offset refers to activities that compensate for residual degradation of biodiversity and ecosystem services, resulting in no-net loss in the ecological value of the impacted land (ten Kate *et al.*, 2004). In the cases where degraded land cannot be fully restored, offsetting becomes essential. Figure 6.2 shows plausible land degradation and restoration scenarios, based on the range of responses outlined in the legend. Each column in the Figure represents a unique scenario, ranging from the current state (column 1, which is same as the future state if all lands not yet degraded are prevented from becoming so) to a scenario that includes all forms of responses (column 5). The land degradation neutrality scenario with offsets is illustrated in column 3.

This chapter evaluates the effectiveness of various responses to halt land degradation and restore degraded land. Specific emphasis is given to land-use types (biomes) or complex degradation drivers and/or processes in assessing the responses. The responses are broadly grouped into two categories: enabling and instrumental, and biophysical and technical (MA, 2005; UK NEA, 2014). The enabling and instrumental responses include: legal and regulatory instruments; policy, institution and governance mechanisms; economic and financial instruments; social and cultural instruments; and rights-based instruments and customary norms. These responses seek to change or encourage human behaviour by creating a conducive environment for landholders, or other stakeholders, to operationalize biophysical and technical responses (i.e., land management practices).

Each response category has a range of appropriate response strategies depending on the form, severity and extent of degradation. Response options must be sensitive to both socio-economic and biophysical aspects of degradation and restoration strategies. Therefore, numerous options are available between enabling and instrumental responses as well as biophysical and technical responses (Liniger *et al.*, 2002; Liniger & Critchley, 2007). In practice, to achieve desired outcomes, land degradation responses need to be implemented simultaneously and in a coordinated fashion (Thomas, 2008) - using interdisciplinary and transdisciplinary perspectives which, in turn, help to fully evaluate the effectiveness of such responses (Reed & Stringer, 2015; STK4SD, 2015). Examples of synergistic response types include:

- Corrective methods (land rehabilitation and ecosystem restoration) that aim to halt and remedy degradation through, for example, conservation of soil and water, protection of vegetation, ecological engineering, and the re-establishment of functional ecosystems.
- Techniques to improve land use and management such as agroecology, agroforestry, conservation agriculture and other sustainable agricultural practices.
- Development of models and integrated natural resource management systems between local and national organizations.
- Implementation of favourable institutional, economic and political mechanisms. These may include: access to markets and sale of products from dry zones; diversification of rural economies; payment for ecosystem services; land ownership rights; access to credit; training for farmers; and insurance systems.
- Cooperation and knowledge exchange between land management, research and policy communities, as well as participatory approaches in research and development.

A detailed catalogue of sustainable land management approaches and technologies is available on the World Overview of Conservation Approaches and Technologies (WOCAT) website: https://qcat.wocat.net/en/wocat/ and in WOCAT publications (e.g., Liniger & Critchley, 2007). In Table 6.1, we present a set of land management strategies or response options illustrating the approaches and technologies outlined above.

Table 6 1 Biophysical and technical (direct) and enabling and instrumental responses to land degradation and restoration.

RESPONSE CATEGORY	MANAGEMENT STRATEGIES AND POLICY OPTIONS
DIRECT BIOPHYSICAL AND TECHNICAL RESPONSES	
Cropland degradation	Landscape approach; conservation agriculture; integrated crop, livestock and forestry systems; enhanced plant genetics; agroforestry; agroecology
Forest land degradation	Protected areas; restrictions on forest conversion; promotion of sustainable forest management practices; fire management; passive and active restoration
Rangeland degradation	Land capability and condition assessment and monitoring; grazing pressure management; pasture and forage crop improvement; silvopastoral management; weed and pest management
Urban land degradation	Improved planning; green infrastructure development; amelioration of contaminated soils and sealed soils; sewage and wastewater treatment; river channel restoration
Wetland degradation	Protected areas; control of point and non-point pollution sources; passive and active measures to restore hydrology, biodiversity and ecosystem function; constructed wetlands
Invasive species	Identification and monitoring of invasion pathways; quarantine measures; eradication measures; mechanical, cultural, biological, and chemical control
Mineral extraction	On-site management of mining wastes (soils and water); reclamation of mine site topography; conservation and early replacement of topsoil; passive and active restoration measures to recreate functioning grassland, forest and wetland ecosystems
Soil quality change	Improved agronomic practices; reduced tillage; increase diversity and vegetative cover in production systems; integrated crop, livestock and forestry systems; improved fertilizer and agrochemical use efficiency; improved irrigation and water use efficiency; reduce deposition of atmospheric pollutants
Water quality change	Integrated land and water management; rainwater harvesting; soil and water conservation practices; desalination wastewater treatment; constructed wetlands
ENABLING AND INSTRUMENTAL RESPONSES	
Responses to the adverse effects of globalisation, demographic change, migration	Trade and consumption; linking trade and environmental protection; voluntary product certification; population policies that interact with land such as resettlement, fertility rate, rural urban-migration
Legal and regulatory instruments	Land-use planning (national, regional, local); social and environmental impact assessments; incentives for sustainable land-use practices; establishment of protected areas
Rights-based instruments and customary norms	Improved land tenure security; clarification of natural resource-use rights; support for ILK-based traditional use practices
Economic and financial instruments	Policy-induced price changes; payments for ecosystem services; biodiversity offsets; improved land tenure security; clarification of natural resource-use rights; natural capital accounting
Social and cultural instruments	Participatory natural resource management and governance; support for ILK-based traditional use practices; eco-certification; promotion of corporate social responsibility;
Protected areas	Legal protection; private and community-based conservation; promotion of ILK-based traditional use
Climate change adaptation planning	Conservation of natural areas with high carbon stores (e.g., peatlands, old-growth forests, mangroves); land-use specific measures to reduce net greenhouse gas emissions; land-use specific adaptation measures
Integrated landscape planning	Sustainable land management; integrated planning and management; zoning
Anthropogenic assets	Capacity-building including: skills and knowledge development; research and technological development; extension; human resource development; infrastructure and facilities
Institutional and policy reform	Establishment of new institutions; strengthening existing institutions; mainstreaming Indigenous and Local Knowledge and Practices (ILKP); improving multi-level governance mechanisms

6.2.2 Response evaluation framework

Here, effectiveness is understood as a measure of the extent to which an activity accomplishes its objectives. Motivations of human behaviour and resilience capacity of natural systems are fundamental considerations when evaluating the effectiveness of land degradation and restoration responses. Based on the chapter-specific conceptual framework (Figure 6.1), a response evaluation framework is outlined in Table 6.2 for direct response options. The response evaluation framework considers a set of assessment criteria to evaluate the effectiveness of individual response options. Such assessment criteria include a range of economic, social, environmental, cultural, technical and political measures (Table 6.2). For example, from an environmental sustainability perspective, a response would be evaluated for its suitability to improve ecosystem functions, generate ancillary benefits (positive externalities) and its potential to address wider sustainability objectives. Similarly, from a technical feasibility perspective, a response would be evaluated on the basis of skill and knowledge requirements as well as the technological sophistication involved. For direct responses, the concept of response hierarchy is also used to evaluate response options - for instance whether a given strategy belongs to avoiding (prevention) or reducing (mitigation) land degradation or reversing (restoration) degraded land, or a combination of them. The effectiveness of response options can also be viewed on the basis of their speed and ease of implementation, time frame, acceptance by local stakeholders, endorsement by experts, institutional capacity, scale of benefits or number of beneficiaries (USAID, 2008).

Table 6 2 Template for assessment of the effectiveness of various response options by land-use types and degradation drivers.

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE Avoid (Av), Reduce (Rd), Reverse (Rv)	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING [High effectiveness (H), Moderate effectiveness (M), Low effectiveness (L), or any combinations: L to M, M to H, L to H]					
			Economic [feasibility, efficiency, effectiveness (on/off-site, direct/ indirect, present/ future), equity -process, distribution, spill-over effect]	Social [equity, inclusivity, participatory, potential to adopt]	Environmental [potential to address environmental sustainability concerns - water security, climate change, biodiversity conservation, ecosystem service provisions]	Cultural [customary practice, local norms and values, ILK]	Technical [skills/ knowledge, technology, sophistication]	Political [legal provisions, institutional structure, political acceptability/ feasibility]
CROPLAND MANAGEMENT	1. 2.	Av/Rd/Rv	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H
FOREST LAND MANAGEMENT	1. 2.	......	......	......	......	......	......	......
......	......	......	......	......	......	......	......	......

6.3 Direct biophysical and technical responses to land degradation and restoration

Land degradation and restoration responses are inherently context specific and such responses vary depending on the extent and severity of the drivers and processes, as well as specific biophysical characteristics of the place or system. In addition, on-the-ground restoration responses may depend on economic, social, cultural and technical factors. Use of case-specific analyses based on major land-use types (see Section 6.3.1) and selected drivers and processes (see Section 6.3.2) to provide an overview of the effectiveness of past and current responses to land degradation and restoration. To evaluate specific responses to the many land-use degradation drivers and/or processes, the following discussion will:

i. Identify specific land and soil management actions, based on both Western science and indigenous and local knowledge and practice (ILKP) that can halt land degradation;
ii. Specify which responses are preventive (i.e., capable of avoiding land degradation) and which are specific to mitigation (i.e., focused on reducing land degradation and reversing, rehabilitating and/or restoring degraded lands);
iii. Examine how well those responses are working and where (i.e., under what geographic, socio-economic and cultural settings);
iv. Provide examples of their effectiveness; and
v. Discuss what messages should be given to key stakeholders regarding the effectiveness of these responses.

6.3.1 Assessment of land-use specific responses

6.3.1.1 Responses to cropland degradation

Cropland soil degradation is very site specific and can occur physically, chemically and/or biologically. Potential responses to degradation include using: (i) a landscape approach; (ii) conservation agriculture; (iii) integrated crop, livestock and forestry systems; (iv) agroforestry; (v) enhanced plant genetics; and (vi) integrated watershed management.

Landscape approach

A landscape approach examines how soil resources, cropping systems, weather patterns, management practices, market development, community preferences and other factors affect ecosystem processes (Kosmas & Kelly, 2012). Indigenous peoples instinctively adopt a landscape approach as their connections to the land incorporate interactions across the landscape and understandings of the connections of all living things (Walsh *et al.*, 2013). The critical point for this response is that there is no single solution, because interactions of all these factors ultimately modify the entire landscape.

The Atlantic Forest Restoration Pact (Melo *et al.*, 2013) in Brazil provides an excellent example of the landscape approach (see Box 6.3). It demonstrated that continuous technology improvement, on-going teaching and community outreach, capacity-building, incorporation of local knowledge, a clear and transparent legal environment and effective economic instruments and incentives were all crucial for success. Other studies (e.g., Baker *et al.*, 2014; Norgaard, 2010) warn against blindly focusing on ecosystem services in lieu of ecological, economic and political complexities encountered when responding to land degradation.

Conservation agriculture

Conservation agriculture, as defined by the FAO, is characterized by three specific actions including: (i) continuous minimum mechanical soil disturbance; (ii) permanent organic soil cover; and (iii) diversification of crop species grown in sequences and/or associations. In general, conservation agriculture principles are universally applicable to all agricultural landscapes and land uses, because they emphasize the use of locally-adapted practices (based on ILKP), biodiversity and natural biological processes above and below ground (Forest People Program & Program, 2010). Interventions such as mechanical soil disturbance, and agrochemical or plant nutrient applications, are optimized so they do not interfere with or disrupt biological soil processes.

Global adoption of conservation agriculture has been increasing steadily (Friedrich *et al.*, 2012; Jat *et al.*, 2014; Reicosky, 2015) as documented by an FAO database that shows approximately 125 million hectares (8.8% of arable cropland) are now being managed using conservation agriculture. However, the FAO (2015) estimates a global growth of almost 32 million ha (26%) within the last five years. The primary limitations for the implementation of conservation agriculture include market pressure for monocrop production, climatic factors, access to conservation agriculture technology, appropriately scaled incentives and information regarding adoption (Jat *et al.*, 2014).

Two perceived conservation agriculture concerns are the high dependence on glyphosates and genetically modified plants. Regarding glyphosate, current safety evaluations have generally not indicated serious risks for human or environmental health (Williams *et al.*, 2000), although concerns persist among some public health researchers (Vandenberg *et al.*, 2017) as well as the International Agency for Research on Cancer (IARC), the specialized cancer agency of the World Health Organization, which classified glyphosate as “probably carcinogenic” to humans in 2015 (International Agency for Research on Cancer, 2015). Nonetheless, Health Canada recently determined that when used according to label directions, products containing glyphosate are not a concern to human health or the environment (Pest Management Regulatory Agency, 2017). Also, implementing conservation agriculture practices does not require the use of genetically modified plants, but rather minimum mechanical soil disturbance, permanent organic soil cover and diversity in crops grown.

The impact of conservation agriculture is illustrated in Table 6.3 which shows several countries with at least 14% of their arable cropland being managed using conservation agriculture practices. Argentina currently has the highest rate of adoption at 74%, and 90% of the 32 million ha increase during the last 5 years is accounted for by data from six countries (Table 6.4). Furthermore, data for India - which was not previously reported (Jat *et al.*, 2014) - accounted for a 1500 ha increase in conservation agriculture. We concur that adoption of conservation agriculture can be an effective preventive and mitigation strategy for addressing global cropland degradation.

Table 6.3 Countries with at least 10% of arable cropland within conservation agriculture. Source: (FAO, 2016).

Country	Conservation Agriculture (1000 ha)	Percent of Arable Cropland	Data Year
Argentina	29,181	74	2013
Paraguay	3,000	63	2013
Uruguay	1,072	44	2013
Brazil	31,811	44	2012
Canada	18,313	40	2013
Australia	17,695	38	2014
New Zealand	162	32	2008
United States of America	35,613	23	2009
Chile	180	14	2008

Table 6.4 Countries with largest recent increases in conservation agriculture. Calculated from values presented by Jat *et al.* (2014) and FAO (2015)

Country	Conservation Area Change (1000 ha)	Data Years
United States of America	+9113	2009, 2007
Brazil	+6309	2012, 2006
Canada	+4832	2013, 2006
Argentina	+3628	2013, 2009
China	+3570	2013, 2011
India	+1500	2013, none previous
Australia	+695	2014, 2008
Paraguay	+600	2013, 2008
Uruguay	+417	2013, 2008
Kazakhstan	+400	2013, 2011

Integrated crop, livestock and forestry systems

Another strategy for restoring degraded cropland (sometimes referred to as sustainable intensification) is to incorporate perennials and cattle into traditional row-crop production systems. In Brazil, sustainable intensification began slowly during the 1970s, as cattle production on native grass and bush lands within tropical savannahs became more extensive. Adaptation of new cattle breeds (mostly Nellore) and grasses such as *brachiaria* led to the development of integrated crop and livestock and integrated crop, livestock and forestry systems. These systems not only increased food and feed production at farm and regional levels, but also improved many ecosystem services (Carvalho *et al.*, 2017; Salton *et al.*, 2014; Sato & Lindenmayer, 2017).

Integrated crop and livestock has been used to restore degraded croplands in North America, Western Europe, Brazil, Uruguay and Argentina (Franzluebbers *et al.*, 2014; Peyraud *et al.*, 2014). Integrated crop and livestock - and integrated crop, livestock and forestry - have increased the amount of cultivated pasture in Brazil to nearly 101 million ha as compared to 57 million ha of native pasture. Although this is impressive, it accounts for only 32-34% of the estimated 274 -293 million animal units that could be produced in Brazil (Strassburg *et al.*, 2014). Striving for full adoption would not only result in substantial restoration of degraded croplands, but also enable Brazil to readily meet human demand for meat, crops, wood products and biofuel feedstocks until at least 2040, without any additional conversion of natural ecosystems (Strassburg *et al.*, 2014).

Agroecology

Agroecological practices encompass a broad array of agricultural technologies that take advantage of natural processes and beneficial on-farm interactions in order to reduce off-farm input use and to improve the productivity and efficiency of farming systems, enhance food security by diversifying crop production and managing environmental and economic risks, and avoid agricultural land degradation (Altieri, 2002; Gliessman, 2014; Pretty *et al.*, 2003) (see also Chapter 2, Section 2.2.4.3 and Box 2.4). Such systems, based largely on indigenous and local knowledge, have been developed and used worldwide by farmers. They typically involve management practices such as cover crops, green manures, intercropping, agroforestry and crop-livestock mixtures that promote organic matter accumulation and nutrient cycling, soil biological activity, natural control mechanisms (disease suppression, biocontrol of insects, weed interference), resource conservation and regeneration (soil, water, germplasm), and general enhancements of agrobiodiversity and synergisms between components (Altieri, 2002; Gliessman, 2014). Agroecological initiatives in many countries in Africa, Asia and Latin America - often promoted by NGOs - have had a demonstrably positive impact on farmers' livelihoods (Altieri *et al.*, 2012; Altieri & Toledo, 2011; Pretty *et al.*, 2003; Pretty *et al.*, 2011) (see also Chapter 5, Section 5.3.3.1 and Box 5.5). Success of such initiatives has been found to depend on human capital enhancement and community empowerment - through training and participatory methods as well as access to markets, credit and income generating activities, and supportive government policies (Markwei *et al.*, 2008; Pretty *et al.*, 2003; Pretty *et al.*, 2011)

Agroforestry

Agroforestry can reduce or reverse land degradation by: (i) maintaining soil fertility through increased carbon inputs, nitrogen fixation and nutrient cycling; (ii) reducing erosion; and (iii) conserving water (quantity and quality) through increased infiltration and reduced surface runoff. It can also conserve biodiversity, improve air quality, reduce reliance on fossil fuels and native forests for fuelwood, help adapt to climate change, and provide economic, social, cultural and aesthetic benefits (Murthy *et al.*, 2016). Agroforestry practices are for the most part rooted in ILK and emphasize the preservation of knowledge, local crop varieties and animal breeds, as well as native socio-cultural organizations (Lemenih, 2004; SRC, 2016b, 2016c). Innovative agroecosystem designs have been modelled on successful ILK-based practices (Altieri & Toledo, 2011; Brondízio, 2008) and it is estimated that, worldwide, as many as 500 million people practice some form of agroforestry (Nair *et al.*, 2009; Zomer *et al.*, 2014).

Box 6.1 Agroforestry responses to cropland degradation (adapted from Nair, 1993)

Agroforestry systems are typically classified on the basis of their structure (i.e., the nature and spatial and/or temporal arrangement of tree and non-tree components). They include:

- *Agrisilvicultural* - encompasses a diverse array of practices involving cultivation and management of trees and/or shrubs for food and/or non-food uses. Generally, in combination with agricultural crops, these subsystems include improved fallow (in shifting cultivation and rotational cropping), multilayer tree gardens and alley cropping. They also include different plantation crop combinations that are used not only for timber and fuelwood, but also as fruit trees within home gardens;
- *Agrosilvopastoral* - which uses domesticated animals, multipurpose woody hedgerows, apiculture, aqua-forestry and multipurpose woodlots in combinations with home gardens and fish ponds; and
- *Silvopastoral* - systems which include plantation crops, animals grazing pasture or rangeland and protein banks which produce concentrated, protein-rich tree fodder outside standard grazing areas.

Agroforestry systems are globally diverse and are widely practiced in:

- Humid and sub-humid tropical lowland regions, where they can help reduce deforestation and forest degradation. In these areas, they overcome productivity constraints of soil degradation caused by unsustainable forest management, poorly managed shifting cultivation, overgrazing, soil acidity, low soil fertility and high rates of soil erosion;
- Tropical and sub-tropical highlands, humid and sub-humid regions in the Himalayans, parts of southern India and Southeast Asia, highlands of east and central Africa, Central America, the Caribbean, and the Andes, where productivity and food security is often constrained by soil erosion, insufficient fallow periods, overgrazing, deforestation and forest degradation, as people seek fodder and fuelwood; and
- Semi-arid and arid regions where lack of precipitation, climatic change and increasing populations exceed the capacity of native forests and pastures.

A wide range of ILK-based agroforestry approaches have been used successfully in many parts of the world (Lahmar *et al.*, 2012; McLean, 2010; Parrotta & Trosper, 2012; Suárez *et al.*, 2012; Uprety *et al.*, 2012; Vieira *et al.*, 2009). In the Sahel, degraded lands have been restored using ILK techniques developed and applied by innovative farmers seeking to reverse desertification and preserve their agropastoral livelihoods (Behnke & Mortimore, 2016) (see also Chapter 4, Section 4.2.6.2). In Burkina Faso, 200 to 300 thousand ha of severely degraded farmland have been rehabilitated by combining ILK soil conservation measures and protecting on-farm trees (Botoni & Reij, 2009; Reij *et al.*, 2005; Reij *et al.*, 2009; Tougiani *et al.*, 2009). Similarly, in southern Niger, traditional agroforestry parklands have increased significantly across nearly 5 million ha through farmer-managed natural regeneration of a variety of native tree species (Reij *et al.*, 2009).

Agroforestry can be very important for mitigating and adapting to climate change in regions facing both land degradation and food security challenges (Mbow *et al.*, 2014; Parrotta & Agnoletti, 2012; Verchot *et al.*, 2007), because it provides poor farmers with alternative pathways to increase productivity and food security (Lasco *et al.*, 2014; Mbow *et al.*, 2014). It also has considerable potential for carbon sequestration (Albrecht &

Kandji, 2003), because the above- and below-ground carbon density of typical tropical agroforestry systems is estimated at 12 to 228 Mg ha^{-1}, with a median value of 95 Mg ha^{-1} (Albrecht & Kandji, 2003). For smallholders, potential carbon sequestration rates generally range from 1.5 to 3.5 Mg C ha^{-1} yr^{-1} (Montagnini & Nair, 2004). The potential of agroforestry to serve as a carbon sink, however, depends on the climatic zone conditions and silvicultural practices including planting density, species choice and length of rotation (Nair *et al.*, 2010).

In summary, agroforestry-based land restoration initiatives are relevant for the planning and/or monitoring of national and international policy objectives related to landscape restoration and biodiversity conservation, due to their potential for: (i) recognising and incorporating indigenous and local knowledge; (ii) combining social development and ecological conservation and restoration objectives; and (iii) fostering cross-sectoral collaboration between local communities, governmental agencies, NGOs, universities and research institutions (Altieri, 2004; Altieri & Toledo, 2011; Chirwa & Mala, 2016; Nair, 2007; Norton, 1998; Ouédraogo *et al.*, 2014; Parrotta *et al.*, 2015; Powell *et al.*, 2013; Walker & Macdonald, 1995)

Use of Enhanced Plant Genetics

The use of drought-resistant crop varieties by smallholder farmers to adapt to climate change and soil degradation in several African countries has been quite successful (Fisher *et al.*, 2010; Tschakert, 2007). By including pulses in mixed cropping systems, water-use efficiency and nutrient cycling were improved (Valentin *et al.*, 2008). Implementation of such practices could reduce global anthropogenic CO_2 emissions by 6 to 17% (Van Der Werf *et al.*, 2010); confirming that good agricultural management can increase productivity and carbon sequestration, while also reducing carbon emissions (West & Marland, 2003). Therefore, combining improved plant genetics with decreased tillage and efficient use of fertilizer and irrigation water can not only increase soil organic carbon, but contribute to climate change mitigation (Lal, 2002).

Integrated Watershed Management

Integrated watershed management provides another strategy to meet global demands of more than 9 billion people by the middle of the twenty-first century. Decreasing tillage frequency and intensity coupled with restoring or increasing soil organic carbon are two mitigation/restoration strategies that have been successfully demonstrated at the watershed scale (Box 6.2).

Box 6.2 Restoration of Degraded Watersheds: an example from China's Loess Plateau. Source: Liu & Hiller (2016); World Bank (2007).

The Loess Plateau in Northwest China occupies approximately 640,000 km^2 and is the dominant geological feature in the middle reaches of the Yellow River basin. The plateau has been inhabited for more than 8,000 years (Peng & Coster, 2007; Wang *et al.*, 2006). The forces that have driven landscape, vegetation and hydrological changes in the Plateau include the dual effects of human land use and climate change (Ren & Zhu, 1994; Saito *et al.*, 2001; Shi, 2002). The plateau's forest cover dropped down to 7–10%, from historical estimates of 50% (Cai, 2002; Liu & Ni, 2002) and 70% of the plateau is affected by soil erosion, 58% of which is extremely severe (Chen *et al.*, 2007) - with soil erosion rates among the highest in the world (Fu, 1989). In addition to downstream sedimentation and eutrophication problems (Wang *et al.*, 2006), dust storms (Luo *et al.*, 2003) and landslides (Zhou *et al.*, 2002) have also been problematic.

From 1994 to 2005, two Loess Plateau Watershed Rehabilitation Projects were implemented in 48 counties in the Shanxi, Shaanxi and Gansu provinces, and the autonomous region of Inner Mongolia. Rehabilitations of physical activities were performed over 35,000 Km^2 and with a total investment of $550 million.

A key factor leading to success in the Grain for Green Program was the integrated watershed management that created effective water harvesting structures. They were crucial for continuous vegetative cover in the large-scale reforestation, grassland regeneration and agroforestry activities (EEMP, 2013). Another, was the significant financial investment that included direct Chinese government expenditures and World Bank loans. This financing provided subsidies for farmers enabling them to restore degraded farmland by planting trees and other vegetation. The subsidies included $122/hectare for seeds and seedlings as well as annual payments for ecosystem services of $49/hectare for two to eight years (Buckingham & Hanson 2013). Specific actions that contributed to the project's success included:

Pre-rehabilitation actions

Project planning - which spanned over 3 years, integrating economic and social well-being of the people with the ecological health of the environment.

Land-use mapping - to optimize selection of cropland versus land left to regenerate naturally.

Adoption of new policies - including bans on planting steep slopes, cutting trees and allowing free range grazing (all to enable re-establishment of local vegetation).

Community participation - emphasizing local input into rehabilitation programmes.

Responses during rehabilitation

Technical - including hard and soft engineering for sustainable water management, terracing and dam construction in deep valleys for erosion and sediment control. Dam construction was continued until the entire gully bottom consisted of flat fields and rich productive croplands that increased farmer income, quality of life and discouraged them from planting on steep slopes.

Greening activities - which stabilized dunes using straw and plantings of grasses, bushes, trees and perennial cash crops.

Post-rehabilitation Responses

Buckingham & Hanson (2013) summarized several positive benefits including:

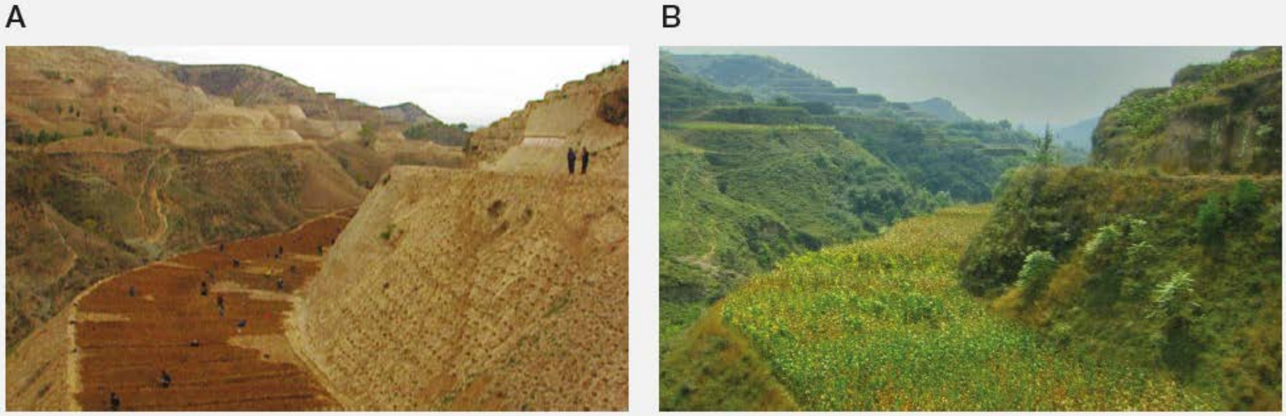
- Increased per capita grain output from 365 to 591 kg ha^{-1} yr^{-1}
- A 95% conversion of sloping land to improved land uses
- A 159% increase in community income
- New infrastructure and development opportunities
- Terracing of ~86,600 ha of new farmland
- A decrease in farming of unstable sloped lands from 451,000 to 278,000 ha
- A 99% decrease in sediment (~300 million tons yr^{-1}) deposited into the Yellow River
- Establishment of ~290,000 ha of shrub and economically valuable trees

Additional benefits of the Grain for Green Program have been reported by Cheng *et al.* (2016); Deng *et al.* (2014); Liang *et al.* (2012); Tsunekawa *et al.* (2014); and Wang *et al.* (2016).

Community development

The Grain for Green Program has resulted in profound lifestyle changes and has benefited many benefits for local people, in a variety of ways. Local communities now enjoy better facilities, infrastructure and amenities, including roads, clean water, electricity, schools, hospitals, new housing and township developments.

Figure 6 3 The Ho Family Gully on the China Loess Plateau before [A late August 1995] and after [B late August 2009] the "Grain for Green" conservation program. Photo Credits: Liu & Hiller (2016).



6.3.1.2 Responses to forest land degradation

Responses to deforestation and forest degradation include preventive measures, the integration of production with conservation objectives (through agroforestry, natural and planted forest management) and restoration. Countries with low or negative deforestation rates have either managed their forests sustainably or restored degraded lands based on one or more of these strategies.

Avoiding deforestation, forest fragmentation and forest degradation

Avoiding deforestation and reducing forest fragmentation is particularly important for forest ecosystems that are still largely intact. It is both more cost-effective and conserves more biodiversity than is possible through restoration, at least in the medium term (Benayas *et al.*, 2009). While the establishment of protected areas has frequently been the only mean to conserve large intact forest areas, other landscape-planning strategies that have been effective in avoiding deforestation, including restrictions of agricultural expansion in ecologically-fragile areas and biodiversity hotspots, and intensification of agriculture in fertile and geomorphologically stable areas (Chazdon *et al.*, 2009; Lambin & Meyfroidt, 2011).

Deforestation can be avoided with controls over domestic and international markets for agricultural products where the supply chain for these products contributes to forest loss and degradation (Macedo *et al.*, 2012). For example, the Soy Moratorium in Brazil, in which traders agreed not to purchase soy from lands deforested after July 2006 in the Brazilian Amazon, resulted in a decrease in annual soy expansion into forested areas from 30% to 1% after 2006 - although expansion of soy cultivation into pastures and cleared land increased (Gibbs *et al.*, 2015), and potential leakage effects of the Soy Moratorium on the Brazilian savannahs and other countries have yet to be assessed.

Many intact (formally or informally protected) forest areas are embedded within human-modified landscapes (Melo *et al.*, 2013), where agriculture and urbanization have significantly modified landscape structure. This is often accompanied with declines in biodiversity due to dis-connectivity among remaining forest patches (Rappaport *et al.*, 2015) and with limited potential to avoid further species loss (Fahrig, 2003). Effective measures to address the negative biodiversity impacts of forest fragmentation require evaluation of the condition and attributes of remaining forest remnants (i.e., their size, shape, degree of isolation, and habitat quality and heterogeneity) and the land-use matrix in which they are embedded (Collinge, 1996).

Landscape planning (discussed further in Section 6.4.3) is an important tool for developing effective actions to avoid further deforestation and/or ameliorate forest fragmentation impacts and through conservation and restoration measures (Banks-Leite *et al.*, 2014; Tambosi *et al.*, 2014). Effective and widely-used measures to increase connectivity, conserve biodiversity and enhance delivery of ecosystem services within fragmented forest landscapes include: maintenance of vegetation corridors in riparian vegetation (Naiman *et al.*, 1993); establishing new fragments or expanding the size of existing ones through restoration (Brancalion *et al.*, 2013); and promoting agricultural practices such as agroforestry in areas surrounding intact forests (Chazdon *et al.*, 2009; Cullen *et al.*, 2001).

Payments for ecosystem services (see Section 6.4.2.3) can also promote sustainable forest management practices, particularly through the REDD+ mechanism (Reducing Emissions from Deforestation and forest Degradation), which has generated innumerable programmes worldwide - involving donors, consultants, experts, policymakers, researchers and communities (Corbera & Schroeder, 2011; Lund *et al.*, 2017). However, the effective implementation of REDD+ and other PES programmes hinges on the resolution of a number of issues related to: local conflicts among stakeholders regarding trade-offs between carbon sequestration and many of the other environmental, economic, social and cultural services provided by forests; community rights; independence from funding; and finding market funds to pay for the ecosystem services (Cadman *et al.*, 2016; Lund *et al.*, 2017; Parrotta *et al.*, 2012).

Firewood and charcoal for cooking and heating represents 55% of global wood harvest, which supplies 2.8 billion people (Bailis *et al.*, 2015) and 11.3% of the global energy demand (Guo *et al.*, 2015). Excessive firewood harvest is a significant driver of forest degradation in many countries (also see Chapter 3, Section 3.4.4.2 and Chapter 4, Sections 4.3.4 and 4.3.5). That said, forests and woodlands can and often are managed sustainably, and firewood demand is in some cases met through the use of by-products from commercial timber harvests (Bailis *et al.*, 2015; Chidumayo & Gumbo, 2013).

Over the last 40 years, concerns over the role of firewood extraction in tropical deforestation and the wood fuel shortages have prompted policy and programme interventions in many developing countries to reduce wood fuel demand and/or increase supplies, or some combination of the two. For the most part, these policy and programme interventions have failed to effectively deal with the problem of charcoal-based deforestation and its associated environmental concerns (Chidumayo & Gumbo, 2013). Nonetheless, some governments - having recognized the importance of firewood and charcoal as a principal source of energy - have sought to regulate and stimulate its sustainable production, especially given that it utilizes a local (and potentially renewable) resource and can generate local income (Chidumayo & Gumbo, 2013).

In some regions, wood fuels are being replaced by cleaner and healthier energy sources, including lignocellulosic bioethanol and biogas (Guo *et al.*, 2015). The environmental, social and economic impacts of land-use changes associated with increased production and other biofuels are the subject of considerable debate (Dai *et al.* 2011; Fargione *et al.*, 2008; Hasenheit *et al.*, 2016; Lambin & Meyfroidt 2011; Saez de Bikuña *et al.*, 2017; Whalen *et al.*, 2017).

Conserving and managing secondary forests

Secondary forests are a major part of many rural landscapes (Aide *et al.*, 2013; Hurtt *et al.*, 2006) and are increasingly recognized as important contributors of goods and services (Bongers *et al.*, 2015; ITTO, 2002), as is the need to incorporate them into land-use planning to balance conservation, production and sustainable livelihood needs. Their high potential to sequester carbon needs to be considered in public policies (Chazdon *et al.*, 2016; Poorter *et al.*, 2016), as well as their ability to restore forests at smallest costs (Bongers *et al.*, 2015). Secondary forests are often managed under adaptive and multiple-use management, not only for timber to provide short-term economic benefits, but also for food and other non-timber products through enrichment plantings with early production species, such as annual crops, fruit trees, palms and bamboos (ITTO, 2002). Managing secondary forests as productive agroforestry systems can be used to conserve biodiversity, limiting modification of the native vegetation, integrating ecosystem services schemes with benefits to local livelihoods (Mukul & Saha, 2017). Such management practices, relying heavily on indigenous and local knowledge, can be found throughout the world (Parrotta *et al.*, 2015).

Sustainable logging

Many criteria and indicators have been developed to guide sustainable forest management (Mendoza & Prabhu, 2003; Pearce *et al.*, 2003), including a comprehensive guide for reduced impact logging and sustainable management of tropical forests (ITTO, 2009; ITTO, 2016). These criteria and indicators are also used in forest certification, a market-based initiative aimed at promoting sustainable forest management (see Section 6.4.2.4). However, in countries where they would be particularly useful, these tools have not been extensively applied because of low consumer demand for sustainably-produced timber. Globally certified forest areas represented 11% of the world's forest cover in 2016, but 87% of certified forests were in the Northern Hemisphere and only 1.2% were in Africa, 3.1% in Oceania and 1.9% in Latin America (UNECE/FAO, 2016). Ninety percent of internationally-verified certification is in the boreal and temperate climatic domains, whereas only 6% of permanent forests in the tropics have been certified up to 2014 (MacDicken *et al.*, 2015).

Commercial and non-commercial planted forests

Planted forests are seen as a degradation driver, particularly when they replace natural forests (Brockerhoff *et al.*, 2008) (also see Chapter 4, Section 4.3.4). However, with the growing demand for wood products, planted forests have become a complementary measure to conserve natural forests when established on degraded lands. In fact, planted forests have reduced harvesting from natural forests globally by 26% (Buongiorno & Zhu, 2014). They currently produce 5 to 40 times more timber yield than certified natural forests (Paquette & Messier, 2010) and supply a quarter of global industrial roundwood production, while occupying only 7% of the world's total forest area (Payn *et al.*, 2015). Reducing potential negative effects and/or enhancing positive effects of establishing planted forests requires rigorous impact assessments that consider the changes in biodiversity and ecosystem services, as well as design and management measures that help to protect biodiversity. Such measures include: setting aside natural habitats along watercourses and establishing biodiversity reserves within large-scale plantation areas; utilizing or further developing silvicultural knowledge to expand the use of native species in planted forests; and adjustments to silvicultural practices to favour local biodiversity in planted forest stands and avoid introducing invasive tree species and/or their pests and diseases (ITTO, 2009).

Forest restoration

Significant opportunities exist to restore forest cover, biodiversity and ecosystem services on formerly forested degraded lands and abandoned agricultural sites (Benayas *et al.*, 2009). According to an analysis conducted by the World Resources Institute and the Global Partnership on Forest Landscape Restoration, more than two billion hectares could potentially be restored worldwide - including 1.5 billion ha considered best-suited for mosaic restoration, in which forests and trees are combined with other land uses such as agroforestry, smallholder agriculture and settlements - and up to about half a billion hectares are suitable for wide-scale restoration of closed forests (Minnemeyer *et al.*, 2011).

A variety of effective reforestation and forest management techniques are used to varying extents to restore forests in degraded landscapes, depending on ecological circumstances and management objectives (Lamb *et al.*, 2005).

These include:

- Protection of natural regrowth from fire, grazing and other stressors inhibiting secondary forest development;
- Protection of natural regrowth and enrichment with commercially, socially or ecologically valuable tree species to improve the economic and social value of these forests;
- Restoration plantings (or direct seeding) using a small number of short-lived nurse trees to accelerate natural regrowth, applicable to sites and landscapes with nearby natural forests that may serve as seed sources;
- Restoration plantings using large number of species from later successional stages, useful for sites lacking nearby natural forest seed sources and/or to promote desired forest structure and species composition;
- Tree plantation mixtures of native species;
- Tree plantation used as a nurse crop with under-plantings of native species not otherwise able to establish at the site;
- Tree plantation monoculture of native tree species; and
- Tree plantation monoculture of non-invasive exotic species.

To optimize biodiversity conservation and enhance the provision of forest ecosystem services, restoration efforts should be planned at the landscape level (Maginnis & Jackson, 2003; McGuire, 2014).

Governments can effectively support forest ecosystem restoration by providing financial and policy support for development of planted forests on previously degraded lands. For example, the central government of the Republic of Korea worked in close collaboration with communities and succeeded in increasing the country's forest area from approximately 35% to 65% between 1955 and 1980. Their approach included a combination of economic incentives and policy coordination, particularly between the forestry and energy sectors to replace firewood with fossil fuels, a process assisted by rural-urban migration (Bae *et al.*, 2012; Park & Youn, 2017) (see also Section 6.4.1 on demographic changes and restoration). By enhancing the profitability of a forest-based economy - through commercialization of timber and non-timber forest products, shaded crops and ecotourism - some governments have contributed to forest conservation efforts while enhancing their benefits to people (Calvo-Alvarado *et al.*, 2009; Chazdon *et al.*, 2009). Livelihood improvements in rural areas that facilitate the transition from firewood to coal or electricity can reduce forest degradation, thereby contributing to land restoration (Dube *et al.*, 2014; Sugiyama & Yamada, 2015).

Responses to forest fire

Fire is most commonly viewed as a driver of forest degradation, but it is also used as a management tool in forest and grassland ecosystem management, particularly by local and indigenous communities (Parrotta & Trosper, 2012) (also see Chapter 3, Section 3.4.6 and Chapter 4, Section 4.2.6.5). For example, the utilization of traditional fire management practices in northern Australia have been shown to yield multiple benefits, not only for the environment to reduce degradation and assist restoration by making landscapes less prone to large wildfires, but also for traditional people (Legge *et al.*, 2011; Russell-Smith *et al.*, 2003; Vigilante *et al.*, 2004).

Two complementary approaches to fire management are commonly used, namely integrated fire management and community-based fire management (FAO, 2011). Integrated fire management focuses on addressing underlying causes for long-term and sustainable solutions, incorporating the five essential elements (research, risk reduction, readiness, response and recovery) and thus integrating all activities related to fire management (FAO, 2011).

Community-based fire management includes the integration of science and fire management approaches with socio-economic elements, at multiple levels, and provides a comprehensive approach to address fire issues that considers biological, environmental, cultural, social, economic and political interactions (Myers, 2006). It involves local-scale fire management, community and volunteer involvement in fire management across private and public lands (FAO, 2011).

While fire suppression is often cost effective for containing small-scale fires, such an approach can increase the future risk of much more damaging fires, especially in forests adapted to low to moderate intensity fire regimes (Stephens *et al.*, 2013). Managing forests for other values will be futile in the long term without managing forest for long-term fire risks and resilience (Jones *et al.*, 2016; Stephens *et al.*, 2013; Tempel *et al.*, 2015).

Box 6.3 Restoration of the Brazilian Atlantic Rain Forest

The Atlantic forest is among the top five global biodiversity hotspots (Laurance, 2009), providing a range of ecosystem services including drinking water for more than 60% of Brazil’s population. However, more than 88% of the original forest has disappeared, largely due to deforestation and agriculture (Pinto *et al.*, 2014), making it one of the highest priority regions for restoration in the world.

The Atlantic Forest Restoration Pact, initiated in 2009, is a regional, multi-stakeholder platform formed by NGOs, research institutions, the private sector and government agencies to coordinate efforts and objectives for restoration (Brancalion *et al.*, 2016; Melo *et al.*, 2013). It links key stakeholders for knowledge sharing and connects those offering or requesting sites for restoration, as well as inputs and technical assistance. The Pact aims to facilitate and implement restoration projects across 17 Brazilian states. It manages both public funds allocated by government budgets and ODA as well as private funds obtained through payments for ecosystem services, offset schemes for Brazilian infrastructure mitigation, water user fees, compensation payments for restoration, grants and microloans for establishing alternative sources of income (Sewell *et al.*, 2016).

The Pact aims to make ecosystem restoration an economic activity - generating opportunities for business, employment and income for local communities, especially in less developed areas. Under the Pact tens of thousands of hectares of forest areas have already been restored, with a long-term target of restoring 15 million ha out of the total Atlantic Forest area of 132 million ha. Restoration goals include: conserving forest biodiversity and enhancing delivery of ecosystem services; reconnecting isolated forest fragments; and re-establishing forests to promote sustainable harvest of timber and non-timber products. A variety of active and passive restoration approaches and methods are being used to conserve small- and medium- sized, privately-owned fragments and restore small areas around protected zones to improve the connectivity of landscapes (Holl, 2017; Pinto *et al.*, 2014; Rodrigues *et al.*, 2011).

6.3.1.3 Responses to rangeland degradation

An estimated 73% of the world's 3.4 billion ha of rangeland is affected by degradation of soils and vegetation (WOCAT, 2009) (see also Chapter 3, Section 3.3.1 and Chapter 4, Section 4.3.2). Rangeland degradation and species loss is mainly caused by overstocking of livestock combined with poor grazing management by nomadic pastoralists and smallholder farmers (e.g., Bestelmeyer *et al.*, 2011).

Strategies to improve grazing land management have been applied at different spatial scales, from global transboundary regional planning and implementation – through governmental control of stocking rates, livestock types and water allocation – to local approaches involving rotation of pastures, controlled burning, fencing and pasture development through replanting, intercropping and removal of woody plants (Latawiec *et al.*, 2017; Reid & Swiderska, 2008). In addition, several indigenous pastoral projects indicated that grazing management systems can also be achieved. Successful strategies include tribal and community coordination and cooperation, integrated and sustainable land use (Haregeweyn *et al.*, 2012; Kong *et al.*, 2014), and hunting to mitigate overgrazing by wild livestock (Gibson & Marks, 1995).

Developing and implementing grazing management plans is an efficient response to avoid and reduce rangeland degradation in particularly sensitive parts of the landscape (e.g., slopes, water points, riparian strips) and for soil and water conservation. Key considerations for effective rangeland management planning include:

- *Land condition* - rainfall and natural runoff pattern, soil fertility and health and pasture biodiversity (both feedstock and livestock) (Bartley *et al.*, 2010);
- *Anthropogenic community structure* - development level of agriculture and municipal infrastructures, level of governmental regulatory capabilities, indigenous and local practices, local stakeholders and land tenure rights (Undersander *et al.*, 2014);
- *Grazing level and distribution* - pasture utilization, stocking rate influence, grazing system and livestock type (Undersander *et al.*, 2014); and
- *Diet gateway* - conversion of pasture into animal product, through herbage quality, legume content and pasture species (Fisheries & Forestry, 2013).

Implementation of grazing land management strategies may involve a combination of existing tools appropriate for specific grazing and pasture management scenarios (Lambin *et al.*, 2014). Effective tools for different pasture types typically consists of:

- *Spatial information monitoring* - which can utilize national and regional governmental data archives and remote sensing resources to assess key features, such as property mapping, paddock size, land types, land use and more. Spatial monitoring is an effective tool for regions that are prone to soil erosion and rangeland degradation, due to overgrazing along slopes, particularly in drier regions (Bartley *et al.*, 2010). Utilization of such available databases, and temporal and spatial analyses, can indicate trends such as vegetation cover, desertification, land uses and other physical parameters essential for rangeland management (Prince, 2016).

- *Land capability and condition assessments* - through field surveys when databases are insufficient. These should include key features, such as specific land capability, land conditions, means of sustaining and improving land conditions, current carrying capacity, potential carrying capacity and more (see Chapter 4, Section 4.3.2).

- *Land resource and use characterization* - including grazing and pasture development parameters, namely land type, fencing, water points, frontages, wetland management, biodiversity conservation measures, legislative responsibilities, tree-grass balance management, wildfire prevention and fire control.

- *Grazing pressure management* - involving economic and regulatory means to control stocking rates, timing livestock growth, herd sizes, grazing management zones and maintain more uniform pasture pressure (Bartley *et al.*, 2010). Effective application of such tools is often difficult as it typically requires coordination and regulation among authorities and other key stakeholders (i.e., pastoralists and farmers) (Latawiec *et al.*, 2017).

- *Pasture and forage crop, enhancement* - through development and management of pasture and forage crops, silvopastoral practices, prevention of sown pasture degradation and development of monitoring tools. Although most pasture and forage crops are grown in cultivated areas, if grazing exhausts natural rangeland, replanting using rangeland vegetation enhancement techniques is needed to preserve their fertility (Undersander *et al.*, 2014).

- *Weed and pest management* - through monitoring, management and control of invasive plants, insects and other pests. The incorporation of indigenous peoples' traditional knowledge and rangeland management practices provide additional approaches for effective weed and pest management (Ens *et al.*, 2015).

- *Evaluation of social and economic potential* - for the adoption of more sustainable pasture management practices, including land tenure types and cultivation systems (e.g., farms, nomadic, rural settlements), as well as cultural aspects such as cattle sanctity (India), the integration of the land uses in local traditions and evaluations of the magnitude and effectiveness of governmental actions (e.g., taxation, law-enforcement) for the relevant community (Latawiec *et al.*, 2017; Reed *et al.*, 2015).

Finally, the assessment of grazing land management strategies should consider effects of each strategy on financial and technological capabilities of local farmers and their economic benefits, the level of local authorities' regulatory management capabilities and, above all, effects of the strategy on physical parameters of the grazing land (Weber & Horst, 2011).

Box 6.4 Grazing control and desertification in arid zones (Egypt-Israel-Jordan)

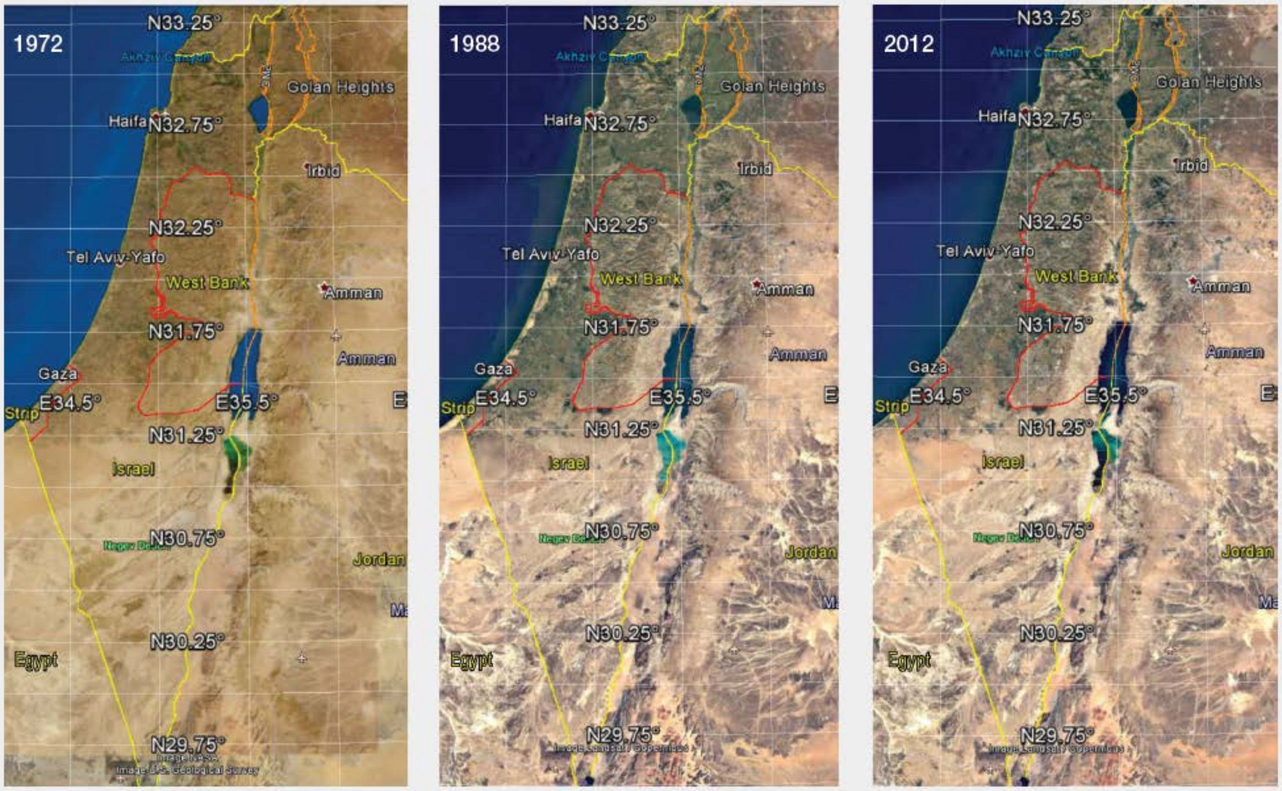
Throughout history, the cultivation of camels, sheep and goats played a major role in Eastern Mediterranean economies, through the sale of their meat, dairy or hair and wool products. During the last couple of centuries most herds were driven by tribes of pastoral nomads, known as Bedouin (Bienkowski & van der Steen, 2001). Until the 20th century, by permit of the Ottoman empire these nomads had access to transboundary traditional pastoral resources; but since the early 20th century - through a series of international treaties and the establishment of new States - several tribes were restricted to the North-Western Sinai Desert. This pasture land restriction gradually degraded the rangeland owing to chronic overgrazing (Meir & Tsoar, 1996), manifested in the albedo difference between both sides of the Egypt-Israel border (Figure 6.4).

Once natural pasture carrying capacity is exceeded by livestock demands (see also Chapter 4, Section 4.3.2), rangeland development actions are required. The dynamic nature of the process is well demonstrated by the temporal shift in vegetation density across the Egypt-Israel border (Warren, 2002). While vegetation density was similar during the years when the border was open (mainly during the 1970s) (Figure 6.4), since 1982 the closed border has been a barrier to grazing herds and, as a result, the vegetation density increased on the Israeli side compared to the Egyptian side of the border (Seifan, 2009). The desert dunes' stability, owing to the development of soil crusts, contributes to landscape resilience against natural phenomena such as large-scale dust storms (Figure 6.4) (Kidron *et al.*, 2017).

While along the Egyptian-Israeli border the disruption of grazing pastoral practice had led to deterioration of natural and human habitats, along the Israeli-Jordanian border (the Jordan Rift Valley) the Jordan River floodplain supplied sufficient rangeland resources, preventing the pasture over-burden. In addition, the Jordan Valley is one of the first locations with documented human settlements and probably the first evidence of livestock farming (Lu *et al.*, 2017; Martínez-Navarro *et al.*, 2012).

One of the differences between the nomadic pastoral grazing typical to the Egypt-Israel border, and the year-round livestock husbandry in pastoral farm and village setting, is better management of rangeland resources. The stationary nature of the Jordan valley shepherds community prevented the overgrazing of pasture land. The ability of the stationary pastoral rural communities to maintain systematic or semi-systematic grazing and rangeland development regimes improved their resilience to climate change and political issues.

Figure 6 4 Comparative satellite view (Google Earth) of the Egypt-Israel-Jordan borderlines in 1972, 1988 and 2012 respectively.



6.3.1.4 Responses to urban land degradation

Amongst the most severe forms of land transformation, urbanization results in land degradation both within and outside of urban areas - through its direct impacts on lands within established and expanding cities and suburban areas and the extension of their ecological footprints beyond their boundaries - leading to impacts on a wide range of ecosystems in surrounding landscapes.

Figure 6 5 Urban and suburban landscapes in Medellin, Colombia: the planned city A, the informal city B, and the quarries C. Source: Medellin Planning Department, 2006.

Responses to reduce these impacts include those that seek to: maintain or improve the health and sustainability of ecosystems within their zones of influence; the health, well-being and safety of urban dweller; and to improve the urban fabric.

Preventive responses to urban land degradation

Responses to urban land degradation fall into two categories, “grey” and “green” responses. Regarding “grey” responses, the New Urban Agenda (http://habitat3.org/the-new-urban-agenda/) incorporates sustainability as its third principle and 56 sustainable urban development commitments (Caprotti *et al.*, 2017; Watson, 2016). Out of these commitments, 3 contain responses to ecological-rural functionality; 3 to water management, mainly as an economic resource; 3 to the green public space, with emphasis on its social function and resilience factor; and 43 to technical and political responses to social and economic problems. Specific “grey” responses to achieve these commitments include urban planning and design instruments to support sustainable land-use management and natural resources by enhancing resource efficiency, urban resilience and environmental sustainability (amongst others).

Figure 6 6 Aerial view of the rooftop garden of a multi-storey carpark in Singapore.

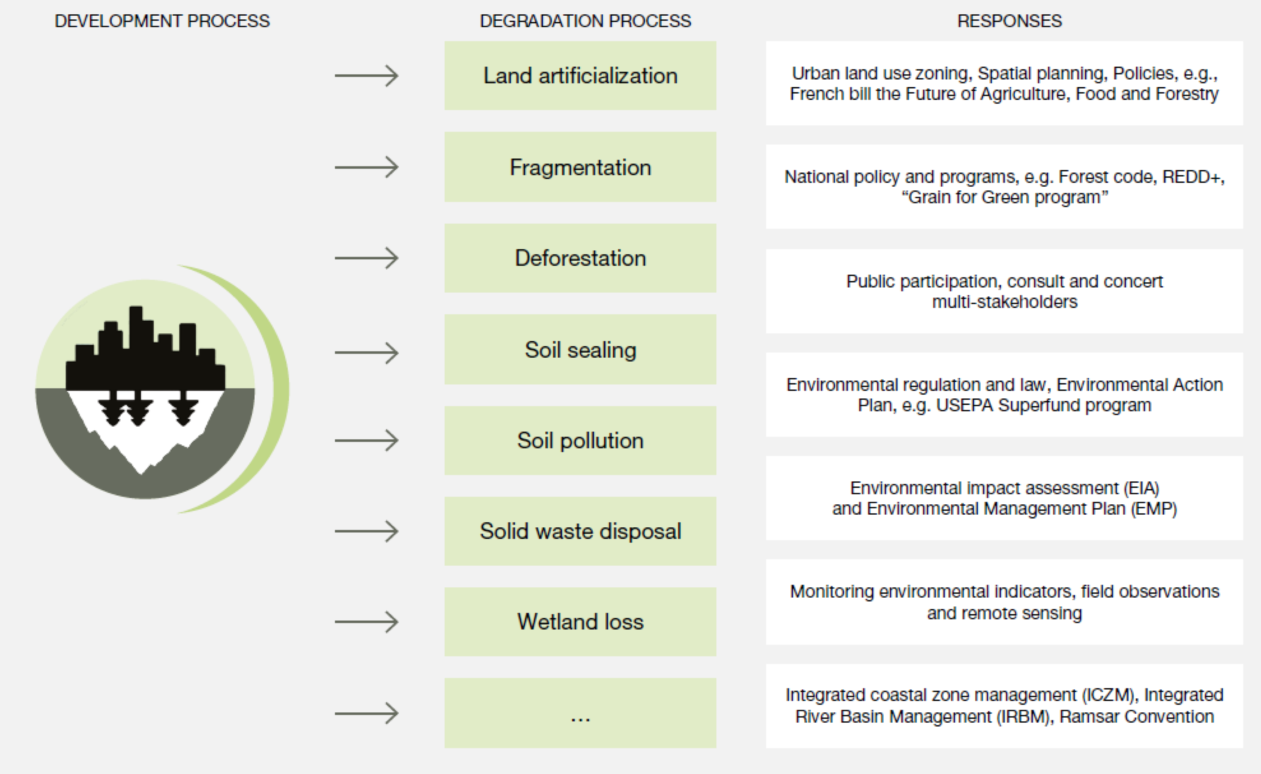
Among the many techniques used to create "green infrastructure" in urban areas, rooftop gardens are one. Photo: Jimmy Tan licensed under CC BY 2.0.

On green responses, the Cities and Biodiversity Outlook of the CBD (2012) highlights opportunities to reduce urban land degradation by utilizing the linkages between urbanization, biodiversity and ecosystem services. Response measures include developing and enhancing existing ecological infrastructure of cities (i.e., parks, gardens, open spaces, water catchment areas), and their ecosystems and biodiversity. It emphasizes the importance of valuation and explicit inclusion of urban biodiversity (also known as natural capital) as a determining factor in the planning and management of cities. Maintaining functioning urban ecosystems not only addresses the problems associated with urban land degradation, but can also significantly enhance human health and well-being as well as contribute to climate change mitigation and adaptation (CBD, 2012). Sustainable urban development includes managing and designing for biodiversity conservation (Aronson *et al.*, 2017; Müller & Kamada, 2011). "Green infrastructure" is widely proposed and, in some places, widely implemented (Hostetler *et al.*, 2011) - using techniques such as planting vegetation on roofs ("green roofs", Figure 6.6), rain gardens, paving with materials that allow infiltration of precipitation protected natural open space, planting native plant species and retaining corridors of non-developed land. These provide habitat for native plants, insects, animals and soil biota (McKinney, 2002).

Restoration practices in urban and built environments

Specific responses to urban land degradation depend on the main issues or processes that need to be addressed, such as soil contamination and soil instability, water contamination, invasive species impact, heat island effects and flooding risk from altered catchment hydrology (Figure 6.7).

Figure 6 7 Land degradation and restoration related policy challenges, goals, instruments and tools and methodologies.



In-built environments restoration practices are closely related to erosion and sediment control during the construction phase to prevent pollution of streams and rivers. Short-term erosion control practices are generally followed by establishment of vegetation for long-term erosion control. Bio-technical stabilization uses structural and biological elements to avoid severe erosion (Buchholz & Madary, 2016; Myers, 1993). These may include non-vegetated structures, such as retaining walls, or soil bio-engineering (the use of plants in bio-technical slope stabilization as the main structural component).

Soil contamination, a process by which the chemical properties of soils are changed, occurs mainly from industrial development in cities through factories releasing wastes that contain heavy metals, organic pollutants and other contaminants to surrounding areas (see also Chapter 4, Section 4.2.4.2). While soil contamination is rarely reversible (Siebielec *et al.*, 2010), it is sometimes possible to use brownfields to produce non-alimentary crops for energy or textiles. In this way, the past industrial soils recover new functions and their imperviousness is reduced (Huot *et al.*, 2015). However, the costs associated with remediation of past pollution in brownfields can be an obstacle to their re-use (EC, 2012). In such cases, financial compensation from the past polluters or the future developers is an approach to restore or improve the function of those soils.

Soil sealing is prevalent where materials such as asphalt, concrete and stone are used to construct buildings, roads, parking lots and other urban infrastructure (see also Chapter 3, Section 3.3.6). Sealing reduces or completely prevents natural soil functions and ecosystem services on the area concerned, including regulation of hydrology and temperature regimes in urban areas (EEA, 2011). Measures to compensate for soil sealing include: (i) re-use of topsoil excavated during building construction and infrastructure development in other urban locations; (ii) de-sealing of certain areas (soil recovery) to compensate for sealing elsewhere; (iii) use of eco-accounts and trading development certificates; and (iv) collection of fees on soil sealing activities, to be used for soil protection or other environmental purposes (EC, 2012). Some financial approaches can also help restore contaminated land, such as the "Superfund" programme of the US Federal government, which has funded decontamination of sites contaminated with hazardous substances and pollutants since 1980 (Acton, 1989; Daley & Layton, 2004).

Increasing urban populations and impervious surfaces intensify heat island effects in cities (also see Chapter 4, Section 4.3.10). Responses to reduce heat island effects include developing or maintaining "green infrastructure," such as urban open spaces and urban forestry initiatives that include tree planting and management (Gill *et al.*, 2007; Miller *et al.*, 2015; Roy *et al.*, 2012). The importance of street trees, urban forests and their multiple benefits is increasingly recognized by urban planners, municipal governments and citizens worldwide (Pandit & Laband, 2010; Pandit *et al.*, 2014) and many cities have made urban greening a priority. Many urban greening tools have been developed, such as the Berlin Biotope Area Factor, the Malmo Green Factor (Hagen & Stiles, 2010), the Seattle Green Factor (Giordano *et al.*, 2017), the Poland Ratio of Biologically Vital Area (Szulczewska *et al.*, 2014) and a public open space planning tool (Bull *et al.*, 2013).

Water system degradation can threaten many cities. Filling rivers and lakes to develop real estate or infrastructure, for example, can alter flow regimes and increase flood risk. As this process is largely irreversible and often very costly, better land-use planning is essential (Hall *et al.*, 2014; Shen, 2015). In addition, water contamination and pollution from industrial wastewater or domestic sewage can have severe impacts on environmental quality and its related services. Water contamination can often be handled as part of brownfields projects - although law enforcement, filtration of wastewater before discharge and education are also effective ways to alleviate water pollution (Buchholz & Madary, 2016; Hall *et al.*, 2014; Kjellstrom *et al.*, 2006; Myers, 1993; Shen, 2015).

Methods to respond to altered catchment hydrology include river channel restoration and management of impervious surfaces through the reduction and adoption of technologies to improve infiltration in parking lots and transportation corridors, and installation of rain gardens. Urban forestry can also aid in hydrologic management through canopy interception. New soil media for cities can also be developed to create soil from waste and thus avoid agricultural soil consumption (Rokia *et al.*, 2014). Quantifying the economic value of green infrastructure can also promote restoration activities or maintenance of green infrastructure in urban areas. For example, Polyakov *et al.* (2017) report that restoration practices aiming to convert a "conventional drain" into a "living stream" in Perth simultaneously increased property price (private economic benefit) and the ecological outcomes such as better habitats for plants and animals (a public benefit), thus providing additional incentives for urban residents or the local authorities to restore degraded urban drains.

There are no panaceas for the urban land degradation issues and processes, and governments in different contexts must consider their financial, technological or political capacities to appropriate select restoration responses. Table 6.5 gives an overview of the effectiveness of different responses to halt or restore degraded urban land.

6.3.1.5 Responses to wetland degradation

Worldwide, the extent of wetlands is estimated to have declined by 64-71% in the 20th century (Davidson, 2014; Gardner *et al.*, 2015; Hu *et al.*, 2017). For several wetland types, such as tropical and subtropical mangroves, recent losses have been as high as 35% since 1980, with a current global area rate of loss of between 0.7 and 3% yr^{-1} (Pendleton *et al.*, 2012). The loss of these freshwater and coastal ecosystems have been estimated to result in more than $20 trillion in annual losses of ecosystem services (Costanza *et al.*, 2014). Consequently, the status of wetland-dependent species remains alarming. The Freshwater Living Plant Index has declined by 76% between 1970 and 2010 (Gardner *et al.*, 2015) (see also Chapter 4, Section 4.2.5.2).

The "wise use" approach of the Ramsar Convention is considered globally as a central tenet of wetland management (Maltby, 2009). Adopted by 169 countries, it builds on the premise that restricting wetland loss and degradation requires the incorporation of linkages between people and their surrounding wetlands (Finlayson *et al.*, 2011; Finlayson, 2012)). The removal of the stressors or pressures that limit the wise use of wetlands (or adversely affect their ecology) is considered the best practice response option for addressing wetland loss and degradation. The Convention has also developed a suite of guidance to support wetland restoration, including a specific resolution on avoiding, mitigating and compensating for wetland losses (Ramsar, 2012).

Ecological restoration of degraded wetlands is a global priority for addressing and reconciling conservation and sustainable development goals (Alexander & McInnes, 2012; Aronson & Alexander, 2013). Successful restoration of wetlands results in self-sustaining and resilient ecosystems dominated by native species (in characteristic assemblages and functional groups) that are part of a wider landscape in which the drivers of wetland degradation have been reduced or eliminated (SERI, 2004).

The most commonly-used responses to restore wetlands include recovering the hydrological dynamics, revegetating, removing invasive species and managing soil profiles. Restoring the hydrological dynamics usually involves either reconnecting the wetland to the tides or river flow (flow re-establishment), or reconstructing the wetlands topography (through surface modification). There has been considerable effort directed toward wetland restoration in some regions. Until 2014, the Wetland Reserve Program in the USA was a voluntary programme for landowners to protect, restore and enhance wetlands, resulting in nearly 1 million ha enrolled (USDA, 2014). In 2014, the first year of the Agricultural Conservation Easement Program, which replaced the Wetland Reserve Program, 168 wetland projects were supported covering about 15,000 ha (Smith *et al.*, 2015; USDA, 2014).

A recent meta-analysis of global wetland restoration (Moreno-Mateos *et al.*, 2012) - involving over 600 restored wetlands - found that those where either surface modification or flow re-establishment were used followed similar recovery trajectories, regardless of whether they were revegetated or not. It also found potential detrimental effects of revegetation measures on the recovery of the plant assemblage in cold climates and in wetlands restored in agricultural areas. This study also concluded that remediation efforts had failed to fully recover wetlands over the first 50 to 100 years (Moreno-Mateos *et al.*, 2012) with recovery of biodiversity and functions increasing to about 75% of the level in undisturbed reference wetlands after that time. Compared to degraded wetlands, however, restoration increased some ecosystem services and biodiversity, but the recovery was highly context dependent (Meli *et al.*, 2014). A study focused on recovery from eutrophication showed that lakes and coastal marine areas achieved a recovery of baseline conditions by an average of 34% and 24%, respectively, decades after the cessation or partial reduction of nutrients (McCrackin *et al.*, 2017)

These results indicate that there is an urgent need to understand how wetlands recover over the long term (20 years or longer) and what actions are most appropriate to restore them. As commonly used indicators of wetland recovery after restoration tend to be very simplistic (e.g., carbon storage), and do not encapsulate the complexity of ecosystems, there is a need to develop and use indicators to evaluate interactions among organisms and with the abiotic environment, for example, through measuring and recovering ecological networks (Anker *et al.*, 2013) with major roles in ecosystem functioning (e.g. decomposition, pollination, dispersal).

In recent decades, efforts to restore coastal wetlands (mangroves, tidal marshes and seagrass beds) have been made in many parts of the world to compensate or mitigate losses resulting from management activities (Hogarth, 2007; Lewis III, 2000; Orth *et al.*, 2012). Efforts have also been made to restore their capacity to provide ecosystem services such as buffering against extreme events (Marois & Mitsch, 2015). Methods for restoring such wetlands may include: active restoration measures (reshaping topography, channelling water flow, mangrove planting and control of invasive species); passive restoration approaches to enhance ecohydrological processes and improve hydrological connectivity; or in certain cases, the creation of wetlands (Zhao *et al.*, 2016). Complementary programmes in coastal planning (based on integrated coastal zone management approaches), marine spatial planning and marine protected areas have been established to address spatial issues. Recent research on economic efficiency of nature-based solutions has shown promising results. For example, maintenance of salt-marshes and mangroves have been observed to be two to five times cheaper than a submerged breakwater for wave heights up to half a metre and, within their limits, become more cost-effective at greater depths. Nature-based defence projects also report benefits ranging from reductions in storm damage to reductions in coastal structure costs (Narayan *et al.*, 2016).

Peatlands form a major proportion of total wetland area in the world and account for a major proportion of global soil carbon stores. Degradation of peatlands contributes significantly to global emissions of greenhouse gases (for example see Hooijer *et al.*, 2010). A range of measures for improving habitat conditions (e.g., regulating nutrient availability, base saturation, introduction of native species), peatland hydrology (e.g., increasing natural rewetting, damming and infilling of ditches, and reducing evapotranspiration) andcatchment management practices have been used in different parts of the world (Andersen *et al.*, 2017; Chimner *et al.*, 2017; Graham *et al.*, 2017).

Wetland creation and rewetting of drained soils are common activities in response to significant wetland loss and degradation on a global scale (Mitsch *et al.*, 1998). Wetland creation – where lands are artificially inundated and utilize natural processes to restore vegetation, soils and their associated microbial assemblages (Aber *et al.*, 2012) – is carried out for various purposes such as water-quality enhancement (treatment of wastewater, stormwater, acid mine drainage, agricultural runoff), flood minimization and habitat replacement (Mitsch *et al.*, 1998). Wetlands created for treating wastewater have been used with good results in many countries, including Cuba, China, USA and Thailand (IPCC, 2014; Land *et al.*, 2016; Vymazal, 2011). Recent advances in the design and operation of these wetlands have greatly increased contaminant removal efficiencies (Wu *et al.*, 2015). Wetlands may also be created unintentionally when the regulation of river flows (i.e., installation of large dams) results in periodic inundation of lands that previously did not experience inundation (Yang *et al.*, 2012).

Addressing the indirect drivers of change often requires policy-level changes, in the form of national policies on wetlands, or mainstreaming the full range of wetland ecosystem services and biodiversity values within sectoral policy and decision-making. Treating wetlands as natural water infrastructure can help meet a wide range of policy objectives such as water and food security and climate change adaptation (Pittock *et al.*, 2015; Russi *et al.*, 2013). Similar mainstreaming approaches, as wetlands as settings for human health (Horwitz & Finlayson, 2011), or wetlands restoration within nature-based approaches for disaster risk reduction (Monty *et al.*, 2016; Renaud *et al.*, 2016), are increasingly gaining traction in policy and decision-making. Considering their role in larger river basins and coastal zones, integrated land-use planning and management of wetlands can ensure that wetlands and their benefits are sustained in the long run (Maltby & Acreman, 2011; Ramsar, 2012). Enhanced understanding of multiple values of wetlands can greatly strengthen stakeholder engagement in mainstreaming wetland restoration agenda and actions (Kumar *et al.*, 2017; Russi *et al.*, 2013).

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
CROPLAND MANAGEMENT	Conservation agriculture	Av, Rd						
	Agroforestry	Av, Rd, Rv						
	Integrated crop, livestock and forestry systems	Av, Rd, Rv						
	Enhanced plant genetics	Rd						
	Agroecology	Av, Rd, Rv						
	Landscape approach	Av, Rd, Rv						
FOREST LAND MANAGEMENT	Agroforestry	Av, Rd, Rv						
	Protected areas	Av						
	Sustainable forest management	Av, Rd						
	Reduced impact logging	Rd						
	Landscape approach	Av, Rd, Rv						
	Restoration (active and passive)	Rv						
RANGELAND MANAGEMENT	Grazing management	Av, Rd, Rv						
	Pasture rotation	Av, Rd						
	Controlled burning	Av						
	Fencing	Av, Rd						
	Replanting	Rv						
	Intercropping	Av, Rd, Rv						
	Weed and pest control	Rd, Rv						
URBAN LAND MANAGEMENT	Green space management	Av, Rd						
	Street tree planting	Rv						
	Brownfield restoration	Rv						
	Removal of invasive species	Rv						
	Green infrastructure development	Av, Rd						
	Amelioration of contaminated soils and sealed soils	Rv						
	Sewage and wastewater treatment	Av, Rd						
	River channel/beach site restoration	Rv						

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
WETLAND MANAGEMENT	Protected areas	Av						
	Control of point pollution sources	Av, Rd						
	Control of non-point pollution sources	Av, Rd						
	Passive measures to allow natural recovery (e.g., control of human/ livestock pressures)	Rd, Rv						
	Active restoration measures (e.g., reshaping topography and hydrology, revegetation, invasion control)	Rd, Rv						
	Constructed wetlands	Rv						

EFFECTIVENESS RANKING OF RESPONSE OPTIONS

High effectiveness
Moderate to high effectiveness
Moderate effectiveness
Variable effectiveness (low to high)
Low to moderate effectiveness
Low effectiveness

6.3.2 Assessment of responses to selected direct drivers and impacts

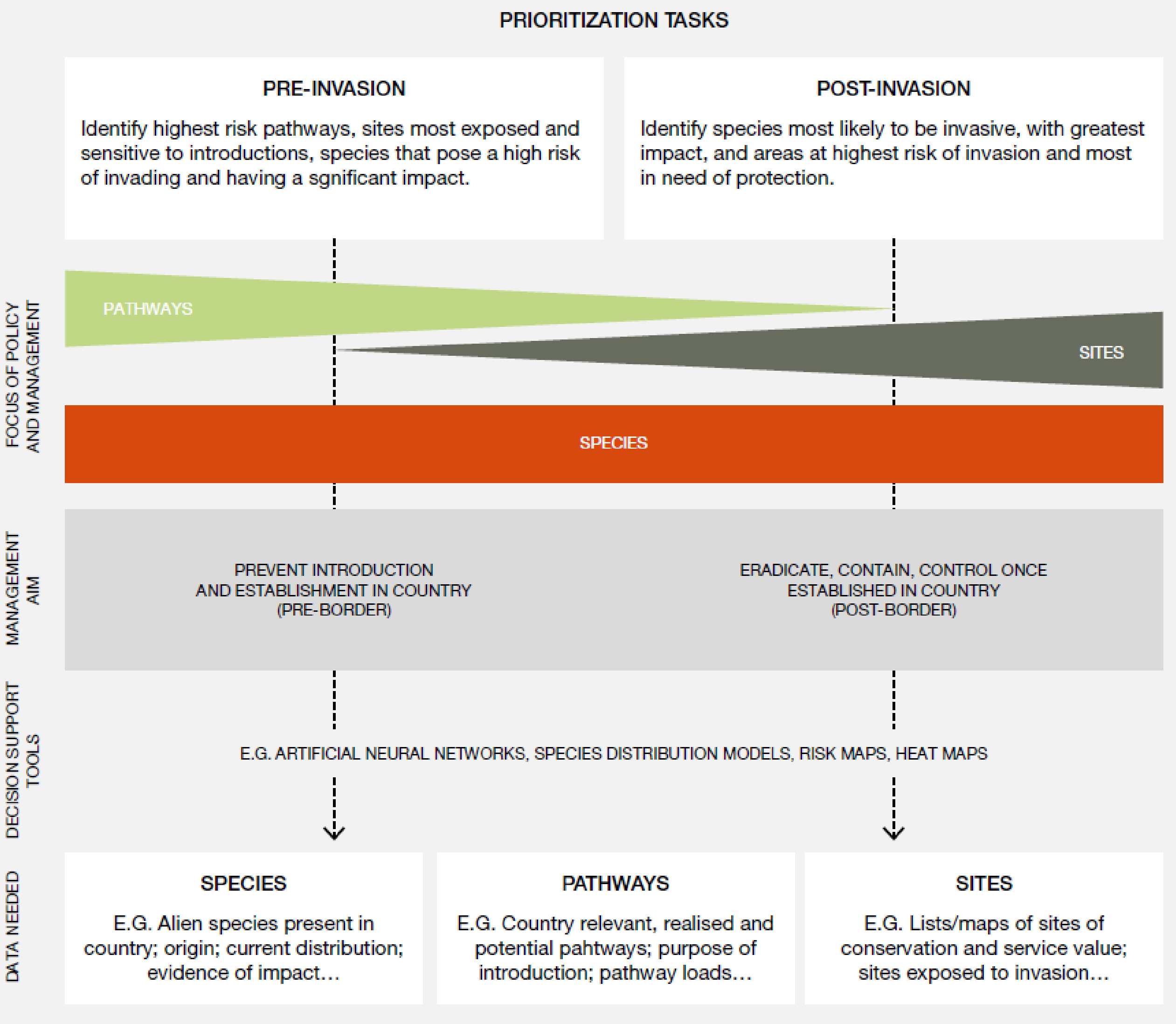
6.3.2.1 Responses to invasive species

Responses to invasion include institutional arrangements, policy and governance tools, as well as management strategies that interact in various ways based on spatial context. Managing invasive species is complex and challenging, primarily because of the dynamic nature of invasion processes, variable effects on different land-use systems (e.g., urban land versus agricultural land), and varying perceptions among stakeholders on ecosystem services or disservices generated by invasive species (Gaertner *et al.*, 2017). Typically, the costs of invasive alien species management strategies exceed available resources, particularly where socio-economic impacts of invasion disproportionately affect less advantaged social groups (Rai *et al.*, 2012; Shackleton *et al.*, 2011). Nevertheless, eradication or control of invasive species is often one of the aims of restoration (D'Antonio *et al.*, 2016).

Local communities in urban areas have detailed knowledge of the impacts of invasive species on biodiversity, their local environment and their values and perceptions of their local environment. To establish approaches to the management and restoration of invaded urban landscapes, engaging with local communities - along with experts in both restoration and invasion ecology, but led by local knowledge and those who continue to live in those landscape - provides innovative approaches and frameworks to manage and restore urban landscapes degraded by invasive species(Fisher, 2011; Fisher, 2016; Gaertner *et al.*, 2012). Local communities understand the importance of managing the landscape and the ecosystem as a whole. Invasive species management using a holistic ecosystem approach and driven by local communities, in differing urban landscapes - including coastal, woodlands, wetlands, rivers and estuaries - has proven to be highly successful in restoring functioning ecosystems. Long-term outcomes include restored urban environments resilient to changing climates with focus on the removal of all invasive species and their replacement with indigenous species (Fisher, 2011; Fisher, 2016; Gaertner *et al.*, 2012). Such an ecosystem approach to tackling invasive species has been adopted by the Sri Lankan Government at the national level and incorporated across policy, strategy, action planning, management and restoration (Fisher, 2015; Sri Lanka National Invasive Alien Species Committee, 2015).

The implementation of practical strategies usually occurs at local and national levels, and involves three successive steps - prevention, eradication and control (see Figure 6.8). In general, the most effective strategy is to prevent introductions of potentially invasive species before their establishment (Allendorf & Lundquist, 2003; Hulme, 2006; Leung *et al.*, 2002); due to the high cost of managing invasive species through eradication and control. *Preventive* measures focus on identifying and monitoring common biological invasion pathways (e.g., intentional and accidental introductions). Trade globalization and expanded transport networks have led to pathway risk assessments becoming the frontline in the prevention of invasions (Hulme, 2009). Pathway risk assessment relies heavily on spatial data, with risk maps that highlighting hotspots of invasion likelihood being a common product (Buckley, 2008). Linked to this is the identification of the invaders themselves and measuring their impacts (Blackburn *et al.*, 2014). This is where tools such as the Global Invasive Species Database (GISD) of the IUCN are useful. Many countries list prohibited species (e.g., categories of invasive alien species) and undertake awareness campaigns to educate the public about the threat to biodiversity posed by invasive alien species. The second component to prevention is interception (Boy & Witt, 2013), including the establishment of environmental biosecurity departments to carry out activities such as search and seizure procedures at first points of entry, as well as quarantine measures to block or restrict incursions. Examples of such bodies are the Australian Government's Department of Agriculture and Water Resources and the Animal and Plant Health Inspection Service in the USA. Such quarantine measures are, however, not necessarily feasible or effective in resource- and/or infrastructure-constrained settings.

Figure 6 8 Prioritization to support cost-effective allocation of resources is part of decision-making at nearly every stage of the invasion process, from preventing introduction of invasive alien species, to preventing their spread, to eradication or containment. Source: McGeoch *et al.* (2016).



Eradication is the next option in the practical response continuum and entails the systematic elimination of the invading species until it can be ascertained that no individuals, viable seeds or other propagules remain in an area (Boy & Witt, 2013). Eradication has been achieved, notably in island settings, with substantially more examples of successful eradication of vertebrate species than plant species (Genovesi, 2005; Glen *et al.*, 2013; Keitt *et al.*, 2011). Social acceptability of invasive animal eradication is controversial due to ethical issues (Cowan *et al.*, 2011; Rejmánek & Pitcairn, 2002; Simberloff, 2009). Early detection and decisive action are crucial for success (Pluess *et al.*, 2012; Rejmánek & Pitcairn, 2002; Simberloff, 2009) as early warning and rapid response systems enhance prompt detection of new incursions and correct taxonomic identification of invaders, assessing related risks and ensuring immediate reporting of relevant information to the competent authorities (EEA, 2011). In South Africa, for example, the National Department of Environmental Affairs has collaborated with the South African National Biodiversity Institute in the implementation of the Early

Detection and Rapid Response programme (Ntshotsho *et al.*, 2015a). Similarly, the European Commission has proposed a formalized early warning mechanism in the EU Regulation on invasive alien species which came into effect in January 2015.

Control of established invaders is the last line of defence, with the primary goal being the reduction of abundance and density in order to minimize adverse impacts. Successful control depends more on commitment and sustained diligence than on the efficacy of specific tools themselves, as well as the adoption of an ecosystem-wide strategy rather than a focus on individual invaders (Mack *et al.*, 2000). For invasive plant species, integrated weed management, which involves a combination of measures (Adkins & Shabbir, 2014), may be effective for long-term control in cases where invasive plants are able to survive individual measures. Generally, four types of control measures are in use for invasive plants: mechanical and/or manual, cultural, biological, and chemical; but "control by use" has also been considered as a control measure.

Mechanical and/or manual control of invasive plant species are often labour intensive, but in countries where communities manage land, and affordable labour is available, manual control is feasible (Rai *et al.*, 2012). Activities like hand-pulling and hoeing are site specific, can be effective in loose and moist soils, and to control small infestations (Sheley *et al.*, 1998). Mowing is most effective for annuals and some perennials (Benefield *et al.*, 1999), success depends on its timing and frequency (Benefield *et al.*, 1999; Rai *et al.*, 2012).

Cultural practices include controlled grazing, prescribed burning, and physical manipulation of habitat. There are several examples of such practices, for instance: controlled grazing to control *Parthenium hysterphorus* and *Centaurea solstitialis* (Adkins & Shabbir, 2014; DiTomaso, 2000); manipulating shading by overstorey to hinder the growth of *Lantana camara* (Duggin & Gentle, 1998); and prescribed burning to control invasion of annual broadleaf and grass species (DiTomaso *et al.*, 2006; Keeley, 2006). Indigenous practices for responding to invasive species provide important opportunities for effective responses and vary across the globe and the landscape (Ens *et al.*, 2016; Ens *et al.*, 2010). However, considering that invasive plants are likely to become established in disturbed habitats, cultural practices do pose a risk of promoting their proliferation (Fine, 2002; Moore, 2000).

Biological control (or biocontrol) is a means for controlling pests such as insects, mites, weeds and plant diseases using these organisms' natural enemies to reduce their abundance, rather than eradicate them (Charudattan & Dinoor, 2000; Ghosheh, 2005). Its effective implementation - based on extensive testing and validation for host-specificity to predict risk and minimize adverse environmental impacts (Delfosse, 2005; Messing & Wright, 2006) - is considered to be a cost-effective, long-term and self-sustaining control measure (Schlaepfer *et al.*, 2005).

Chemical control (use of biocides) is probably the most widely-adopted measure to control invasive plant and insect species. It is also the least desirable due to unintended adverse impacts on other non-target species in the surrounding environment and human health impacts (Giesy *et al.*, 2000; Khan & Law, 2005; Williams *et al.*, 2000). It is financially feasible under certain conditions such as high-value crops, at roadsides, public parks or on small areas (Adkins & Shabbir, 2014). Of concern is the growing global incidence of herbicide resistance in agricultural weeds (Heap, 2014; Preston, 2004). Herbicide resistance threatens to undermine control efforts and, consequently, underscores the need for integrated management (Kohli *et al.*, 2006; Shabbir *et al.*, 2013).

In terms of the effectiveness for controlling invasion of *Prosopis* spp., invasive species with global reach, mechanical and chemical measures are costlier than biological and "control by use" measures (van Wilgen *et* *al.*, 2012). But these latter control measures have been found less effective to reduce the invasions (FAO, 2006; Shackleton *et al.*, 2014). In Kenya and Ethiopia, *prosopis* has also been managed through "control by use" method (e.g., firewood, producing electricity for local use), but without any noticeable impacts on invasions (Zimmermann *et al.*, 2006). Biological control to manage prosopis has been found more effective in Australia with the use of four biological control agents: *Algarobius bottimeri, A. prosopis, Evippe* species, and *Prosopidopsylla flava* than in South Africa where three seed-feeding beetles: *A. prosopis, A. bottimeri and Neltumius arizonensis* were used (van Klinken, 2012; van Klinken *et al.*, 2003).

Box 6.5 The South African Working for Water programme

South Africa has a long history of problems with invasive alien plant species and management of biological invasions (Marais & Wannenburgh, 2008; Ntshotsho *et al.*, 2015a; Richardson & van Wilgen, 2004; van Wilgen *et al.*, 2002). These invasions pose a threat to human well-being by negatively impacting the provision of ecosystem services such as water and grazing (van Wilgen *et al.*, 2001). For example, it was estimated that the 1.5 million ha of land dominated by invasive alien plants were responsible for a total reduction of 1.44 million m^3/yr in mean annual runoff (van Wilgen *et al.*, 2012; Versfeld *et al.*, 1998). For a water-scarce country this is a substantial impact.

The Working for Water programme , arguably South Africa's largest nationwide conservation project, was initiated in 1995 with the primary aim to clear invasive plant species in order to increase water supply (Marais & Wannenburgh, 2008; van Wilgen *et al.*, 2002) while generating employment for marginalized people (Ntshotsho *et al.*, 2015a). Government funding to the programme increased from an initial f R25 million/yr (approx. $1.7 million/yr) in 1995, to R1.28 billion/yr (approx. $88 million/yr) in 2013/14 (WfW historical expenditure, http://sites.google.com/site/wfwplanning).

Figure 6 9 Images of the Upper Berg River Dam site in 2006 (left) and in 2015 (right). Source: ©2016 Cres/Spot Image & Image ©DigitalGlobe.

Figure 6.9 Images of the Upper Berg River Dam site in 2006 (left) and in 2015 (right)

The Working for Water programme has always adopted an integrated approach to invasive alien plant control, combining manual and chemical measures together with biocontrol. The programme is strongly supported by several pieces of legislation, primarily the Conservation of Agricultural Resources Act No. 43 of

1983 and the National Environmental Management: Biodiversity Act No. 10 of 2004, and their Regulations. Since its inception, the programme has maintained close links with the research community and has been influenced by scientific research (Ntshotsho *et al.*, 2015a). More than a million ha have been cleared since the beginning and employment opportunities are provided to approx. 20,000 individuals annually. Because of its positive societal and environmental impacts, the programme has grown and diversified into other programmes and, together, they now all fall under the Natural Resources Management umbrella programme.

At a local level, a recent assessment of one of the projects has demonstrated significant water gains (Ntshotsho *et al.*, 2015b). Modelling shows that clearing of the upper catchment of the Berg River Dam (Figure 6.9), which covers an area of approximately 12,000 ha, has resulted in estimated water gains of between 9.0 and 12.7 million m3/yr. This gain represents 7 to 10% of the capacity of the 126.4 million m3 dam. The dam is located within one of South Africa's 21 strategic water source areas (these are areas that occupy 8% of South Africa's land area and supply 50% of the country's surface water) (Nel *et al.*, 2013) and is the second most important source of water for Cape Town.

Improved water supply is not the only potential benefit of invasive alien plant eradication. Another project looking at the rangeland impacts of invasion has shown that *Acacia mearnsii* can reduce grazing capacity by 56% and 72% on lightly and densely invaded sites respectively, whereas clearing can reverse these losses by 66% within 5 years (Yapi, 2013). This translates to 2 to 8 hectares required to support one large livestock unit (ha/LSU) on uninvaded and densely invaded sites, respectively. Improved pasture condition has a direct positive impact on livestock condition and this can lead to improved human well-being at the household level (Ntshotsho *et al.*, 2015b). This has been demonstrated in yet another Working for Water project which looks beyond just the clearing of invasive alien plant species (*Acacia* spp.) and takes a land stewardship approach. Indigent communities in a rural part of South Africa were trained, guided and supported, through the programme to restore communal land. After two growing seasons post-clearing, there was discernible improvement in the physical condition of cattle. The cattle owners were then assisted to sell their stock to commercial butchers in the area in two auctions that generated revenue totalling just over R1.3 million (~$89 300) for the 63 households involved. The success of the Working for Water programme can be attributed to four interconnected factors at project level: commitment, passion, strategic planning and the consideration of context (Ntshotsho *et al.*, 2015b). In addition, political buy-in and long-term commitment of funds by government are equally important for the success of the programme at national level.

6.3.2.2 Responses to mineral extraction

The significant effects of mining surface lands include complete removal of ecosystems, hydrological disruption and degradation of soil resources during removal, storage and re-instatement (Harris *et al.*, 1996) (see also Chapter 3, Section 3.4.7.3). The use of heavy equipment and soil stockpiling during mining remains a major limitation to quickly re-establishing ecosystem structure and function (Harris *et al.*, 1989). Potential off-site impacts, particularly the generation of acid mine drainage, need to be minimized by on-site management.

Reclamation, rehabilitation and restoration of these sites to a variety of end-uses entails overcoming abiotic and biotic barriers or limitations to establishing functioning ecosystems (Hobbs & Harris, 2001). An overemphasis on idealized optimal conditions has often led to prescriptive targets for restoration, with the danger that this limits variability and spontaneity in the restored ecosystem (Brudvig *et al.*, 2017; Hiers *et al.*, 2016). Approaches include active intervention such as re-contouring, planting, soil amendment, inoculation, animal re-introduction and “spontaneous redevelopment” (Parrotta & Knowles, 2001; Prach *et al.*, 2013; Šebelíková *et al.*, 2016; Walker & Del Moral, 2009), with a variety of possible post-mining uses from natural systems to agriculture (Howieson *et al.*, 2017).

Sound waste management and rehabilitation plans are key elements in environmental restoration following the closure of mines (Adiansyah *et al.*, 2008). Topsoil management is of course critical, but only after a replacement of overburden and landscape reformation (Harris & Birch, 1989; Parrotta & Knowles, 2001). However, activities related to site rehabilitation yield no capital returns to mining operations and can have significant impacts on their operational costs and economic feasibility. Therefore, in less developed economies with weak mining governance, mitigation plans may be neglected.

On mined lands, active restoration is required to trigger natural processes of succession and to develop functioning soils (Gardner & Bell, 2007; Koch & Hobbs, 2007; Skirycz *et al.*, 2014; Tischew & Kirmer, 2007). The use of native species tolerant to heavy metals (metallophytes), and others capable of rapid soil development (e.g., nitrogen-fixing legumes), is a priority for restoration of contaminated mining sites (Ginocchio & Baker 2004; Whiting *et al.* 2010). However, this is not important when non-metalliferous materials have been extracted, especially coal, which covers a significant portion of the total area affected by surface mining, despite the fact that some sites suffer from an acidic pH, which is usually addressed by liming. A wide range of responses is available, ranging from “spontaneous regeneration”, through direct seeding and planting, to animal species reintroduction (see Stanturf *et al.* 2014 for a major review on this). Although significant research into physical management, organic and inorganic additions, plant reintroduction and fungal propagule inoculation has been carried out, the restoration of mined lands remains an intractable problem, with estimates of recovery varying from 10 to1000 years. Predicting time for ecosystem recovery is in practice difficult to determine, as different ecosystem characteristics recover at different rates, depending on degradation and disturbance type, site topology, on-site resources and off-site recruitment potential (Curran *et al.*, 2014; Jones & Schmitz, 2009; Spake *et al.*, 2015). Frouz *et al.* (2013) demonstrated that restoration to simple shortgrass prairies could be achieved faster than complex communities in tallgrass prairie and forest, on essentially the same post-mining substrates.

When only sub-soils and overburden materials are available for reclamation and/or restoration after mineral extraction, the addition of topsoil and composts can greatly aid establishment of vegetation (Spargo & Doley, 2016) and fauna (Cristescu *et al.*, 2013). Active intervention with fertilizers and soil amendments can enhance nutrient cycling and tree establishment (Howell *et al.*, 2016), and inoculating soil with appropriate mycobionts (especially mycorrhizal fungi) can aid tree establishment and survival (Asmelash *et al.*, 2016; Hoeksema *et al.*, 2010).

Soil ecology research has been used extensively to track the changes in sites subject to restoration programmes (Harris, 2003). Earthworm reintroduction has a positive effect on ecosystem service re-establishment (Boyer & Wratten, 2010), but only where they are natives. Mine site restoration in the Jarrah forest of Western Australia has been considered a largely successful case in terms of restoring vegetation (Grant & Koch, 2007) and fauna (Craig *et al.*, 2017). However, Banning *et al.* (2011) demonstrated that 26 years after mine restoration in these restored forests, microbial communities were not able to use the same range of carbon substrates than the reference sites. Nonetheless, progress towards a “reference” was more rapid than in less intensive programmes of restoration where fewer plant species and soil stockpiling were used; as opposed to the direct soil replacement and multiple tree species planting practices used in the Jarrah restoration programme.

Plant species additions, especially trees (Chodak *et al.*, 2015), can influence the eventual composition of the soil biota as well as chemical and carbon cycling (Harris, 2009; Józefowska *et al.*, 2017). Furthermore, by amending post-mining soils with "live" soils from a desired reference state site can enhance the rate at which ecosystem characteristics recover on drastically disturbed post-mined sites (van der Bij *et al.*, 2017) and these amendments can control the assembly of vegetation communities to reach the "desired" plant community configuration (Wubs *et al.*, 2016). Moving from stockpiling soils during mining operations, to "direct replacement" involving careful handling of soils during transfer, secures both better plant establishment and below-ground invertebrates, especially earthworms (Boyer *et al.*, 2011). Moreover, the re-use of stockpiled soil materials - combined with on-site waste mineral resources - can ensure a more complete and functionally-capable soil microbial community in post-mining sites (Kumaresan *et al.*, 2017).

"Spontaneous regeneration" is an approach which has been used extensively in Central and Eastern Europe, principally on post-coal opencast (strip) mines. Here, sites are re-contoured but not planted and can effectively regenerate. Šebelíková *et al.* (2016) demonstrated that while the species richness of such spontaneously regenerated sites were no different than that of sites reclaimed by active forest planting were after 20-35 years post-mining, they tended to be more diverse in terms of species of conservation interest (11 as opposed to 4 IUCN Red List species). Further, in many cases, woodland vegetation may become established on a successional trajectory through spontaneous regeneration after just 20 years on previously forested sites, but wetland sites are more variable in their progress (Prach *et al.*, 2013; Tropek *et al.*, 2010). Spontaneously regenerated sites provide better cover for establishing climax woody species than those sites which are deliberately planted (Frouz *et al.*, 2015). An essential caveat here is that without a readily available source of seeds and fungal spores that are able to reach these sites by natural means, such successional processes may take much longer.

6.3.2.3 Responses to soil quality changes

Healthy soils are a prerequisite for meeting global food, feed, fibre and energy needs (FAO, 2015). To meet those needs, while sustaining or improving soil health or soil quality, several soil and crop management response strategies have been developed - including various combinations of tillage, crop rotation, nutrient management, cover crops and other practices collectively referred to as "agronomic practices". Other response strategies include agroecology, organic farming, ecological intensification, conservation agriculture, integrated crop livestock and integrated crop livestock forestry systems. All of these strategies have different energy intensities, effects on biodiversity and levels of reliance on agrichemicals. These must be balanced through site-specific decisions which also recognize inherent constraints including climate change, acidification and salinization.

To monitor the effects of any response strategy, several soil health and/or soil quality indicators have been identified: biomass growth, development and productivity (Ponisio *et al.*, 2015); increased soil biodiversity and function (Birkhofer *et al.*, 2008; Roger-Estrade *et al.*, 2010); and species richness across a continuum from the field, to the farm, to the landscape level (Egan & Mortensen, 2012). Ideally, producers voluntarily select the most appropriate combination of practices to meet economic, environmental and social goals, but science-based regulations may be imperative in some situations (Chasek *et al.*, 2015; Karlen & Rice, 2015).

Soil health and quality have become essential for evaluating profitability and, as a guideline, for avoiding and reducing land degradation or restoring degraded lands due to their influence on: water entry, retention and release to plants; nutrient cycling; crop emergence, growth and rooting patterns; and ultimately yield. One of the most important soil health and quality changes, associated with any response strategy, is an increase soil organic carbon, because it directly influences a multitude of soil properties and processes. For example, applying animal or green manures can improve soil health and quality by increasing soil porosity, enhancing soil structure (i.e., binding of sand-, silt-, and clay-size particles), decreasing compaction, increasing aggregation and decreasing wind and water erosion.

Tools for assessing the effects of various response strategies on soil health and quality - at level of the field, farm, catchment, or larger areas - include the Soil Management Assessment Framework (Andrews *et al.*, 2004; Cherubin *et al.*, 2016) and the Comprehensive Assessment of Soil Health protocol (Moebius-Clune *et al.*, 2016). The EU Thematic Strategy for Soil Protection addresses soil health and quality and land degradation by striving to ensure that soils can provide seven critical functions: (i) food and other biomass production; (ii) storing, filtering and transformation of materials; (iii) habitat and gene pool of living organisms; (iv) physical and cultural environment for humankind; (v) source of raw materials; (vi) acting as a carbon pool; and (vii) archive of geological and archaeological heritage. This has been done by integrating soil protection into several European Community Policies (Toth, 2010), since efforts to establish a universal “Soils Framework” were unsuccessful.

Soil health and/or quality responses to selected degradation drivers

A combination of high-yielding, water-efficient plant varieties, the adoption of reduced- or no-till farming practices, improved pest and pathogen management, and optimizing planting schedules and crop rotations can improve soil health and quality, while reducing production costs and helping to mitigate atmospheric greenhouse gas (GHG) emissions. Burney *et al.* (2010) concluded that appropriate, site-specific combinations of those practices reduced GHG emissions by 161 GtC between 1961 and 2005, while Canadell & Raupach (2008) concluded that reforestation of 231 million ha could lead to an increase in carbon sink capacity from 0.16 to 1.1 Pg C yr^{-1}, between now and 2100. Afforestation of unused, marginal and abandoned land, as well as harvesting forests more frequently, could further promote carbon sequestration (Bird & Boysen, 2007; Harris *et al.*, 2006; Liu & Hiller, 2016; Valatin & Price, 2014). For China, Canadell & Raupach (2008) estimated that 24,000 km^2 of new forest was planted - offsetting an estimated 21% of China’s 2000 fossil fuel emissions. Better harvest management and prevention of forest fire or other disturbances can further increase forest carbon storage capacity (Liu *et al.*, 2016; Pilli *et al.*, 2016) and soil health.

Acidification

Cropland acidification (see Section 4.2.2.1) is caused by both natural and anthropogenic processes (Bhattacharya *et al.*, 2015; Günal *et al.*, 2015; Koch *et al.*, 2015) and has been calculated to reduce farm gate returns in Australia by $400 million per annum through lost production (Koch *et al.*, 2015). Response strategies include reducing atmospheric deposition and use of acidifying soil amendments such as anhydrous ammonia. Transitioning from long-term, high-rate nitrogen fertilizer applications and continuous cropping without organic inputs, in Africa, has been recommended to mitigate acidification (Tully *et al.*, 2015).

Acidification increases the mobility and leaching of exchangeable base cations (calcium, magnesium, potassium and sodium), decreases soil buffering capacity and increases concentrations of aluminium, magnesium and several heavy metals that are toxic to most plants. Therefore, the most direct approach to manage acidification is to apply lime (CaCO3) or other basic materials. This increases base saturation, decreases concentrations of aluminium, magnesium and other contaminants, improves the acid-base status of streams draining the area and stimulates recovery of biotic resources (Battles *et al.*, 2014; Johnson *et al.*, 2014). Unfortunately, liming is less effective for acidified subsoil, as time is required for lime to penetrate through topsoil before it can neutralize the acidity (Johnson *et al.*, 2014). Another response strategy is to change the amount and type of nitrogen fertilizer which Chen *et al.* (2008) reported influence soil acidity as follows: $(NH_4)_2SO_4$ > NH_4Cl > NH_4NO_3 > anhydrous NH_3 > urea. Acidification can also be reduced by decreasing atmospheric acid deposition. This has been occurring in Western Europe since 1980, because of increased air quality regulations (Virto *et al.*, 2015), but forest recovery remains limited because simply reducing acid input decreases aluminium and magnesium concentrations more rapidly than it increases base saturation.

Salinization

Salinization negatively affects soil health and quality by impairing productivity and several ecosystem functions. Globally, 23% of all irrigated land is classified as saline (FAO, 2014). Response strategies such as: (i) preventing excessive groundwater withdrawal and seawater intrusion, (ii) irrigating only where there is proper drainage, (iii) increasing aquifer recharge; and (iv) improving land and water management decisions, have been developed in response to an estimated $27.3 billion in lost crop production, alone (Qadir *et al.*, 2014).

In humid regions such as Canada, Northern Great Plains in the USA and Western Europe, a combination of geological conditions, climate patterns and cultural practices (tillage, crop selection, fallow lands and so on) have created saline seeps. The saline seeps form when soil water, not used by plants, moves below the root zone through salt-laden substrata to impermeable layers, and eventually flows to depressions where the water evaporates and leaves deposits enriched in sodium, calcium, magnesium, SO_4-S and NO_3-N which subsequently retard plant growth (Black *et al.*, 1981). This latter process is much more severe in arid and semi-arid regions (Anker *et al.* 2009). Response strategies include diverting surface drainage from recharge areas and intensifying cropping systems to fully utilize precipitation (MAFRI, 2008).

In Europe, most saline areas are located in areas with a Mediterranean climate (i.e., Spain, Greece and coastal parts of France and Portugal), often the result of improper irrigation (Virto *et al.* 2015). Suggested responses include: using high-quality (low electrical conductivity) irrigation water; applying sufficient irrigation water to leach soluble salts below the plant root zone; planting of salt tolerant cultivars; implementing phytoremediation with halophytes and subsequently harvesting them; adding calcium sulfate or strong acids; and increasing organic matter (FAO-ITPS, 2015). Another approach is to restrict the use of natural water resources to quantities that drain into terminal reservoirs as oceans, saline or dip aquifers (Schaible & Aillery, 2012). Growing salt-tolerant crops often have an added soil health and/or quality benefit, because they generally support the formation of stable soil aggregates that improve infiltration and resistance to wind erosion, while also decreasing surface crusting. Finally, there are several agro-hydro-salinity models such as SALTMOD, DRAINMOD-S or SAHYSMOD that can predict water distribution and salt balance, thus helping to reduce or even prevent salinization.

Soil management strategies to enhance soil health and/or quality and mitigate degradation

Tillage frequency and intensity, crop rotation, animal and/or green manure application, cover cropping, grazing intensity and agroforestry can improve soil health and/or quality (Wingeyer *et al.* 2015; Veum *et al.* 2015) and avoid, reduce or reverse land degradation by increasing biomass content and biodiversity. Tillage is especially important (Hammac *et al.*, 2016), because it affects surface cover and the size, composition and activity of the biological community below ground (Lehman *et al.*, 2015). Tillage also affects soil structure and stability, aeration, water balance and nutrient cycling - although response time when converting from high to low impact activities can take a decade. Soil health and quality changes - in response to fertilizer management, cover crops, animal or green manure applications, biochar and/or compost applications and site-specific management - also require time to be detectable. This temporal effect is therefore the basis for recommending soil health and quality monitoring to avoid, reduce or reverse land degradation. Finally, policy changes and especially national regulations, are currently very limited; relying instead on industry "best-practice" approaches to avoid further degradation and reductions in soil functional capacity (Chasek *et al.*, 2015).

Agroecological and ecological intensification approaches can enhance soil health and/or quality, reduce destruction or degradation of semi-natural ecosystems and homogenize landscape structure (Dumanski, 2015) (see also Chapter 5, Section 5.3.3.2). Ecological intensification involves actively managing farmland to increase natural processes that support production, including better biotic pest regulation, nutrient cycling and pollination (Bommarco *et al.*, 2013; Tittonell, 2014). Both ecological intensification and agroecology (see Section 6.3.1.1) emphasizes making smart use of ecosystem functions and services at field and landscape scales, to enhance agricultural productivity, reduce reliance on agrochemicals and thus avoid further land-use conversion. As a practice for preventing or mitigating cropland degradation and maintaining or improving soil health and quality, planting a green cover between crop rows has been suggested because it reduces soil erosion. However, the cover crop can use a considerable portion of the plant-available water. Hence good, data-driven and science-based management practices are essential for a win-win outcome in these practices.

Many have advocated "organic" farming practices to enhance carbon sequestration (Gattinger *et al.*, 2012), reduce cropland soil degradation and avoid unintended consequences such as impaired water quality and/or quantity associated with intensive agricultural practices (Cambardella *et al.*, 2015). Typical organic farming practices include the application of composted animal manure, use of forage legumes and green manures and extended crop rotations. National regulation and/or policy changes may help advance organic farming, but costs of production, tillage for weed control and possible yield reductions, are still often cited as being significant.

Conservation agriculture (see Section 6.3.1.1) encompasses many different practices that, in combination, can avoid, reduce and even reverse land degradation (Dumanski, 2015; Farooq & Siddique, 2015; Lal, 2015a, 2015b). Implementing conservation agriculture practices can improve soil health and quality by intensifying production, enhancing environmental benefits and protecting against water pollution. Conservation agriculture can also help increase soil organic carbon content, conserve soil structure and ensure or enhance soil microbial biomass.

By preventing excessive or uncontrolled livestock grazing, ensuring that crop residue removal is not excessive, decreasing wind and water erosion and avoiding depletion of soil organic matter, integrated crop, livestock and forestry practices provides a multitude of benefits for soil health and quality. Optimal response strategies will differ between arid or semi-arid ecosystems and humid areas, and success very much depends on the biome type. In some areas, national grazing regulations can influence whether land is managed sustainably or not (Nielsen & Adriansen, 2005). The practices can be optimized by implementing evaluation schemes focused on soil organic matter, because of the influence it has on several soil health and/or quality properties and processes. However, even though soil organic matter content is effective for assessing and monitoring effects of the land-use policies and optimizing crop, livestock and forestry integration (Toth, 2010), it is a poor surrogate for characterizing soil biodiversity.

In summary, several different management strategies can be used to avoid or mitigate soil health and/or quality changes and many can be implemented in developing countries. Regardless of the specific practice, the most important strategy may be to adopt policies that ensure efficient, economical and sustainable methods are being used to enhance soil health and quality and avoid further land degradation.

Use of indigenous and local knowledge (ILK) with scientific inputs can be an effective response to reduce or reverse soil degradation (see Box 6.6 for an example of highly effective ILK use to enhance soil health).

Box 6.6 Use of farmers' knowledge to enhance soil health in India

An extensive indigenous and local knowledge (ILK) base for natural resource conservation and management exists in most countries. In India, where traditional soil and water conservation practices are implemented under a variety of agroecological conditions, many agronomic practices including terracing, applying soil amendments, harvesting water, controlling seepage, recharging groundwater, optimizing tillage and using different land configurations, are influenced by ILK (Mishra, 2002).

One example focused on soil health is the use of mixed and diversified cropping systems. In rainfed areas, farmers use traditional practices to grow various annual crops (including millet) that exploit different growth habits and rooting patterns. Those differences enable the crops to use nutrients and soil water from different soil layers, thus increasing resource-use efficiencies. In turn, this results in more rapid canopy closure which reduces weed growth and competition with the annual crops, as well as the erosive impact of intensive (monsoon) rainfall when it does occur. Furthermore, the sequence of crops is selected in a manner that enables the above-ground crops to be harvested before the underground crops and to support grazing of crop residues by animals. The combination of residual root biomass, crop residue, animal excreta and farmyard manure helps sustain the soil organic matter content, which in turn improves soil health, crop nutritional status and economic returns to the farmers.

6.3.2.4 Responses to water quality changes

Land-based pollution and degradation of freshwater and coastal ecosystems have implications for both the health of aquatic, coastal and marine ecosystems (see Chapter 4, Sections 4.2.4 and 4.2.5), as well as food and water security, human health and exposure to flood risk (see Chapter 5, Sections 5.3.2, 5.8.1 and 5.8.2). Local responses to water resources pressures - exacerbated by climate change impacts in many regions - focus primarily on improved crop and soil management (see Sections 6.2.1.1 and 6.3.2.4) as well as ILK related to water conservation and management. They also include a variety of other water management approaches such as: construction of large or small dams, reservoirs and irrigation systems; wastewater treatment; river and stream rehabilitation; and development of advanced water management technologies (CGIAR, 2016).

Integrated land and water management is an effective response to ensure catchment-scale hydrological balance and to minimize the occurrence of extreme hydrological events (floods and drought) and their impacts on people. Other responses applied to agricultural land management (see Sections 6.3.1.1 and 6.3.2.4) include: improvements in rainfed agricultural productivity (through, for example, increased use of drought-resistant crop varieties); managing soil health and fertility; managing soil moisture in rainfed areas; increasing efficiency of irrigation systems and improving on-farm water productivity; and managing environmental risks associated with agricultural intensification (FAO, 2011). An example of a management programme that has had some success in improving water quality and ecosystem health is the Chesapeake Bay Program: a regional partnership established in 1983 that directs and conducts the restoration of the Chesapeake Bay in the mid-Atlantic region of the USA. This Program, and the 2014 Chesapeake Bay Watershed Agreement, coordinates efforts of various state, federal, academic and local watershed organizations. The aim is to build and adopt policies which support the goal of reducing the amount of pollutants and nutrients from upstream land-based sources - particularly nitrogen and phosphorus from agricultural runoff that have, since the 1950s, resulted in extensive eutrophication and hypoxia of the region's rivers, estuaries and marine ecosystems (Goesch, 2001; Hagy *et al.*, 2004; Kemp *et al.*, 2005).

Responses to hydrological regime changes include the use of soil and water conservation techniques, judicious land management practices and the provision of incentives to landholders and communities (Brunette & Germain, 2003). The use of mobile-based networks and apps allows for rapid, reliable decisions on monitoring, acquiring and processing real-time data on water level, rainfall, runoff, water quality and leakage detection. Such systems help farmers to optimize irrigation and obtain (cloud-based) information on soil data - allowing them to determine the amount of water necessary to produce the maximum yield in a given irrigation zone. Responding to a drought of historic severity, California started a pilot programme to install smart water meters that detect leaks and optimize water use at the household level. At the same time, they are using sensors for smart irrigation control to reduce water consumption by the State's large agricultural producers (IWA 2015).

The coordination of environmental, economic, trade and development policies can promote practices that improve natural resource-use efficiency, which is essential for countries with relative water shortages. New solutions for appropriate water balance have been devised, such as water trading, cloud stimulation and climate-smart technologies.

Water quality technologies such as desalination and wastewater treatment are energy intensive and may be expensive and/or produce effluents that must be disposed of. One prominent challenge in water reuse (particularly potable reuse) lies in community acceptance, because many people are inherently averse to drinking or using reclaimed water (Brown & Davies, 2007). Uses of non-potable reclaimed water that are more widely acceptable include agricultural irrigation, industrial processes, street washing, toilet flushing and landscaping. Greywater can also be used for irrigation but, like wastewater, it must undergo some treatment to remove oil, surfactants and other organic contaminants before it is applied to crops (Travis *et al.*, 2010). Reclaimed water also has potential uses in urban and suburban landscape maintenance and other non-agricultural spaces, thereby reducing the use of potable water for non-drinking purposes. Industrial processes that utilize reclaimed water include evaporative cooling, boiler feed, washing and mixing (Levidow *et al.*, 2016; Thoren, Atwater, & Berube, 2012).

Wastewater treatment using constructed wetlands (see Section 6.3.1.5) has been used effectively in both developed and developing countries (IWA, 2015; SIWI 2010). Making these systems more automated, low maintenance and user-friendly may help promote widespread implementation of small-scale systems, that together can save vast amounts of potable water (IWA 2015).

Effective water management solutions range in their cost, accessibility and energy efficiency. Most demand-based management strategies tend to be relatively low cost, and by reducing water consumption, they decrease pressure on water resources. Rainwater and runoff harvesting techniques are often energy neutral and include low-cost practices that can be used almost anywhere (Mekdaschi-Studer & Liniger, 2013).

Technologies for addressing water challenges are becoming more advanced and increasingly energy efficient (IWA 2016; UN Water 2015), but unfortunately many of the countries with the greatest need for more reliable water supplies lack the economic means to implement them. Some promising examples of alternative water management technologies being used in developing countries (IWA 2016) include:

- Small-scale rural greywater reuse systems in rural Madhya Pradesh in India, which was so effective in reducing water demand and improving sanitation that similar systems were later implemented to serve over 300 schools and 1,500 households, thus avoiding contamination of soils and water, and negative impacts on human health (Godfrey *et al.*, 2010);
- In the village of Cukhe, on the outskirts of Hanoi in Vietnam, rainwater harvesting systems (costing less than $400) that consisted of screens, settling tanks with calm inlets, UV filtration and first flush systems were installed. They eliminated the need for expensive bottled water to supply potable water and avoided groundwater contamination by arsenic and sewage runoff. Furthermore, by using previously less-trusted groundwater to meet outdoor and non-potable needs, the village was able to diversify its water supply and conserve rainwater (Nguyen *et al.*, 2013).

A comprehensive understanding of the water-energy nexus is therefore needed in decision-making about technological options and considerations for clean, renewable energy sources should be incorporated into projects as much as possible (IWA 2016). Because no solitary solution is globally applicable, water managers and relevant stakeholders must together find the solutions most appropriate to the social, economic, political, institutional and environmental conditions of a given area (IWA 2015). A nearly globally-standardized set of best available technologies or techniques aimed at optimizing systems of integrative pollution prevention and control have been developed, primarily for the industrial sector (Entec, 2009; Geldermann & Rentz, 2004; Karavanas *et al.*, 2009). Similarly, best practice guidelines for water harvesting, based on experiences from throughout the world, are also available (Mekdaschi-Studer & Liniger, 2013).

Box 6.7 Improving food security in Ethiopia through agrometeorological monitoring

Ethiopia, where one in three people currently live below the poverty line, has one of the world's largest populations dependent on the vagaries of annual rainfall (ECSA & WFP, 2014). When droughts occur, very large numbers of people can be adversely affected by crop production shortfalls. At times, as many as 7.6 million people may require emergency support. Since Ethiopia has many inaccessible regions, an objective, country-wide, geographic assessment of conditions called the Productive Safety Net Program has been developed (FAIS, 2012; GOE, 2015).

The Program uses a numerical model - the water resource satisfaction index - which can be related to crop yield using a linear yield-reduction function, specific to each crop. In this way, crop yield is modelled at the start and end, and for the entire season (Senay & Verdin, 2003). In addition to water, other factors that affect food security - such as poor roads and the cost of grain transport (Rancourt *et al.*, 2014) - are taken into account.

Since the water resource satisfaction index is a numerical index, it can be used for comparisons within and over multiple years; for example, the number of seasons when the crops failed completely between 1982 and 2011. Figure 6.10 shows that while mountainous highland areas experienced increases in rainfall during this period, the region in the rain shadow, in Tigray, became drier and less productive - with the area experiencing failed seasons in most years increasing to the east. The South-central and Southern Ethiopian regions, where most of the population is located, has experienced declines in rainfall over a thirty-year period (Funk *et al.*, 2005). This is due to both the changes in rainfall, as well as higher temperatures driving increased evapotranspiration. An advantage of the country-wide method is that it can show where rainfall anomalies are affecting crop yield, considering multiple drought-sensitive crops. Detecting and responding to changing rainfall, and consequent agricultural productivity, are key ways for Ethiopia to anticipate food security issues and respond early. In many countries at risk of food insecurity, similar schemes are used (e.g., Brown 2008, the Famine Early Warning System, FEWS; GEOGLAM Crop Monitor for Early Warning, https://cropmonitor.org).

Figure 6 10 The number of seasons that have a water requirement satisfaction index value of 50% or less for small grains between A 1982-1991, B 1992-2001 and C 2002-2011 in Ethiopia. The higher the number, the more failed seasons; D population density per square kilometer in 2020. Source: GPWv3 CIESIN (2005); Brown *et al.* (2017).

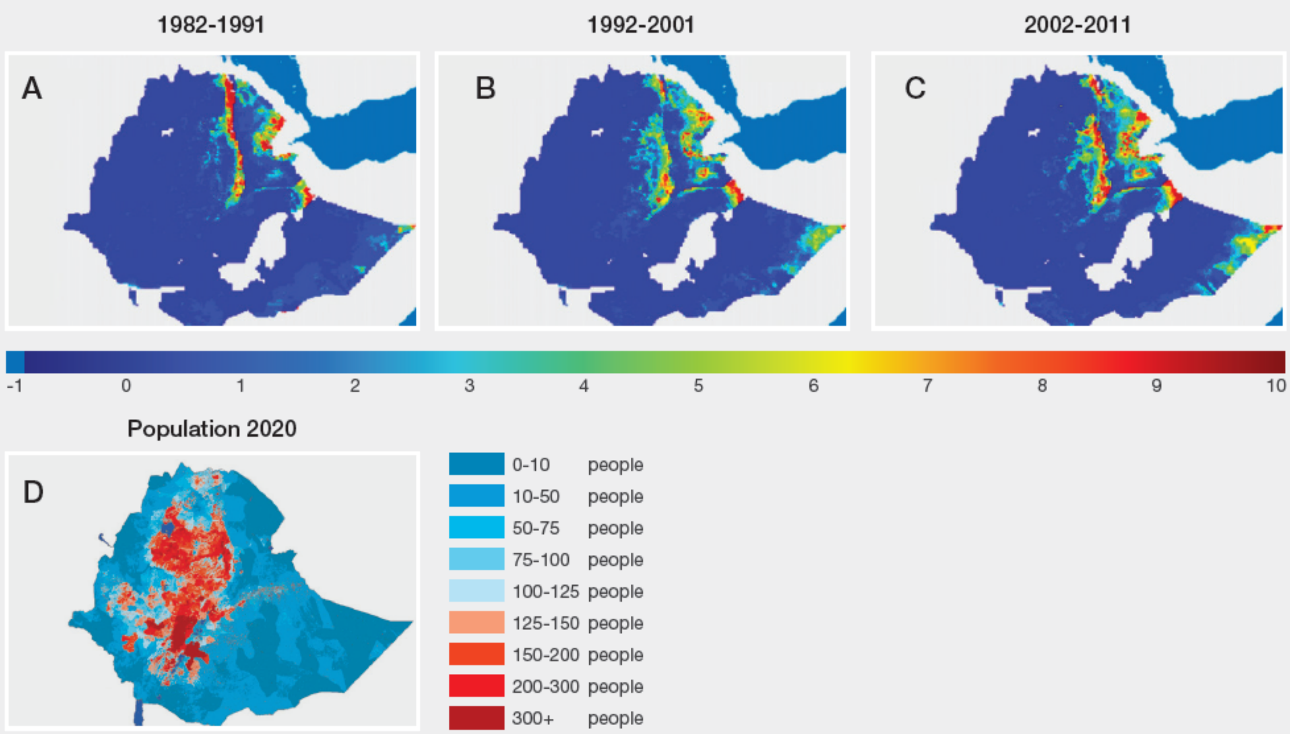


Table 6 6 Summary of direct biophysical and technical responses, their nature and relative effectiveness in avoiding, reducing or reversing land degradation caused by invasive species, mineral extraction, soil quality change and water quality change.

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social acceptability	Environmental desirability	Cultural acceptability	Technical feasibility	Political acceptability
INVASIVE SPECIES MANAGEMENT	Identification and monitoring of invasion pathways	Av						
	Quarantine measures	Av						
	Mechanical control	Rd						
	Cultural control	Rd						
	Biological control	Rd						
	Chemical control	Rd						
MINE SITE MANAGEMENT	On-site management of mining wastes (soils and water)	Rd, Rv						
	Reclamation of mine site topography	Rv						
	Conservation and early replacement of topsoil	Av, Rd						
	Passive restoration measures to recreate functioning grassland, forest and wetland ecosystems	Rd, Rv						
	Active measures to restore natural hydrological dynamics, biodiversity and soil profiles	Rd, Rv						
SOIL QUALITY IMPROVEMENT	Managed or rotation grazing	Rd, Rv						
	Agroecological management	Av, Rd, Rv						
	Conservation Agriculture	Av, Rd						
	Organic farming	Av, Rd, Rv						
	Reduced tillage frequency and/or intensity	Av, Rd						
	Increased crop diversity and perennials	Av, Rd, Rv						
	Using cover crops	Av, Rd, Rv						
	Crop rotation	Av, Rd, Rv						
	Fertilizer management	Rd						
	Adding animal or green manure	Rd, Rv						
	Adding compost or biochar	Rd, Rv						
	Provide adequate drainage	Rd, Rv						
	Erosion control	Av, Rd, Rv						
	Phytoremediation	Rd, Rv						
	Repositioning eroded soil	Rd						

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
WATER QUALITY IMPROVEMENT	Rainwater harvesting	Rd, Rv	High	High	High	High	High	Moderate
	Wastewater treatment	Av, Rd	High	High	High	High	High	High
	Constructed wetlands	Rv	High	High	High	High	High	High
	Desalination	Rd	High	High	High	Moderate	High	Moderate
	Integrated land and water man-agement	Av, Rd, Av	High	High	High	Moderate	Moderate to high	Moderate to high
	Soil and water conservation practices	Av, Rd, Rv	High	High	High	Moderate	Moderate to high	Moderate to high
	Point source pollution control	Av, Rd	Moderate	High	Moderate	High	Low to moderate	Moderate to high
	Non-point source pollution control	Av, Rd	Moderate	Low	Low	High	Low to moderate	Low to moderate

EFFECTIVENESS RANKING OF RESPONSE OPTIONS

- High effectiveness
- Moderate to high effectiveness
- Moderate effectiveness
- Variable effectiveness (low to high)
- Low to moderate effectiveness
- Low effectiveness

6.4 Enabling and instrumental responses to land degradation and restoration

Enabling and instrumental responses are intended to address the direct and indirect causes of land degradation, thus avoiding further degradation and ultimately restoring or rehabilitating the land. The responses are broadly grouped into policy instruments, institutions, governance and anthropogenic assets (infrastructure, human resources, capacity, technology and indigenous or local knowledge-based practices) (MA, 2005a). This section complements Section 6.3 by briefly assessing potential responses to key indirect drivers and then assessing effectiveness of policy, governance and institutional responses to land degradation.

6.4.1 Responses to indirect drivers: globalization, demographic change and migration

Indirect drivers including pollution, migration, globalization, consumption patterns, energy demand, technology and culture can degrade land in many ways (see Chapter 3, Sections 3.6.3 and 3.6.4). The optimum response to those drivers will depend on which driver is most influential, how it interacts with other indirect drivers, the current institutional, policy and other governance factors (see Chapter 3, Section 3.6.2). As comprehensive evaluation of all indirect drivers is impractical (see Chapter 3, Section 3.6 for details), this section focuses on three: globalization, demographic change and migration. Although increased globalization and international trade can reduce economic growth barriers, they also bring environmental challenges, including land degradation. For example, increased demand for food and fuel in Asia and Europe led to rapid expansion of soybean production in the Amazon, Chaco and Cerrado biomes - pointing to how the shortening of supply chains, facilitated by information and transport technology, affects land-use decisions in distant places (Garrett *et al.*, 2013; Liu *et al.*, 2013). Responses to control the unintended consequences of globalization, international trade and consumption preferences in developed and developing countries involve raising public awareness, multi-sectoral and coordinated governance arrangements between private and public sectors, and the use of innovative policy instruments (Lambin *et al.*, 2014) (also see Section 6.4.2 and Chapter 8, Section 8.3).

Responses to land degradation caused by globalization and international trade of commodities include linking trade and environmental protection as a continuum from local to global levels (Lambin & Meyfroidt, 2011), with the use of policy instruments (e.g., tariffs). In conjunction, voluntary product certification schemes have been used to regulate land use, trade and consumption patterns, and have been environmentally effective for coffee (Lambin & Meyfroidt, 2011). The introduction of eco-certification of forest products in the early 1990s did not halt the decline of biodiversity in the tropics, as was intended, but it raised awareness and increased dissemination of knowledge on comprehensive sustainable forest management by embracing economic, environmental and social issues at a global level (Rametsteiner & Simula, 2003). Maintaining social and environmental standards for production, supply chain and consumption practices is imperative to minimize the ecological footprint of globalization and international trade.

Demographic change not only affects local land use and cover, but is also associated with land degradation and biodiversity loss at multiple spatial and temporal scales. Population density and other demographic factors (e.g., population structure, growth rate, migration dynamics and gender inequality) have complex relations with land degradation per se, and their impacts differ greatly (Waggoner &

Ausubel, 2002), often due to differences in affluence and behaviour. Responses to land degradation and restoration actions are more effective when aligned with high-level population policies that take into consideration specific population and land degradation interactions. Policy responses to address human-land interactions versus population change are not the same. The former may focus on reducing negative impacts of agricultural activities on biodiversity and land condition through sustainable intensification or other means (see Sections 6.3.1.1, 6.3.1.2 and 6.3.2.3), whereas the latter focuses on resettlement, fertility rate and rural-urban migration.

Forest ecosystem recovery through natural regeneration following rural-urban migration is well documented for many parts of Latin America (especially Patagonia, Northwest Argentina, Ecuador, Mexico, Honduras and the Dominican Republic) and for non-forested ecosystems (e.g., montane deserts and Andean tundra ecosystems of Bolivia, Argentina and Peru) (Aide & Grau, 2004). In Puerto Rico, forests have recovered from a low of less than 10% of the island's land area in the late 1940s to more than 40% in the 2000s, as a result of rural-urban migration (Grau *et al.*, 2003). In Misiones, Argentina, rural emigration "reduced" deforestation by 24% compared to a "no-migration" scenario. If future emigration rates increase, deforestation will be reduced by 26% in 2030 compared to the current trend (Izquierdo *et al.*, 2011). Within Latin America and the Caribbean, 362,430 km^2 of woody vegetation recovered between 2001 and 2010 because of outward migration and socio-economic changes (Aide *et al.*, 2013).

Migration-related land sparing and forest transitions have occurred historically in developed countries, but are now happening in many developing countries (Lambin & Meyfroidt, 2011). In China, ecological migration is a driver for resettlement policies and actions to promote ecosystem recovery (Wang, Song, & Hu, 2010). For example, the Chinese government has relocated millions of people from ecologically vulnerable areas, such as mountain areas of Guizhou and Shannxi province, to other rural or urban areas to facilitate land recovery (Chen *et al.*, 2014). From 2000 to 2012, about seven million farmers in Western China, alone, were relocated to areas within or outside their provinces (Tsunekawa *et al.*, 2014). However, this kind of relocation (for ecosystem recovery) requires careful assessment of its effectiveness and long-term impact. A study in a resettled area of north-western China found that water scarcity and its associated risks have not been alleviated due to land degradation (Fan *et al.*, 2015).

Voluntary rural-urban migration is a common adaptation response to land degradation. Household migration and depopulation of the countryside can lead to ecological restoration (Gao *et al.*, 2014; Wang *et al.*, 2010). In recent years, with the exception of North America, several parts of the world have experienced depopulation in mountain regions due to climate change and socio-economic conditions (Black *et al.*, 2011; Piguet, 2012). This trend has contributed to land restoration through natural processes in mountain regions.

6.4.2 Institutional, policy and governance responses

Institutional, policy and governance responses are designed to create, enable and implement actions on the ground to avoid, halt and reduce land degradation or reverse/restore degraded lands. The effectiveness of these responses is primarily associated with their design and implementation, including the type of policy instrument used and access to anthropogenic assets (e.g., research and technology development, institutional reform and capacity-building). This section focuses on types and effectiveness of policy instruments for guiding long-term decisions to avoid, halt and reduce land degradation and to restore degraded land at national and local levels (also see Section 8.3). Figure 6.11 illustrates several land degradation and restoration challenges and the associated policy goals, instruments, and support tools and methods to address them.

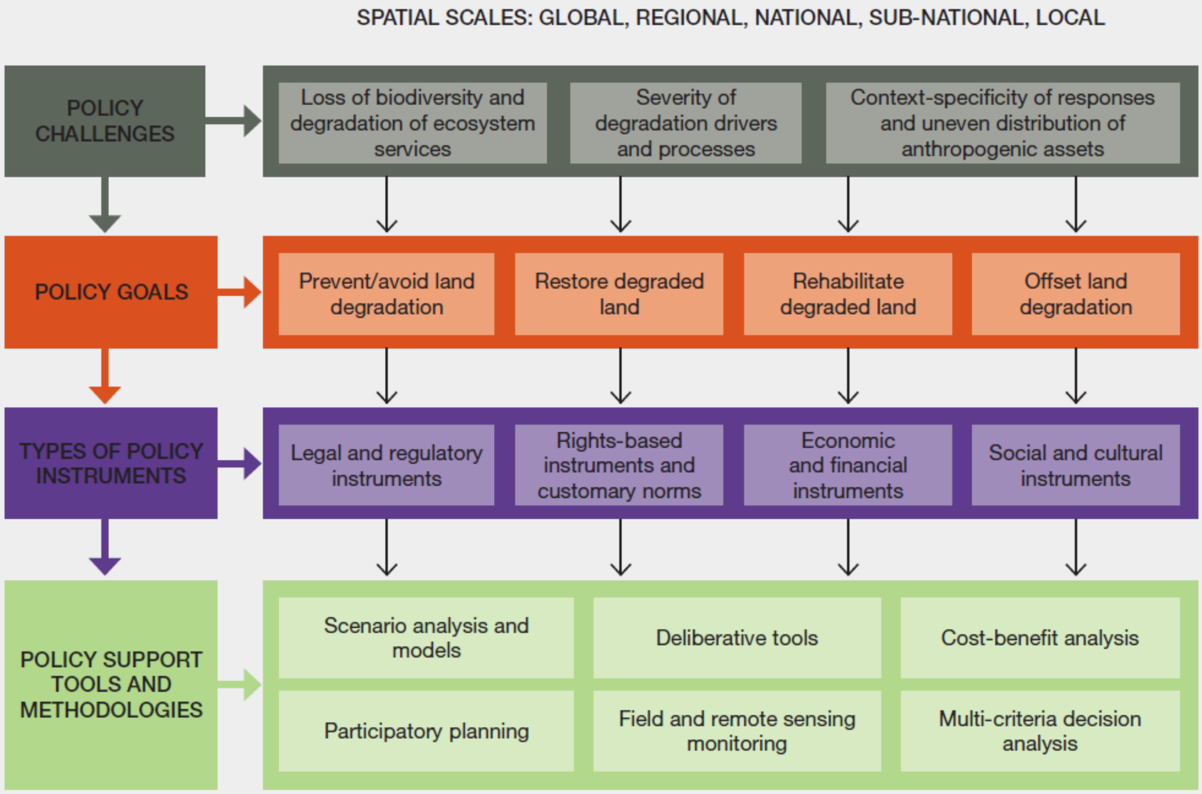


Figure 6 11 Land degradation- and restoration-related policy challenges, goals, instruments, tools and methodologies.

The appropriate policy instrument may depend on the spatial scale (i.e., local, regional, national or global) needed to achieve policy goals - although the same policy instrument can be applied at two different spatial levels for related policy goals. In Figure 6.11, the horizontal arrows expand the policy domain while the vertical arrows show relationships among policy support tools, methodologies and challenges. The vertical arrows thus represent many combinations that can contribute to one or more policy goals and challenges. Land-management policies and instruments are effective only when land managers are supported by those policies and have the means, commitments and control to restore, maintain or improve the quality of land (ELD, 2015). Furthermore, the appropriate policy instrument choice to promote sustainable land-management practices or landscape restoration depends on its environmental effectiveness, costs of implementation, monitoring, enforcement, distributional effects and conformity with other policies and political preferences (Low, 2013). This means that to be effective, policy instruments must be: economically and technically feasible; environmentally beneficial and desirable; and culturally, socially and politically acceptable (see Section 6.2.2).

6.4.2.1 Legal and regulatory instruments

Legal and regulatory instruments are used to encourage land managers to operate within the prescriptions of a given policy. The effectiveness of such instruments depends on specific policy settings (Alterman, 1997; Kairis *et al.*, 2014). For states that control land management, the first and most commonly-used legal and regulatory instrument - to avoid land degradation and to reduce or reverse adverse consequences of improper land use - is planning at national or regional (master plan) and local (zoning map) levels. The second set of instruments involves legal frameworks designed for industrial and agricultural activities based on national or regional standards.

Planning is a legal response according to the principle of subsidiarity and division of powers between public authorities (Dumanski, 2015; ESPON, 2012). This kind of legal response allows authorities to manage land use. Land planning and associated zoning enable the division of land for specific uses (e.g., natural, agricultural, or urban areas, limited housing density and/or urban growth areas, cluster zoning and/or obligation to build in continuity areas), and to establish legal or contractual conservation easements (Dissart, 2006; Hassan & Lee, 2015; Yucer *et al.*, 2016). In support of local planning, national and local authorities may also use other legislative and regulatory instruments, such as land-use or building permits, purchase of development rights, eminent domain (used in the most sensitive areas, e.g., coastal zones), or freezing the use of certain lands through land reserve funds. Territory control also allows the use of tax incentives, such as tax relief for non-waterproof or non-constructible lands, to maintain or relocate farming operations (Dissart, 2006).

International law can influence national policies related to soil protection and even compel states to adopt new legislation (Hannam & Boer, 2001; Leibfried *et al.*, 2015; Montanarella & Vargas, 2012). Local planning is thus subject to national and international law which can provide indirect protection for soils, safeguarding of wetlands and groundwater (e.g., Directive 2000/60/EC on Groundwater Protection of the Ramsar Convention; Dooley *et al.*, 2015; Kløve *et al.*, 2011), management of coastal land (eminent domain and/or easement), establishing targets for land degradation neutrality (Dooley *et al.*, 2015), management of public domain forests and conservation of biodiversity (e.g., UNCCD, CBD, Directive 2009/147/EC on the conservation of wild birds). International law can also improve national policies by converging policies within the same geographical territory across state boundaries (e.g., Cuypers & Randier, 2009; Directive 92/43/EC on the conservation of natural habitats and of wild fauna and flora**,** Alpine Convention).

Planning is also an instrument to avoid and reduce land degradation, commonly used in response to urban sprawl (Artmann, 2014), land encroachment (Gennaio *et al.*, 2009; McWilliam *et al.*, 2015), impermeability (Prokop *et al.*, 2011) and drought (Wilhite *et al.*, 2014). Indirectly, it works against the loss of organic matter and biodiversity, as well as flooding and soil compaction (DeFries *et al.*, 2010; Turbé *et al.*, 2010; Vu *et al.*, 2014).

The second most common set of legal and regulatory instruments used to avoid land degradation is based on legal frameworks designed to regulate economic activities known to be associated with land degradation (i.e., a similar approach to industrial regulation). Negative impacts on land and ecosystems from economic activities can also be mitigated through environmental impact assessments (Prieur, 2011) and provision of offsets for residual impacts of development activities. In addition to applying environmental standards on development activities, incentives such as eco-conditionality on financial assistance can also be adopted to minimize land degradation. Examples of such incentives include providing shares in favour of reducing the use of pesticides, enhancing crop diversification, converting to organic farming and organizing short distribution channels (Arnalds & Barkarson, 2003; Billet, 2008; Bodiguel, 2014; Pretty *et al.*, 2001; Singh, 2015, 2016). Incentives can also be used to reduce soil pollution or contamination, compaction or impermeability, and loss of organic matter or biodiversity. For example, EU farm policy promotes environmental protection with "agri-environment measures" that provide payments to farmers who participate in such measures (on a voluntary basis) to pursue a number of management practices. Such practices include: the management of low-intensity pasture systems; integrated farm management and organic agriculture; preservation of landscape and historical features such as hedgerows, ditches and woods; and conservation of high-value habitats and their associated biodiversity (Baylis *et al.*, 2008; Bodiguel, 2014; Bredemeier *et al.*, 2015; Bureau & Thoyer, 2014; Dal Ferro *et al.*, 2016; Huttunen & Peltomaa, 2016; Russi *et al.*, 2016).

The mechanism by which legal and regulatory instruments typically operate is based on the "polluter pays" principle, with an obligation to restore the site - failing of which requires an equivalent compensation to be paid for the damages suffered. To rehabilitate or compensate the residual effect of development (e.g., after a strategic environmental assessment or an environmental impact assessment) or contaminated sites, the project proponent is responsible for remediating impacted sites or contaminated soils when project activities end (Sirina *et al.*, 2013). Public authorities often assist in restoring sites (Lecomte, 2008; Steichen, 2010; Veenman, 2014). In the case of brownfields redevelopment/orphan site, restoration can be the direct responsibility of public authorities (Reinikainen *et al.*, 2016; Van Calster, 2005; Vanheusden, 2007).

For states that either do not control their land or have land management authority, contractual approaches are often used. These are characterized by the implementation of national plans (e.g., national plan against desertification or forest protection). Such plans establish a link between public authorities and indigenous or local communities, in the form of contracts, to adopt practices for soil conservation, choice of crops and farming practices, reduction or ban on clearing (Lavigne Delville, 2010; Mekouar, 2006; Plançon, 2009; Reij & Smaling, 2008; Sietz & Van Dijk, 2015)). The effectiveness of contractual arrangements as a response to land degradation varies depending on contract provisions. The contract holders can respond to reduce soil degradation, following a response hierarchy of prevention, mitigation and offsets (Adugna *et al.*, 2015).

Regulatory and legal responses to land degradation are in principle substantive and definitive, usually including specific preventive (fear of punishment) and curative (repair of environmental damage) measures. But how these measures have been operationalized in reality varies considerably, raising questions on their effectiveness (especially for the EU) (Paleari, 2017). The effectiveness of regulatory responses can depend on who is responsible for, who is impacted by, and the context of land degradation. For example, it was found that farmers in South-western Canada preferred voluntary policies (education, advice, grants) to reduce soil erosion and encourage soil conservation, even though they perceived regulatory approaches (penalties, cross-compliance, direct control) as being potentially more effective (Duff *et al.*, 1991).

In a study focused on the politics of land-use planning in Laos over the past three decades, Lestrelin *et al.* (2012) showed that land-use planning helped to reconcile different land uses, and interests among central and subnational governments, local actors, as well as national and foreign institutions. In another, multi-level analysis in Laos, Broegaard *et al.* (2017) found that cumulative effects of different legislations can reduce the potential positive impacts of legal reforms implemented to strengthen the rights of rural households (e.g., private property rights and planning processes). In a study of Wildlife Management Units in Mexico - with a focus on environmental policy instruments designed to promote ecosystem conservation and rural development via sustainable use of wildlife by local populations - Gómez-Aíza *et al.* (2017) highlighted the effectiveness of policy instruments as well as the importance of simultaneously adopting bottom-up and top-down management approaches. The protection of land depends on integrating the needs of local populations in policy instruments and understanding social vulnerabilities (McNeeley *et al.*, 2017).

Establishing protected areas to conserve biodiversity from human actions is a legal and regulatory response which often avoids land degradation. The management effectiveness of protected areas is discussed in Section 6.4.2.5.

6.4.2.2 Rights-based instruments and customary norms

A human rights-based approach in the fight against land degradation and desertification has been recognized as an important tool, because it brings together the legal strengths of international human rights and environmental law. This combination of laws can thus be used to combat land degradation and restore degraded lands at local to international levels.

The Global Mechanism of the UNCCD, for example, is supporting interested countries in the national land degradation neutrality target-setting programme, by helping to define national baselines, measures and targets to achieve land degradation neutrality (Orr *et al.*, 2017). Protecting human rights is one of the principles underpinning the vision of land degradation neutrality (Orr et al, 2017). The Voluntary Guidelines for Responsible Governance of Tenure of Land, Forests and Fisheries in the Context of National Food Security also applies existing governance standards, especially for human rights, to the management of land (Seufert, 2013; Windfuhr, 2016). Similarly, trade in agriculture and rights to food as human rights apply to land management (Cottier, 2006; Mechlem, 2006). What is unknown is whether and to what extent these human rights-based standards are taken into consideration as state parties take policy steps and make financial and human resource investments to achieve restoration of degraded lands.

Although the link between human rights and land degradation has been established in academic literature and soft law documents, it lacks legally-binding mechanisms at the international level, to operationalize the rights-based approach for restoration. In order to achieve Zero Net Land Degradation, legal and scientific literature has suggested the development of a global soil regime (Boer & Hannam, 2015; Lal *et al.*, 2012; UNCCD, 2012), that could take the form of a Protocol to the UNCCD and/or the Convention on Biological Diversity, or a separate convention focused on soil conservation.

A crucial element of a human rights-based approach to land degradation is the gender dimension (Lal 2000; UNCCD 2011). For example, in 2011 the UNCCD established an Advocacy Policy Framework on gender and “gender-sensitivity” - which is now seen as an important principle for achieving land degradation neutrality (Orr *et al.*, 2017). However, additional efforts (including financial support) will be needed to make sure that commitments on gender issues are actually implemented (Broeckhoven & Cliquet, 2015). The gender dimension of ecological restoration and benefits of mainstreaming it remain underexplored, but several recommendations have been made on how to improve it (Broeckhoven & Cliquet, 2015). They include using human rights instruments as a legal basis to push for greater involvement of women in restoration practices and for addressing underlying social and gender inequalities.

Empirical evidences from many developing countries suggest that halting resource (forest) degradation is possible and often effective when customary practices of local people and their rights to fulfil basic needs (e.g., fire wood, fodder) are incorporated in resource governance mechanisms (Agrawal & Ostrom 2001; Forest People Program & Program, 2010; Madrigal Cordero & Solis Rivera, 2012; Ostrom *et al.* 1999). States should ensure that policy, legal and organizational frameworks for tenure governance recognize and respect, in accordance with national laws, legitimate tenure rights (including those based on tenure) that are not currently protected by law (FAO, 2012).

It is important to recognize that customary practices (or local and/or indigenous practices) adopted by local people do have significance in halting land degradation and sustainable land management. Understanding the enabling socio-cultural factors – which could be defined on the basis of a rights-based approach, customary practices, and/or participatory processes – are instrumental to the success of land degradation or restoration responses. Thus, when designing responses to land degradation drivers or processes, local knowledge and customary practices should be given a high priority (Reed & Stringer, 2015).

6.4.2.3 Economic and financial instruments

Institutional, market and policy failures create differences in private and social costs, resulting in under-pricing of scarce resources (Panayotou, 1994) - including land and the associated goods and services it provides (Requier-Desjardins *et al.*, 2011). Externalities in land-use practices leads to socially sub-optimal, inefficient results (i.e., the costs of unsustainable land management practices are disproportionately borne by “off-site” parties who do not receive any compensation). Conversely, many sustainable land management practices benefit the public, whereas the costs of adopting them fall on the “on-site” actors (Low, 2013). Consequently, the actions taken by actors to avoid or reduce land degradation or to facilitate the adoption of sustainable land management practices would be less than socially desired due to such external effects (CBD, 2011).

Economic and financial instruments internalize such externalities from (un)sustainable land management practices into product price mainly through two types of incentive mechanisms: restrictive and supportive. Restrictive incentives for negative externalities (e.g., emission taxes, emission trading and quantity standards) are based on the polluter pays principle for negative externalities. Supportive incentives for positive externalities (e.g., subsidy and various types of payment for ecosystem services) are based on a beneficiary pays principle for positive externalities (Panayotou, 1994; Rode *et al.*, 2016).

The instruments to correct institutional, market and policy failures related to land degradation and restoration include the use of existing markets by inducing price changes (e.g., taxes, subsidy, bonds and so on) and/or the creation of new markets by providing new economic incentives (e.g., payment for ecosystem services, biodiversity offsets, conservation banking, natural capital accounting and so on.) (Initiative, 2015; Requier-Desjardins *et al.*, 2011; Sterner & Coria, 2012). The effectiveness of these instruments is highly context dependent, because of the interplay among broader socio-economic, institutional and policy environments - including the value systems and motivations of targeted actors (Beymer-Farris & Bassett, 2012; Kosoy & Corbera, 2010). In the following paragraphs, we synthesize empirical evidence on the use of these instruments and their effectiveness in avoiding, halting and reducing land degradation and restoring degraded lands.

Policy-induced price change

The effect of policy-induced price changes on halting land degradation or restoring degraded land depends on site-specific conditions. In some situations, higher agricultural commodity prices may encourage land management practices that accelerate degradation, especially when land tenure is insecure. In others, higher prices can provide scope for soil conservation measures that yield longer-term benefits. Examining the various interactions and trade-offs between agricultural development policy and land degradation, in the case of Sudan, Abdelgalil and Cohen (2001) found that four policies - namely price incentives, defined property rights, poverty reduction and enhanced human capital - were associated with reduced land degradation. While Zhao *et al.* (1991) found that commodity price distortions were associated with land degradation that negatively affected agricultural production in 28 developing countries, Pagiola (1996) found no simple relationship between price distortions and farmers' incentives to adopt soil conservation measures in developing countries. In Kenya, higher commodity prices incentivized farmers to adopt conservation measures on less productive steep slopes, but decreased investment on less steep slopes. In the Philippines, lower corn prices - after removing import tariffs - had the effect of conserving soil and reducing soil erosion in areas marginally suited to corn production (Briones, 2010). Similarly, European farm subsidies to meet good agricultural and environmental standards have been effective for erosion control, ground water management and increasing soil organic matter (Sklenicka *et al.*, 2015). These findings emphasize the importance of "getting prices right" and the need to adopt sustainable land and water management practices in agricultural production.

Payment for ecosystem services

Payment for ecosystem services, whereby services providers are financially rewarded by beneficiaries in return for otherwise "non-market" services, is a potentially economically-efficient way of achieving desired environmental and social outcomes. This instrument has been used in integrated conservation and development projects and can be effective in cases where proper institutional support is provided (Campos *et al.*, 2005; Engel *et al.*, 2008; Krause & Loft, 2013; Kroeger 2013; Wunder *et al.*, 2008; Zabel & Roe 2009). Allowing land managers to internalize some of the positive externalities created by sustainable land management - through payment for ecosystem services schemes - is seen as an important means to achieve land degradation neutrality (Mirzabaev *et al.*, 2015). In practice, these schemes have been financed by: (i) private beneficiaries of ecosystems services (i.e., individuals, organizations or companies), but are less common (Milder *et al.*, 2010; Sattler & Matzdorf, 2013; Tacconi, 2012); and (ii) governments or public agencies (e.g., agri-environmental programmes in the EU; Sattler & Matzdorf, 2013). The effectiveness of payment for ecosystem services schemes, however, varies considerably. The well-known Costa Rican programme is often considered as a successful case, because it had the effect of increasing forest cover and improving rural livelihoods (Porras *et al.*, 2014). The agri-environmental programmes in the EU are prone to adverse selection and moral hazards, reducing their effectiveness (Quillérou *et al.*, 2011; Quillérou & Fraser, 2010). The effectiveness of payment for ecosystem services schemes also depends on whether the payment is for temporary or permanent measures, with the latter generally being more effective.

Reducing emissions from deforestation and forest degradation in developing countries (REDD+) is a payment for ecosystem services scheme specifically focused on restoration of degraded forest land. Under REDD+ governments or multinational organizations compensate communities in developing countries for avoided deforestation and related climate-smart forest management. A recent review of the role of community-based forest management to achieve forest carbon benefits and social co-benefits suggests that REDD+ is likely to reduce forest degradation but not necessarily deforestation (Pelletier *et al.*, 2016). Some scholars argue that REDD+ is a cost-effective climate change mitigation policy (Komba & Muchapondwa, 2016), while others criticize REDD+ as a new conservation fad (Lund *et al.*, 2017; Redford *et al.*, 2013) that limits access to forests, compromises local people's customary rights (Poudel *et al.* 2014; West, 2012) and slows or reverses the promising trend of community-based forest management and governance in developing countries (Phelps *et al.*, 2010). The available evidence strongly suggests that the effectiveness of REDD+ to deliver climate change mitigation benefits - while reducing deforestation and forest degradation, biodiversity loss and providing social and economic "co-benefits" - depends on how its land management activities are implemented and the extent to which livelihood needs, governance, rights and social equity issues are addressed in REDD+ programme design, implementation and monitoring (Parrotta *et al.*, 2012).

Conservation tender or green auction among landholders, to act or manage the lands by adapting conservation practices, is considered an innovative payment for ecosystem services scheme (Latacz-Lohmann & der Hamsvoort, 1997; Latacz-Lohmann & Schilizzi, 2007). The oldest conservation tender programme is the Conservation Reserve Program in the USA which started in 1985 (USDA Farm Services Agency, 2011). Under the Conservation Reserve Program landowners' bids are ranked based on the Environmental Benefit Index: the ratio of ecological value of environmental benefits supplied and the value of the bid (Hanley *et al.*, 2012). In a review of the programme, Ferris and Siikamaki (2009) concluded that - even after about 25 years of implementation - it continues to be viewed positively by both conservation and agricultural communities. Farmers view that it is beneficial, because it is voluntary, does not transfer property rights, provides guaranteed income for the length of the contract and has the potential for supporting commodity prices by removing some land from production (Ferris & Siikamäki, 2009). Conservationists value the programme's conservation contributions such as habitat improvements, wildlife conservation and the provision of other ecosystem benefits (Ferris & Siikamäki, 2009). Like other OECD countries, Australia has also practiced conservation auction in the form of bush tender or eco-tender contracts (Eigenraam *et al.*, 2007; Stoneham *et al.*, 2003), landscape recovery auctions that include biodiversity and other environmental benefits (Hajkowicz *et al.*, 2007) and the Tasmanian Forest Conservation Fund (Binney and Zammit 2010). In a variety of land management and conservation contexts, scholars have found that bidding scheme for conservation contracts, to allocate government ecological funds, are practical, feasible and more cost-effective than fixed payment programmes (e.g., Connor *et al.*, 2008; Latacz-Lohmann & Schilizzi, 2007; Pannell *et al.*, 2001). They also claim efficiency gain on allocation of public funding through competitive bidding for ecological restoration.

However, payments for ecosystem services approaches may result in motivational "deadweight", providing unnecessary rewards for activities that would have occurred irrespective of payments (Beymer-Farris & Bassett, 2012; Kosoy & Corbera, 2010). For example, landholders who previously used sustainable land-use practices for various reasons would expect financial incentives under payment for ecosystem services schemes (Frey & Jegen, 2001; Reeson & Tisdell, 2008). To avoid such inefficiencies, engaging landholders in payment for ecosystem services programme design and the implementation of stewardship actions through cost-share programmes are considered by some to be more effective (Lukas, 2014; van Noordwijk & Leimona, 2010). Payments for ecosystem services approaches often promote economic values from a technocratic and economic perspectives and ignore indigenous and local knowledge and practices, human-nature relations and interactions, and social, cultural and spiritual values originated from such relations and interactions (Turnhout *et al.*, 2012, 2013), which need to be integrated in design and implementation of payment for ecosystem services schemes to enhance their effectiveness.

Biodiversity offsets

Biodiversity offset or ecological compensation has been introduced in many countries (OECD, 2016) to help balance economic development and environmental conservation goals. In principle, it is the last step in the mitigation (or response) hierarchy: avoid, minimize, restore and compensate (offset). One scenario of offsetting involves a developer - affecting land or habitat through activities such as mining, housing, industrial and infrastructural development (on the "impact site") - compensating for the resultant habitat loss by financing habitat restoration in a degraded land elsewhere (on the "offset site") of equivalent ecological value (Hahn *et al.*, 2015). From an economic perspective, offsetting is a combination of a cap (on habitat loss) and trade system in which the "spoiler" of habitats pays for restoration, possibly through a payment for ecosystem services scheme (Bull *et al.*, 2013; McKenney & Kiesecker, 2010; OECD, 2015). Offsets can be direct (on the ground actions) or indirect (e.g., funding for conservation programmes) and involve key concepts such as no net loss, additionality, permanence, timeframe, uncertainty, and monitoring and evaluation (BBOP, 2012; IUCN, 2014; Spash, 2015).

Biodiversity offsetting is common in the USA and Australia, while ecological compensation is common in the European Union where, for example, any loss of designated Natura 2000 sites must be compensated and this is done by government agencies on a case-by-case basis. The USA's wetland mitigation/banking, stream mitigation, and conservation banking programmes are among the world's largest offset programmes (OECD, 2016). Conservation banking involves legally-mandated biodiversity offsets, modelled after wetland banking (McKinney *et al.*, 2010). However, critics of the conservation banking system argue that the approach places too much focus on the compensation (offsetting) aspect and neglects earlier stages of the mitigation hierarchy (Hough & Robertson, 2009), resulting in a poor performance of the mechanism (Kihslinger, 2008; National Research Council, 2001). For example, an evaluation of 391 wetland offset projects in Massachusetts showed that 54% were not in compliance with the wetland regulations (Brown & Veneman, 2001). Similarly, Ambrose and Lee (2004) found that 46% of the 250 sites surveyed in California failed to replace key wetland ecosystem services. This could be due to the shortcomings of on-site and off-site compensatory mitigation provided directly by permittees, which has been substituted by wetland mitigation banking, a third party variation of off-site mitigation in recent years and also found to be more effective over the permittee-responsible mitigation (Briggs *et al.*, 2009; Orr *et al.*, 2017; Ruhl & Salzman, 2006). In Australia, biodiversity offsets have been widely used to compensate the residual impact of development, but the monitoring and verification of offset activities to achieve zero net loss remain inadequate (Martine Maron *et al.*, 2012; Office of the Auditor General Western Australia, 2017) and ecological compensation guidelines have often been neglected in practice (Briggs *et al.*, 2009; Coggan *et al.*, 2013). As a result, the effectiveness of offsets or compensation mechanisms to stop biodiversity loss remains debatable (Maron *et al.*, 2010, 2012, 2015). Similar to payments for ecosystem services approaches, biodiversity offsetting also promotes commodification of nature and economic values (Robertson, 2004; Turnhout *et al.*, 2013). For effective conservation and management of biodiversity through biodiversity offsetting, capturing and acting up on diverse forms of social values created and perpetuated through human-nature relations and interactions is essential (Turnhout *et al.*, 2012, 2013). Under the land degradation neutrality approach, the UNCCD's Science-Policy Interface recommends that ecological compensation should use land potential to ensure equivalence in exchange, and follow the response hierarchy of: avoid > reduce > reverse land degradation (Orr *et al.* 2017).

Property rights

Well-defined property rights on common property resources (e.g., forests and rangelands) and tenure security on agricultural lands are efficient ways to internalize externalities arising from these land uses (Panayotou, 1994). Halting forest and rangeland degradation through the adoption of community-based management - facilitated by common property regimes - has been successful in many places and contexts across the world (Agrawal & Ostrom, 2001; Ostrom, 1990, 1999). Establishing a land rental market for agricultural land could support sustainable farming (Sklenicka, 2016). For example, the emergence of land rental markets in central and eastern European countries, after 1990, helped to reduce land fragmentation and potential land degradation following the decommissioning of state farms (Sklenicka, 2016).

Although the costs of inaction in the face of global land degradation almost always outweigh the costs of actions (Giger *et al.*, 2015), a severe lack of investments on sustainable land management often persists, because appropriate effective incentive structures are virtually inexistent - especially for private landholders (Mirzabaev *et al.*, 2016). Box 6.8 presents various examples of the economics of land degradation and highlights the need for secure land tenure, information and market access, and appropriate incentive structure to halt or reverse land degradation.

Box 6.8 Case studies on economics of land degradation and improvement

In Sub-Saharan Africa, low livestock productivity was found to be a major cause of land degradation and conversion (rangeland to cropland) (Nkonya *et al.*, 2016). Results show that adoption of soil fertility enhancing practices, as a solution, requires improvement in market infrastructure (i.e., market access and advisory and extension services) along with the provision of appropriate incentive schemes (Nkonya *et al.*, 2016). As an incentive, conditional fertilizer subsidies were effective in promoting use of nitrogen-fixing trees in agroforestry systems.

In Central Asia, the key factors in promoting the adoption of sustainable land management practices include: better market access; access to extension; private land tenure; learning from other farmers; livestock ownership; lower household sizes; and lower dependency ratios (Alisher Mirzabaev *et al.*, 2016).

In an analysis of nationally-representative household surveys, Gebreselassie *et al.* (2016) found that access to agricultural extension services, secure land tenure and market access are important incentives for sustainable land management and its associated technologies. In addition, collective action to manage grazing lands and forests - fostered by local institutions - can successfully address land degradation.

In Niger, Moussa *et al.* (2016) found that enhancing government effectiveness - by giving communities a mandate to manage natural resources and incentivizing land users to benefit from their investment - played a key role in realizing simultaneous improvements in land management and human welfare.

In a total economic value-based study on the drivers of land degradation in India, Mythili and Goedecke, 2016 found that agricultural input subsidies and “decreasing land-man ratios” are two major determinants of land degradation at state levels - suggesting that reform of environmentally-harmful input subsidies is necessary. A similar study from Kenya, Tanzania and Malawi found that halting land degradation involves secured land tenure, improved market access and extension services on sustainable land management practices among agricultural households (Kirui, 2016; Mulinge *et al.*, 2016).

The Chinese national ecosystem assessment (2000-2010) reported that investment in restoration and preservation of natural capital has improved the provision of major ecosystem services at the national level, although with very little effect on habitat loss and environmental pollution (Ouyang *et al.*, 2016).

Natural Capital Accounting as a response to land and ecosystem degradation

Land degradation and loss of biodiversity are symptomatic of the failure to account fully for the value of natural capital in decisions made by individuals, businesses and governments (MA, 2005; Groot *et al.*, 2010). Natural capital accounting involves integrated physical and monetized accounts that show the type, quantities and qualities of the stocks of renewable and non-renewable natural assets, including land and biodiversity based assets - available and used, in a country or region - and the diversity of flows of services generated by them (ONS, 2017; TEEB, 2012). Examples include the UN’s System of Environmental-Economic Accounting (UN, 2014) and the World Bank’s Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES, 2017). Natural capital accounting has also been used to design and justify business responses to environmental pressures and corporate responsibilities, including the management of land and biodiversity impacts (TEEB, 2012) (see Section 6.4.2.4 on corporate social responsibility).

To date, most progress in natural capital accounting has been made in the development of physical accounts of asset stocks and service flows as a basis for subsequent valuation (Guerry *et al.*, 2015; UNDESA, 2017), usually with a focus on land use and conversion (EEA, 2016; EU, 2013), land and soil degradation (EEA, 2016; EU, 2014; Graves *et al.*, 2015; Robinson *et al.*, 2014; Robinson *et al.*, 2017) and biodiversity loss (UNEP-WCMC, 2016a). For example, natural capital accounting supported actions in the

Uganda National Development Plan II to restore degraded ecosystems (UNEP-WCMC, 2016b) - focusing on spatially-specific land cover, ecosystem extent, non-timber forest products and iconic mammals. Losses of natural ecosystems were associated with land conversion to agriculture, particularly for forests (29% remaining) and moist savannahs (32% remaining). From a policy response perspective, the accounts show that protected area designations performed well by avoiding the loss of natural ecosystems and securing benefits of managed wildlife tourism. Large areas of potentially natural vegetation were identified for sustainable harvesting of non-timber forest products, simultaneously maintaining species richness (UNEP-WCMC, 2016b).

The potential of natural capital accounting rests on the integration of physical and economic assessments (Remme *et al.*, 2014, 2015) in order to inform policy choice. Using the case of Kalimantan, Indonesia, Sumarga *et al.* (2015), show how natural capital and ecosystem accounting supports land-use planning through improved understanding of trade-offs between agriculture, forestry, carbon sequestration, wildlife and recreation services - especially when there is pressure to convert land to plantations. In the context of Small Island States, natural capital values, for international tourism, informed the introduction of a Green Departure Tax on tourists to fund protection of coastal biodiversity – for example, in the Republic of Palau, Micronesia (Weatherdon *et al.*, 2015). Hein *et al.* (2016) use cases of natural capital accounting from Europe and North America to value existing and likely future capacity to supply ecosystem services associated with, for example, soil organic carbon, timber harvesting and scenic views. Ruckelshaus *et al.* (2015) review experience of moving from natural capital accounting's "promise to practice", including its use in over 30 payment for ecosystem services and investment planning projects in Latin America (Box 6.9).

Despite numerous natural capital accounting initiatives and pilot projects, and the awareness it raises (Guerry *et al.*, 2015), the use of natural capital accounting for actual policy decisions remains relatively low, especially in developing countries (Edens & Graveland, 2014). A survey of 42 respondents from 17 countries (Virto *et al.*, 2018) showed that data availability and institutional barriers - including lack of political support and leadership - have constrained progress in adoption of natural capital and ecosystem accounting. In a first instance, rather than attempting to devise comprehensive natural capital accounting assessments of land-based ecosystems (Bartelmus, 2015), a staged, interactive approach focused on key indicators of land and biodiversity condition, as well as the economic consequences of change, may be more effective (Ruckelshaus *et al.*, 2015).

While mainstreaming natural capital has its supporters (Daily *et al.*, 2011; Remme *et al.*, 2015; Ruckelshaus *et al.*, 2015), the capitalisation of land and biodiversity values can: marginalize other culturally-resonant evaluative criteria (Sullivan, 2014); be confined to the "the nature that capital can see" or measure (Robertson, 2006); and serve to reinforce established worldviews, entitlements and practices dominated by political and economic imperatives (Robbins, 2012). Nonetheless, natural capital accounting can serve as a monitoring response to assess changes in the physical state and value of natural capital (land, biodiversity and ecosystem services) and as an evaluation tool to support decisions by governments and businesses - provided that an inclusive and collaborative approach is used to incorporate cultural and social values.

Box 6.9 Natural Capital, Ecosystem Accounting and Watershed Management in Colombia (Source: Ruckelshaus *et al.*, 2015)

Natural capital accounting was used to guide investment priorities and payments for watershed services under the Water for Life and Sustainability programme in Cali, Colombia. The programme was funded by water users, including sugar growers and producers, The Nature Conservancy and local NGOs. Working with stakeholders and drawing on biophysical data and local knowledge, a combination of simple scenario modelling and ranking of options was used to explore preferred watershed outcomes. Investment portfolios were drawn up, including options for grazing control, silvopastoralism, reforestation and restoration of degraded land. Working with available data, biophysical models contained in the *INvest* model were used to explore the effect of land-use change on erosion, sediment loss and/or retention and water yield. Options were assessed on their relative cost effectiveness to deliver target outcomes and then selected up to the limit of available funds. This more '"data and resource intensive", yet better targeted approach, gave an estimated threefold increase in return on investment for sediment retention compared with investments based on participants' general willingness to fund. Lessons from this experience are being used to support initiatives on over 30 new watershed funds in Latin America (Guerry *et al.*, 2015; Ruckelshaus *et al.*, 2015).

Figure 6 12 Mixed land-use mosaic and forest restoration in the Cali River Watershed, Colombia. Photo: courtesy of James Anderson under CC BY-NC-SA 2.0

These economic valuation and incentive-based instruments provide governments, NGOs and the private sector additional avenues to assess and avoid degradation of land, biodiversity and ecosystem services. However, a careful assessment of the limitations and suitability of these instruments is needed before using them in given social and cultural contexts. In policy practice, a mix of policies and regulations are usually required to define minimum environmental standards and restrictions on practices known to result in unacceptable environmental risk. By harnessing market forces to achieve intended outcomes, economic instruments are often used to complement, rather than substitute, legal and regulatory instruments and locally evolved institutions for environmental governance (Barton *et al.*, 2013; Cashore &

Howlett 2007). The current enthusiasm for monetization and market-based mechanisms in natural resource management - such as natural capital accounting and payment for ecosystem services - has potential for mobilizing new sources of funding for land degradation remedies; despite uneven access and fairness of these market-based mechanisms (Andersson *et al.*, 2011).

Benefits and costs of ecological restoration

Landowners, communities, governments and private investors need to understand the immediate and long-term costs and benefits of restoration activities in order to make optimal restoration investment decisions (BenDor *et al.*, 2015). The literature on full cost-benefit analyses of restoration projects is scarce (Aronson *et al.*, 2010; Bullock *et al.*, 2011): either restoration costs are not fully accounted for or the benefits to society are not examined in detail (De Groot *et al.*, 2013). For example, out of over 20,000 restoration case studies examined by The Economics of Ecosystem and Biodiversity initiative, only 96 studies provided meaningful cost data, with significant variations in costing methods and breadth and quality of cost-related information (NeBhoever *et al.*, 2011). Nevertheless, it is clear that restoration costs vary with restoration aims, timescales considered, the degree of degradation, ecosystem type and restoration methods used (Aronson *et al.*, 2010; Bullock *et al.*, 2011; Daily, 1995; De Groot *et al.*, 2013; NeBhoever *et al.*, 2011; UNCCD, 2017; Verdone & Seidl, 2017). Similarly, on the benefits end, most available studies often only considered financial benefits or private benefits (Barbier, 2007; De Groot *et al.*, 2013). Failure to incorporate a broader set of non-marketed values of restoration - such as the provision of wildlife habitat, climate change mitigation and other ecosystem services (Barbier, 2007; De Groot *et al.*, 2013) - discourages public and private investment in restoration projects (Verdone & Seidl, 2017). In addition, the use and choice of discount rates to assess present value of future benefits, an unresolved issue in the literature, affects net estimated benefits of restoration (Farber *et al.*, 2006). Some ecosystem service values cannot be monetized (e.g., cultural services that reflect spiritual values) and hence require a different approach than monetary valuation to estimate their value. However, recent advances in valuing non-marketed benefits of ecological restoration, and subsequent incorporation of such values and a wider range of social discount rates in cost-benefit analyses of restoration projects, still point to restoration investments being economically beneficial (De Groot *et al.*, 2013; Verdone & Seidl, 2017).

A study of fourteen Latin American countries estimated annual losses from desertification at 8-14% of agricultural gross domestic products (Morales *et al.*, 2011), while another study estimated the annual global cost of desertification at 1-10% of agricultural gross domestic products (Low, 2013). Using the benefit transfer method, Costanza *et al.* (2014) found that, across all biomes, the ecosystem service values lost due to land degradation and conversion ranges from $4.3 to $20.2 trillion per year. In a study that specifically considered only the values of managed forests (for wood, non-wood and carbon sequestration) and natural forests (for recreational values, passive use values and carbon sequestration values), Chiabai *et al.* (2011) estimated that projected degradation and land-use change would cost $1,180 trillion in forest ecosystem services, over a 50-year period (2000-2050). While these studies provide useful indications of the magnitude of land degradation costs, the many challenges in estimating the cost of land degradation at local and national scales remains a challenge for quantifying costs at the global level.

Box 6.10 Cost-benefit analyses of restoration

In a meta-analysis of restoration projects in over 200 studies that considered costs (i.e., direct costs, capital costs and management costs of restoration process, but not the opportunity costs) and known benefits (ecosystem services, not other indirect benefits), De Groot *et al.* (2013) reported that only 94 estimates on costs and 225 estimates on benefits of ecological restoration were found across 9 major biomes, including coastal systems, coastal wetlands, inland wetlands, freshwater rivers and/or lakes, tropical forests, temperate forests, woodlands and grasslands. The mean total economic value (in 2007 $/ha/yr) of all ecosystem services from these biomes were estimated at $28,917, $193,845, $25,682, $4,267, $5264, $3013, $2588, $2871, respectively. Cost estimates included original restoration costs, 5% per year maintenance costs as the financial costs of capital from year 2 onwards and 2.5% for coastal and wetland systems - whilst the benefits included the sum of the monetary values of 22 ecosystem services in the form of total economic value estimates. The project costs vary between several hundreds to thousands of $/ha (for grasslands, rangelands and forests) to several tens of thousands (inland waters) (Neßhöver *et al.*, 2011). De Groot *et al.* (2013) considered 12 alternatives scenarios: 6 based on 100% maximum restoration costs under 3 benefit scenarios (75%, 60% and 30% of the mean benefit values) and 2 discount rate scenarios (-2% and 8%); and 6 based on 75% maximum restoration costs under 3 benefit scenarios (75%, 60% and 30% of the mean benefit values) and 2 discount rate scenarios. Under all possible scenarios, the benefit-cost ratios were greater than 1.0 for inland wetlands, tropical forests, temperate forest, woodlands and grassland biomes – with the highest (35) for grasslands under a best-case scenario (75% restoration costs, 75% benefits at -2% discount rate), and less than 1 for coastal systems, freshwater, and coastal wetlands under a worst-case scenario (100% restoration costs, 30% benefits and 8% discount rate). While considering a slightly modified benefits (100% and 60% of total economic value), costs (100% and 130% of typical restoration costs), discount rate (-2%, 2% and 5%), and two-time horizons (20 years and 30 years) scenarios for the same 9 biomes, Blignaut *et al.* (2014) reported that the average benefit-cost ratio varies between 0.4 (for coastal systems) and 110 (for coastal wetlands) with most of the biomes at about 10 on average.

A recent cost-benefit analysis of the Bonn Challenge - a global initiative initiated in 2011 with the aim to restore 350 million hectares of degraded forest and agricultural land by 2030 - provides new insights on the value of investing in restoration (Verdone & Seidl, 2017). In this analysis, the extent of degraded area was based on the Global Assessment of Soil Degradation (GLASOD), calibrated to determine areas of degraded, managed and natural forests in each forest biome and across 12 world regions (Verdone & Seidl, 2017). It considered different benefit types (private, public or both), land degradation types (light, moderate, extreme or severe), forest management types (natural or managed) and discount rates (4.3% following Nordhaus, 2014 and 1.3% following Stern, 2007). In this analysis, average costs of restoration ranged from $214-3790/ha (mean: $1276 ± $887/ha); based on comprehensive data from a World Bank project database and TEEB reports for four degradation levels: *light* (mean - one standard deviation); *moderate* (mean); *severe* (mean + one standard deviation); and *extreme* (mean + 2 standard deviations). As one would expect, the average restoration costs increased with the extent of degradation: $389, $1276, $2163, and $3051/ha in the light, moderate, severe and extreme degradation categories, respectively (Verdone & Seidl, 2017) (cf. http://www.worldbank.org/projects and teebweb.org for more information). Estimated benefits of forest restoration, in terms of wood products (including wood fuel), were derived following Chiabai *et al.* (2011) - with adjustments for expected productivity losses of wood products due to degradation (i.e., 10%, 25%, 50% and 100% for light, moderate, severe and extreme degradation levels) (Daily, 1995). Benefits for services - including recreation and passive use benefits - were derived from a meta-analysis of 59 and 27 studies, respectively, and carbon sequestration benefits from a study on social costs of carbon sequestration ($43.46/ton) (Nordhaus, 2014). The results of this analysis suggest that achieving the Bonn Challenge target of restoring 46% of the world's currently degraded (managed and natural) forests would costs \$0.299 trillion - providing a net present value of benefits of \$2.254 trillion (benefit-cost ratio of 7.54, considering both private and public benefits from forests at a 4.3% discount rate), \$0.565 trillion (benefit-cost ratio 1.88, considering only private benefits at a 4.3% discount rate) and \$9.245 trillion (benefit-cost ratio 30.92, considering both private and public benefits at a 1.3% discount rate) (Verdone & Seidl, 2017). In the case of a "private benefits only" scenario, only 197 million ha could be profitably restored, and to meet Bonn Challenge restoration target governments would have to provide landowners a total subsidy of approximately \$139 billion or \$911/ha (also see Chapter 5, Section 5.2.3.4).

Conventionally, restoration is viewed by countries as a cost to be paid, rather than an investment that has tangible, beneficial returns (Bullock *et al.*, 2011). However, the available evidence strongly supports the view that restoration of degraded lands is a worthwhile investment that brings multiple benefits and can outweigh costs (Blignaut *et al.*, 2014; Bullock *et al.*, 2011). For example, in a study of large-scale landscape restoration in Mali, Sidibé *et al.* (2014) found that adapting agroforestry is economically beneficial at the local and global levels; providing local benefits to farmers in the range of \$5.2 to \$5.9 for every dollar invested and with net present values ranging between \$17.8 and \$62/ha/yr when discounted at 2.5%, 5%, and 10% over a time horizon of 25 years. When carbon sequestration is integrated in the analysis, practicing agroforestry and reforestation options yield up to \$13.6 of benefits for every dollar invested (at a discount rate of 5%), equivalent to a value of \$428.8/ha/year.

Investments in restoration have also been found to create jobs. Using an input-output model to estimate the direct, indirect (business to business) and induced (household spending) impacts of restoration on the economy in the USA, BenDor *et al.* (2015) analyzed 45 restoration programmes with an average programme cost of \$44.4 million. Their analysis indicated that the number of jobs created per \$1 million invested in restoration programmes range from 6.8 wetland restoration at county level (Department of the Interior, 2012) to 39.7 on national level forest, land and watershed restoration (Pollin *et al.* 2008). Moreover, the number of direct, indirect and induced jobs supported by these projects ranged from 14.6 per \$1 million invested for hydrologic reconnection, to 33.3 per \$1 million invested for invasive species removal. In the State of Oregon, the number of jobs supported by restoration projects ranged from an estimated 14.7 jobs/\$1million invested for in-stream restoration to 23.1 jobs per \$1 million invested for riparian restoration (Nielsen-Pincus & Moseley 2010). The employment multiplier ranged from 2.7 to 3.8 and economic output multipliers ranged from 1.9 to 2.4 for all projects. In Massachusetts, ecological restoration investment supported about 9.9 jobs per \$1 million for wetland restoration (with dam removal) to 12.9 jobs per \$1million invested for tidal creek recreation (Industrial Economics Inc., 2012).

The employment multiplier for the restoration industry ranged from 1.48 (Edwards *et al.*, 2013) to 2.87 (Shropshire & Wagner, 2009) and corresponding output multipliers are 1.60 and 2.59, respectively. The employment multiplier of restoration projects is comparable to that of other industries, including the oil and gas industry (Price Waterhouse Coopers, 2011), agriculture, livestock and outdoor recreation industry - with employment multipliers of 3.0, 2.33, 3.34 and 1.97, respectively (BenDor *et al.*, 2015). In a national survey of businesses that participate in restoration work in the USA, BenDor *et al.* (2015) estimated that direct employment of 126,000 workers generates \$9.5 billion in economic output (sales) annually. The indirect linkages and increased household spending - through restoration-related investment - accounts for 95,000 additional jobs and \$15 billion in economic output (BenDor *et al.*, 2015).

Despite the increasing awareness of the importance of natural ecosystems and sustainably managed working lands, in conserving biodiversity and providing ecosystem services - as well the social, economic and ecological benefits to be derived from rehabilitating degraded lands - investments in restoration are hampered by the typically short time horizon of private investment and land-use decisions, including low discount rates applied in economic analyses. For example, when forest restoration is viewed from a financial accounting lens that ignores public values and the intergenerational nature of forest restoration, it discourages investment despite the long-term societal benefits. Fulfilling large-scale restoration goals requires creating economic incentives and schemes (e.g., payments for ecosystem services and REDD+) that encourage landowners to recognize and capture public values of restoring degraded land, particularly in severely degraded landscapes.

6.4.2.4 Social and cultural instruments

Social and cultural instruments used to halt land degradation and restore degraded lands include: community-based (participatory) approaches in natural resource management; the integration of indigenous local knowledge and practices in land restoration and reclamation; public engagement and awareness-raising (eco-labelling, certification, education and/or training); corporate social responsibility; and voluntary agreements, amongst others. The complex and dynamic nature of land degradation drivers and processes requires flexible approaches to halt land degradation – which embrace a diversity of social and cultural knowledge and values from public and private sectors (Scherr, 2000; Shiferaw *et al.*, 2009).

Participatory approach in resource management and governance

Community-based natural resource management is a participatory approach for natural resource management and governance prevalent in many countries. It allows devolution of authority to local users to exercise their rights to manage and govern these resources. Decentralized community-based approaches have been proven effective in restoring degraded forests and conserving soils and water in many parts of the world (Agrawal & Ostrom, 2001; Ostrom *et al.*, 1999); including Australia, where involving indigenous communities in such approaches has been effective (Hill *et al.*, 2013; Pert *et al.*, 2015). In Nepal, the development and practice of community forestry since the late 1970s has been a successful response to halt deforestation and reduce the severity of associated soil erosion and landslides, prevalent in 1960-70s (Eckholm, 1976; Pandit & Bevilacqua, 2011) . This has involved devolution of forest management and governance authority to local forest users organised into “community forest user groups” (Acharya, 2002). As a result, it is estimated that the forest area in Nepal has increased from 37.4% in 1985-86 to 40.4% in 2015 (DFRS, 2015) (see Box 6.13).

Despite anecdotal evidence on the successes of community-based resource management, a meta-analysis of 41 studies from 13 countries in Asia, Africa and Central America focusing on three types of outcomes (forest condition and land cover, resource extraction and livelihoods) found that community- based forest management was associated with improved forest condition (i.e., greater tree density and basal area), but not with other indicators of global environmental benefits (Bowler *et al.*, 2012). The effectiveness of community forestry varies greatly with specific contexts, rights and management rules (Robinson *et al.*, 2014), and the main factors affecting effectiveness include forest area per person, level of monitoring and clarity regarding property rights (Nagendra, 2007; Pagdee *et al.*, 2006) .

Stakeholder participation in resource management and governance – supported by institutional structures and policies – can effectively facilitate interventions designed to halt land degradation or restore degraded lands (Reed & Dougill, 2008). For instance, improved land tenure in the Philippines has been associated with effective soil conservation (Briones, 2010), which in turn help to maintain land productivity and provide a form of safety net for farmers. On the other hand, scholars also note that a rise in insecure land tenure, involving both family and communal land, has been a major cause of unsustainable land use (Agrawal, 2002; Ostrom, 1990). Within community-based forest management or restoration programmes, Geist and Galatowitsch (1999) found that knowledge transfers in these programmes enhance social learning and self-esteem of the participants.

Cultural considerations on land use and management

Cultural context influences the choices that people make regarding land-use practices, in both long and short time frames. The drivers of land degradation from a cultural perspective include: changing cultural context of land; loss of cultural identities; and loss of cultural relevance of place-based indigenous and traditional ecological knowledge (Agrawal, 2002; Berkes, Colding, & Folke, 2000; Hartmann *et al.*, 2014; Ostrom, 1990; Parrotta & Trosper, 2012) (see Chapter 2, Section 2.2.2 and Chapter 3, Section 3.3.2.1).

Effective cultural responses to land degradation and restoration include the maintenance of traditional land-use practices and support for traditional knowledge which commonly underpins these practices (also see Sections 6.3.1.1 on agricultural practices and 6.3.1.2 on forestry practices). There is considerable evidence that the disparagement of the epistemological values and perspectives of traditional (particularly indigenous) communities that view nature/land and culture/values as indivisible (Claus *et al.*, 2015; Hartmann *et al.*, 2014), has been a major factor behind both the commercial exploitation and degradation of lands, as well as conservation measures that exclude traditional uses (Hartmann *et al.*, 2014; Parrotta & Trosper, 2012). The preservation or revival of ILK – and associated local and indigenous land-use practices – have been key to cultural resurgence and improvements in land management practices to avoid degradation in many parts of the world (Berkes, 2017; Berkes *et al.*, 2001; Corntassel & Bryce, 2012; Dublin et al, 2014; Parrotta & Trosper, 2012; Trosper, 2017; Ramakrishnan, 2002). Long *et al.* (2003) describes how youth ecology camps – where tribal adults teach youths how to care for their land – is an effective way to promote: restoration in more subtle ways; the passing on of cultural traditions sustaining the collective action needed for successful restoration work by providing a vision for restoration; a sense of place and community; and guidance for decision-making. In successfully opposing mining and logging operations on their traditional lands, many indigenous groups have also reproduced and transformed their identities and worlds (Poirier, 2010) through innovative practices around their land-based resources (Haglund *et al.*, 2011).

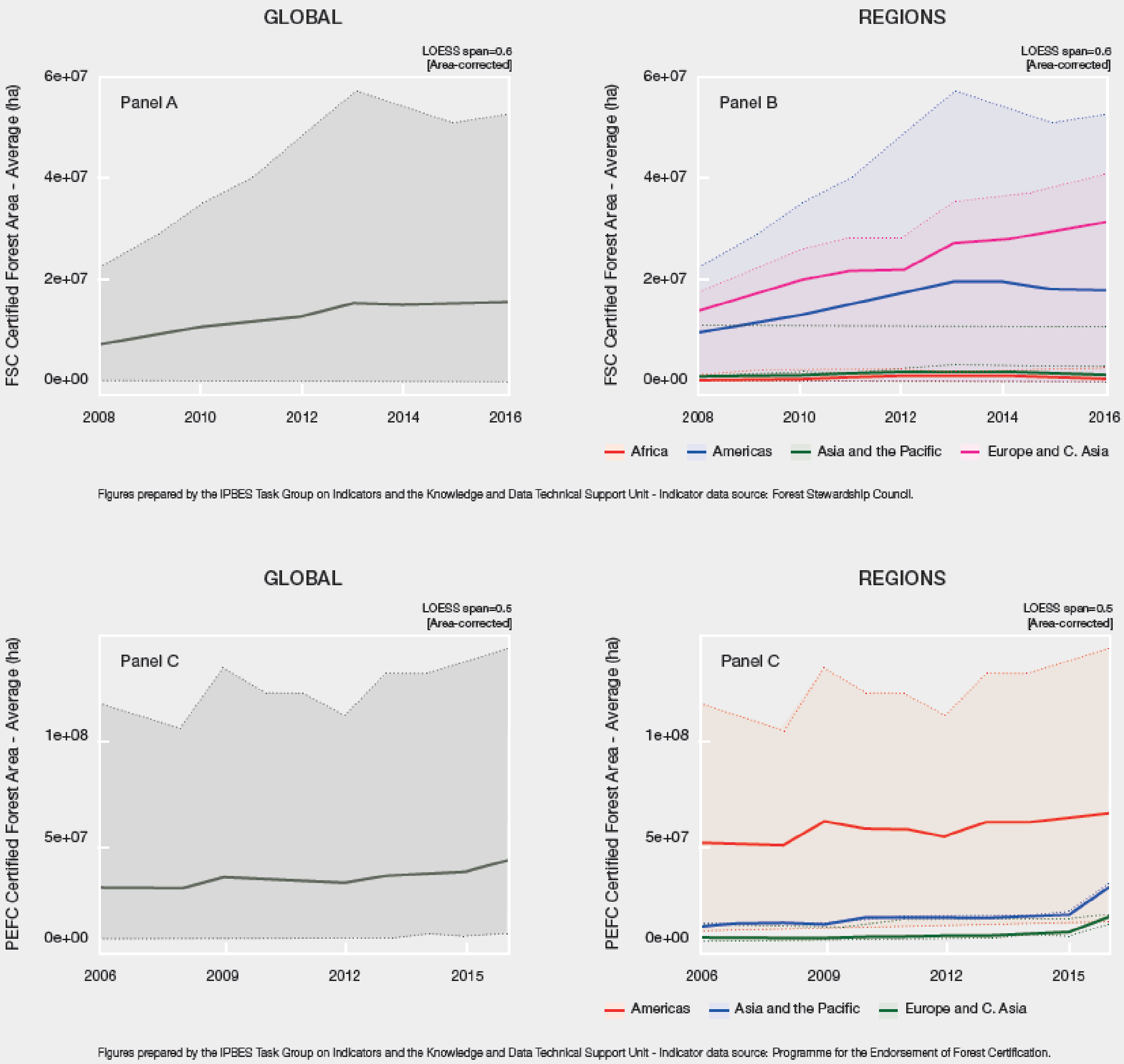
The adoption of soil conservation measures often faces cultural barriers when their implementation is perceived as a cost to local farmers, while benefits accrue at regional to global levels (Knowler & Bradshaw, 2007). Farmer decisions are strongly influenced by socio-economic factors (role of subsidies, quotas, cost savings) (Boardman *et al.*, 2003; Lahmar, 2010) and changing farmers' practices is a challenge for the adoption of voluntary soil conservation measures (Mbaga-Semgalawe & Folmer, 2000; Sattler & Nagel, 2010; Wauters *et al.*, 2010). In such contexts, participatory approaches have been found to be effective in promoting the adoption of soil conservation measures (Bewket, 2007), with economically- and environmentally-beneficial outcomes (Shiferaw & Holden, 2000).

A deliberate focus on otherwise "hidden" or "hard-to-value" cultural aspects such as the revitalization of ILK-based cultural practices (Hartmann *et al.*, 2014; Kittinger *et al.*, 2016) and faith-based beliefs (Cochrane, 2013) has been found to yield positive outcomes for halting and reversing land degradation. However, since ILK and associated natural resource management practices are influenced by history and contested locally, their representations within collaborative land restoration efforts can also trigger dissatisfaction amongst participants (Shepherd, 2010). For instance, the literature produced around the REDD+ programme has described how the matter of community tenure rights is also an extremely contentious issue given the inevitable vested interest of the dominant actors (e.g., government agencies, local elites) to maintain a dominance over land ownership (Ngendakumana & Bachange, 2013).

Certification

Eco-certification (or eco-labelling) is a voluntary instrument that has been applied to certain crops and forest products (e.g., coffee and timber). In principle, eco-certification enables consumers who prefer “green goods” to identify the good and purchase them in a price-differentiated market, which can address the environmental problems associated with production of goods by creating incentives for producers, otherwise difficult to handle with regulatory instruments alone (Lambin *et al.*, 2014). Studies examining the impacts of eco-certification schemes have found limited economic benefits of certification, but significant social and environmental benefits. In comparing certified and non-certified coffee growers and their land-use practices, certified coffee growers were found to be adopting environmental-friendly practices in Colombia (Rueda & Lambin, 2013) and they had a higher biodiverse coffee farms in Mexico (Mas & Dietsch, 2004). Eco-certification of forest products – through, for example, the Forest Stewardship Council (FSC) or the Program for the Endorsement of Forest Certification – provides some assurance that these products are from a responsibly managed forest (natural, semi-natural and plantations) with respect to: biodiversity conservation; the protection of critical ecosystem services; and the promotion of social, economic, cultural and ethical dimensions of sustainable forestry. While there is little evidence of positive environmental or socio-economic impacts of forest product certification, at the global level (Dauvergne & Lister, 2010), positive local impacts have been documented in Brazil, Malaysia and Indonesia (Durst *et al.*, 2006). In Indonesia, the effectiveness of FSC on social and environmental outcomes was evaluated using matching technique between FSC-certified timber concessions and non-certified logging concessions (Miteva *et al.*, 2015) . They estimated that between 2000 and 2008, FSC reduced aggregate deforestation by 5%. In addition, they note that FSC reduced firewood dependence by 33%, respiratory infections by 32%, and malnutrition by 1% on average across participating households (Miteva *et al.*, 2015). Figure 6.13 shows the area of certified forests under FSC and the Program for the Endorsement of Forest Certification schemes – indicating that certified forest area is on the rise at global and regional levels, with some regional differences. In 2016, Canada (>50 billion ha) and Finland (17 billion ha) had the greatest areas of certified forests, at the country level, under FSC and the Program for the Endorsement of Forest Certification schemes, respectively (IPBES, 2017).

Figure 6 13 Annual certified forest areas managed under Forest Stewardship Council (Panel A and B) and Endorsement of Forest Certification (Panel C and D) schemes at global and regional levels.



Corporate social responsibility

Among other forms of corporate social responsibility, natural capital accounting has also been used to design and justify business responses to environmental pressures and corporate responsibilities, including the management of land and biodiversity impacts (TEEB, 2012). Natural capital accounting broadly follows the accounting conventions of balance sheets and profit and loss accounts to reflect natural assets and service flows respectively, as well as exposure to natural capital risk (Trucost, 2013). Of particular interest is the Natural Capital Coalition (NCC, 2016), comprising over 250 collaborating organizations, which has produced The Natural Capital Protocol: a standardized framework supported by a toolkit to identify, measure and value impacts and dependencies of businesses on natural capital. The Coalition has assembled over 60 cases studies of natural capital accounting assessments and responses, half of which cover specific corporate applications and half covering topic- and location-specific cases (NCC, 2016). Many contain data and methods that may be applicable for use elsewhere. For example, Denkstatt (2016) used The Natural Capital Protocol to review water replenishment options for the Coca Cola Company showing, for example, that wetland restoration provided particularly high benefits beyond those linked to water conservation alone. Novartis, a multinational pharmaceutical company, used the Protocol to assess the monetized impact on natural capital for the Novartis Group and its supply chain (reported in NCC, 2016). For Novartis operations in Argentina, it was shown that alongside initiatives to improve energy and material use, contributions to forestry projects (prompted by the desire to offset the company's environmental footprint) generated net positive benefits through carbon sequestration, increased biodiversity and watershed protection. The approach has been integrated into the company's Financial Social and Environmental Accounting system and its Corporate Responsibility programme. In a similar vein, Hugo Boss used the natural capital accounting framework to assess the effects on ecosystems services of the supply chains for their cotton, wool and leather fashion goods (Zeller *et al.*, 2016). In their case, cotton cultivation and sheep farming accounted for large shares of monetized natural capital impacts for the clothing sector, while tanning processes dominated environmental costs for footwear. The assessment is being used to promote environmental provenance in the supply chain for their products, including the use of natural, less environmentally-burdensome substitute materials and processes. Despite these notable efforts, systematic reviews of the empirical evidence on direct correlation between corporate social responsibility and prevention of land degradation are scarce.

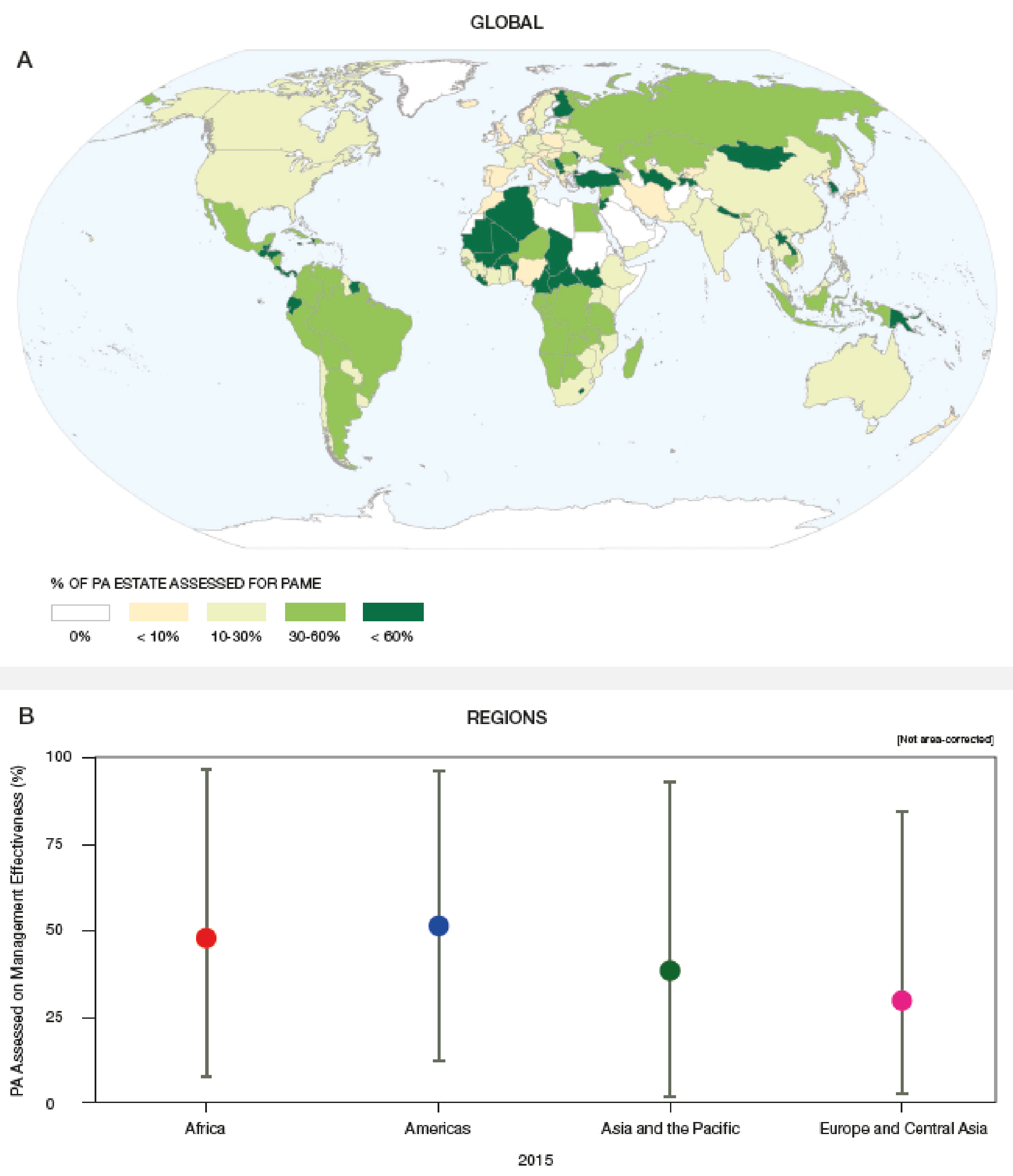
6.4.2.5 Protected areas

Protected areas are widely regarded as one of the most successful measures implemented for the conservation of biodiversity. The global community has committed to protect 17% of terrestrial areas by 2020, in line with Aichi Biodiversity Target 11 (Pringle, 2017; SCBD, 2014).

Since the mid-1990s, various methodologies have been developed for assessing protected area management effectiveness. Assessment data from all over the world have now been collated in the Global Database for Protected Area Management Effectiveness which contains records of almost 18000 assessments of protected area management effectiveness. The database includes information about the methodologies and indicators used, and records details of individual assessments. As of January 2015, nearly 18,000 of the assessments had been collated in the database, representing over 9000 protected areas, with 3,666 sites having multiple assessments. Some 17.5% of countries have already assessed the effectiveness of 60% of their protected areas. The differences in proportion of protected area assessed for effectiveness, by country and region, are given in Panel A and B in Figure 6.14.

Figure 6 14 Proportion of protected area assessed for management effectiveness by country (Panel A) and region (Panel B).

Source Panel A: Coad *et al.* (2015); Source Panel B: Figures prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit - Indicator data source: UNEP-WCMC and IUCN (2016), Protected Planet: The Global Database on. Protected Areas Management Effectiveness (GD-PAME) [On-line], [Oct 2016], Cambridge, UK: UNEP-WCMC and IUCN. Available at: www.protectedplanet.net



Empirical evidence on protected area management effectiveness is mixed. A systematic literature review of impact evaluation papers that used a composite-single indicator for measuring effectiveness, (Coad *et al.*, 2015) found a positive correlation between overall management performance score and biodiversity outcomes for 5 of the 9 reviewed final studies (Henschel *et al.*, 2014; Zimsky *et al.*, 2010, 2012). It remains unclear whether this lack of correlation with the impact of protected areas in some studies is real, meaning either that protected area management has no impact on biodiversity outcomes or more plausibly that good management (as measured by protected area management effectiveness scores) is necessary but not sufficient to ensure effective conservation (Carranza *et al.*, 2014).

Protected area effectiveness correlates with basic management activities such as enforcement, boundary demarcation and direct compensation for local communities – suggesting that even modest increases in funding would directly increase the ability of designated parks to protect tropical biodiversity (Bruner *et al.*, 2001). Further evidence indicates that the rate of conversion of landscape is lower in protected areas. Examining the impact of protected areas globally – by matching analysis of protected and unprotected areas – Joppa and Pfaff (2011) found that legal protection had reduced landscape conversion in 75% of 147 countries. Using the same matching technique, Andam *et al.* (2008) evaluated the impact on deforestation of Costa Rica's renowned protected area system (between 1960 and 1997) and found that protection reduced deforestation. They argued that approximately 10% of the protected forests would have been deforested had they not been protected. Based on an assessment of the impacts of anthropogenic threats to 93 protected areas in 22 tropical countries, the parks were found to be an effective mean to protect tropical biodiversity by stopping land clearing, and to a lesser degree by mitigating logging, hunting, fire and grazing (Bruner *et al.*, 2001). In Dana Reserve, Jordan, degradation has been partially reversed by agreeing with local farmers and herders to reduce stocking density of goats by 50% and providing alternative livelihood options through ecotourism and craft development (Schneider & Burnett, 2000).

On the other hand, protected areas are not always effective in halting land degradation. Liu *et al.* (2001) examined remotely-sensed data before and after the establishment of the Wolong Nature Reserve (established in south-western China to protect pandas) and found that habitat loss and fragmentation inside the reserve had unexpectedly increased to levels that were similar to or higher than those outside the reserve. Watson *et al.* (2014) reviewed the history and effectiveness of protected areas and found that conservation would be effective by establishing protected areas that are large, connected, well-funded and well-managed. Focusing on understanding causes of land degradation and deforestation in the Wildlife Reserve of Bontioli (Burkina Faso), Dimobe *et al.* (2015) found that despite the classification of two protected areas, vegetation cover was reduced over a 29-year period due to conversion of woodland and wooded savannahs to agricultural lands. They concluded that this was due to the lack of long-term adaptive management and conservation strategies in the communal areas and recommended strengthening the scientific foundation for greater involvement of local populations and staff in conservation and management activities.

Indigenous protected areas as a response

Globally, 18% of land is formally recognized as either owned by, or designated for, indigenous peoples and local communities. Within the 18%, 10% is owned by indigenous peoples and local communities and 8% is designated for (or "controlled by") indigenous peoples and local communities (Rights and Resources Initiative, 2015). For example, Australia has included Indigenous Protected Areas as a key part of the National Reserve System, in recognition that indigenous Australians have managed their country for tens of thousands of years. There are 70 dedicated Indigenous Protected Areas across 65 million hectares – accounting for more than 40% of the area of the National Reserve System – which protect biodiversity and cultural heritage and provide employment, education and training opportunities for indigenous people (The Natural Resource Management Council, 2010).

6.4.2.6 Climate change adaptation planning

Even though climate change is a threat in itself as well as a threat multiplier (see also Chapter 3), adapting to climate change to avoid land degradation impacts is closely linked to land-based resource management (of croplands, forests, rangelands, urban lands, wetlands and so on). Specific responses to climate change mitigation and adaption based on land-use types have been discussed in earlier sections (such as cropland in Section 6.3.1.1 and forests in Section 6.3.1.2). The focus in this section is on climate change adaptation planning, noting however that assessing its effectiveness in terms of avoidance of future impacts is difficult partly due to high uncertainty around climate change itself (Füssel, 2007).

Given the pervasive influence of climate change on socio-ecological systems, climate change adaptation planning has important implications for land resource management and conservation (Lawler, 2009). Climate change adaption depends on a variety of factors including: land-use domains; adaptation purpose, timing and planned horizon; form and measures of adaptation (i.e., technical, institutional, legal, educational and/or behavioural); actors (people at different hierarchy levels from farmers to many public and private organizations); and general context (environmental, economic, political and cultural). Thus there is no single best approach for assessing, planning and implementing climate change adaptation measures (Füssel, 2007).

To design, plan and implement effective adaptation measures, certain pre-conditions should be fulfilled (Füssel, 2007) and adaption barriers need to be systematically identified (Moser & Ekstrom, 2010). Such pre-conditions for effective climate change adaptation planning include: awareness of the problem; availability of adaptation measures; information about the measures; availability of resources to implement the measures; cultural acceptability of the measures; and incentives for implementing these measures (Füssel, 2007). To enhance effectiveness of climate change adaptation plans and strategies, Moser and Ekstrom (2010) proposed a framework to diagnose the barriers, which is underpinned by four principles and consists of three components. The four principles underpinning the framework are: (i) socially-focused but ecologically-constrained; (ii) actor-centric but context-aware; (iii) process-focused but outcome and/or action-oriented; and (iv) iterative and messy, but linear for convenience (Moser & Ekstrom, 2010). Three components to identify adaptation barriers include:

i. *process of adaptation* – understanding the barriers, planning adaptation options and managing the implementation of adaptation options;
ii. *structural elements of adaptation* – the actors, larger context in which they act (governance and broader human-biophysical environment) and the system of concern (the object or system upon which they act); and
iii. *overcoming the barriers through interventions* – spatial and/or jurisdictional and temporal barriers (Moser & Ekstrom, 2010).

The uncertain and varying nature of climate change impacts in different places and land-use systems necessitates adaptive management, which has often been referred to as a critical adaptation strategy for resource management (Lawler, 2009). A broader spatial approach (e.g., landscape or regional approach) and temporal perspective (e.g., scenario-based planning) has been argued for climate change adaption planning to manage land and ecosystems (Lawler, 2009; Peterson *et al.*, 2003). For example, scenario planning allows managers and planners to evaluate multiple potential scenarios of change, for a given system, in order to develop alternative management goals and strategies (Peterson *et al.*, 2003) – which in turn enhance the effectiveness of an adaptive management approach (Lawler, 2009). In the context of climate change and managing forests in the future, Millar *et al.* (2007) suggest that management strategies should promote both resistance and resilience to climate change impacts in forest ecosystems. For example, restoring ecosystem functions of a degraded land through restoration would increase resilience of the system (Julius *et al.*, 2008). Similarly, Harris *et al.* (2006) argue that a focus on ecosystem structure in restoration planning – in the context of changing climate – is challenging and that a focus on process (ecosystem services) rather than structure (species composition) may be a preferred option.

Many industrialized countries have developed comprehensive national adaptation assessments (e.g., the USA and Canada) (Lemmen & Warren, 2004; Scheraga & Furlow, 2001), while adaptation assessments in developing countries have usually been conducted as a part of bilateral or multilateral assistance schemes (Leary *et al.*, 2013) or the National Adaptation Program of Action processes. In addition, adaptation to climate change has been increasingly considered in regional- and local-level planning (e.g., regional forest management plan of Western Australia; see Conservation Commission of Western Australia, 2013; and the City of Melbourne Climate change adaptation strategy and action plan; see Commonwealth of Australia, 2013). However, in a systematic review of climate change adaptation literature comprised of 39 studies from developed countries between 2006 and 2009, Ford *et al.* (2011) found limited evidence of adaptation actions, even in developed nations. Those adaptation interventions that are found in practice are localized (municipality level) and funded through higher-level government interventions mostly concentrated on transportation, infrastructure and utility sectors and based on non-structural adaptation responses (i.e., management strategies, plans, policies, regulations, guidelines or operating frameworks to guide planning) (Ford *et al.*, 2011). In addition, their review highlighted that stakeholder engagement in adaptation planning and implementation, and adaptation actions did not focus on vulnerable populations (Ford *et al.*, 2011).

Addressing land degradation through climate change adaptation planning requires a broad-base integrated and adaptive approach involving all affected stakeholders. The failure to mainstream cultural and economic considerations – relevant to land degradation into environmental or other sector policies – has led to policy failures in many countries, including several in Africa (Kiage *et al.*, 2007; Koning & Smaling, 2005). As countries are affected differently by climate change-induced land degradation, adaptation plans and their effectiveness will vary depending on the socio-economic context of the place or system in question. For example, in a survey of 127 agro-pastoralist households in Kenya, Speranza *et al.* (2010) found that poverty limited any responses related to markets, while lack of skills limited adaptation capacity to droughts and climate change. They conclude that building adaptive capacity through extension services, maintaining infrastructure and embedding indigenous knowledge in adaptation plans would be effective adaptation measures for agro-pastoral communities (Speranza *et al.*, 2010). Indigenous communities have adapted to change for centuries and their practices and knowledge provide effective responses in land management responses (Fisher, 2013).

6.4.3 Integrated landscape approach as a response

Three main approaches have been used to respond to land degradation and land restoration through land planning at different scales: (i) sustainable land management; (ii) zoning; and (iii) integrated landscape planning and management. Although they share general motivations and objectives, they have different specific reaches.

Sustainable land management

In order to achieve socio-economical goals, sectoral policies typically have particular objectives when it comes to land, for example: agriculture and grazing consider soil quality, water availability and connectivity to markets; mining projects analyse the territory in terms of mining demands and mining stocks; transportation and energy infrastructure sectors focus on efficiency in terms of technical feasibility and competitiveness; while the housing sector considers urban expansion and land availability. Consequently, each policy has its own “map”, with a biased and fragmented approach to land. This fractional approach to social and environmental issues can result in overlapping maps and in inequitable and unsustainable use and transformation of land.

To address these limitations, spatial management responses to land degradation at national, regional and local levels need to combine and complement sectoral planning in ways that improve the resilience of socio-ecological systems, while supporting social and economic development, by using scientific evidence-based land-use information and tools. This goal can be achieved by delineating and modelling changing scenarios, and through the promotion of coordinated and concerted actions involving governments, private sectors and civil society.

The land-use planning (zoning) approach

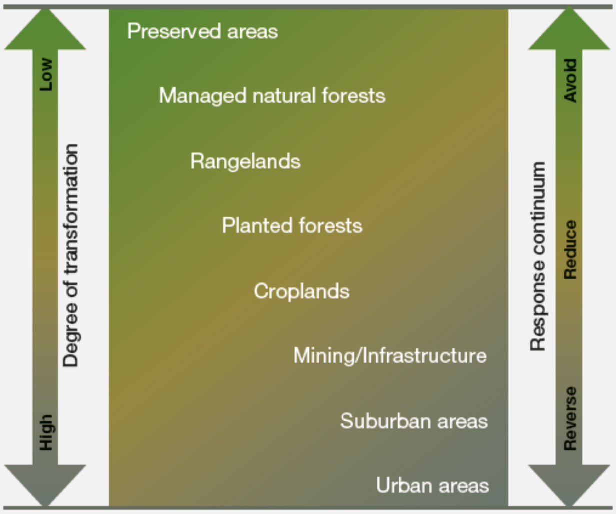
"Land-use planning is a systematic and iterative procedure carried out in order to create an enabling environment for sustainable development of land resources which meets people's needs and demands. It assesses the physical, socio-economic, institutional and legal potentials and constraints with respect to an optimal and sustainable use of land resources, and empowers people to make decisions about how to allocate those resources" (FAO & UNEP, 1995).

Land-use policies - which are often developed under spatial development frameworks at some administration level - involve spatial planning or zoning (i.e., allocation of the distribution, extent and intensity of land uses in a given landscape). Many jurisdictions have found that biodiversity conservation, sustainable resource management and the restoration of degraded habitats are best accomplished using a landscape-based approach. Pressures on the landscape and natural resources continue to grow due to increased population levels, urbanization and intensification of agriculture. An integrated, strategic landscape approach to biodiversity conservation is proving to be the most effective and efficient coordinate stewardship, resource management and planning activities.

Integrated landscape planning and management

An integrated landscape approach is a regulatory response to land-use planning and practice (see Section 6.4.2.1). It seeks to better understand the interactions between various land uses and stakeholders by integrating them in a joint management process (GLF, 2014) and is essential for development of sustainable land-use and livelihood strategies in rural areas (FAO, 2017). It allows for an encompassing consideration of a range of land uses in a given landscape – from pristine natural areas to highly transformed urban areas – into an integrated approach to make land-use decisions for multiple purposes and functions, as illustrated in Figure 6.15. Governments and organizations such as WWF, IUCN, and the World Bank argue that a landscape approach would bring environmental gains, enhance synergies and minimize trade-offs compared to sectoral approaches (e.g., agriculture, forestry, urban lands and so on) of managing lands within a resource-constrained context to reap more value from existing resources.

Figure 6 15 A schematic diagram showing the degree of land transformation (none or minimum in dark green colour to substantial transformation in dark grey colour) resembling land use types from preserved natural areas to urban areas with a response continuum (avoid, reduce and reverse).



Within the landscape approach for land conservation or restoration, scholars argue the merits of land sharing (i.e., wildlife-friendly farming) versus land sparing approaches (i.e., intensification of production to maximize agricultural yield) (Collas *et al.*, 2017; Law & Wilson, 2015; Mertz & Mertens, 2017; Phalan *et al.*, 2011). A landscape approach that embraces an integrated land-sharing philosophy has been increasingly promoted in science, and in practice, as an alternative to conventional, sectoral land-use planning, policy, governance and management. Sayer *et al.* (2013) have provided 10 principles for a landscape approach for reconciling agriculture, conservation and other competing land uses. They include: (i) continual learning and adaptive management; (ii) common concern entry point; (iii) multiple scales; (iv) multifunctionality; (v) multiple stakeholders; (vi) negotiated and transparent change logic; (vii) clarification of rights and responsibilities; (viii) participatory and user-friendly monitoring; (ix) resilience; and (x) strengthened stakeholder capacity (Sayer *et al.*, 2013).

Integrated landscape approaches may be effective for land resource management and governance for a number of reasons. They can correct the inability of sectoral approaches to: sufficiently address the interests of other sectors (such as nature protection versus livelihood needs of the poor); consider spatial spill-over effects of policies and decisions (i.e., decisions of a land use in one area is linked to environmental pollution, biodiversity loss, water shortage, erosion elsewhere within the landscape – downstream of a watershed, for example); or to better understand the linkages between humans and their surroundings (Arts *et al.*, 2017). For example, based on their analysis of the main environmental problems in mining areas, Lei and others (2016) recommend the utilization of a landscape strategy for planning and evaluating the ecological restoration and sustainable development of mining areas.

Role of the private sector

Businesses dependent on landscape resources have a central role to play in sustainable sourcing and collaborative actions to address water scarcity, biodiversity decline, deforestation and climate change (Goldstein *et al.*, 2012; Kissinger *et al.*, 2013; Natural Capital Declaration, 2015). There are notable examples of landscape-level restoration initiatives promoted by the private sector (WBCSD, 2016), such as the Landscapes for People, Food and Nature Initiative (http://peoplefoodandnature.org), and Commonland (http://www.commonland.com/en). However, out of 428 documented multi-stakeholder landscape partnerships, only a quarter involved private companies (Scherr *et al.*, 2017).

Nonetheless, experience indicates that initiatives for landscape restoration, sustainable farming, watershed management and natural capital accounting offer entry points for mutually beneficial cooperation, creating value, reducing risk and strengthening local relationships (Scherr *et al.*, 2017). Furthermore, natural capital accounting methods have facilitated multi-partner, private-public funding mechanisms for landscape initiatives (Shames *et al.*, 2014). For example, European supermarket chains, international development agencies and local non-government organizations came together to invest in enhancing natural capital through support for small farmers, soil and water conservation and wildlife protection in Kenya's Lake Naivasha Catchment (Shames *et al.*, 2014). Commonland brings together investors, companies, farmers and/or landholders for long-term, large-scale landscape restoration to create four types of returns from the land: inspiration, social capital, natural capital and financial capital. In a recent report of Community of Practice Financial Institutions and Natural Capital, formed by 15 financial organizations, van Leenders and Bor (2016) argue that although the project is in its early stages, financial institutions have been investing in natural capital to measure their impact and manage their risks while taking steps towards a green economy. Innovative financial instruments, such as green bonds and crowdfunding, can accelerate this transition (van Leenders & Bor, 2016).

Landscape governance

A key prerequisite for effective landscape governance – in view of halting or reversing land degradation – is the clarification of the spatial extent (territory) of the landscape to be conserved or restored and stakeholders involved (see Box 6.12). Several authors show that there has been a shift in considering the "territory" from a restricted involvement of only the actors who are technically supposed to conserve and/or restore the site, to a larger and more complex mosaic territory involving all the stakeholders concerned with the restoration site (Couix & Gonzalo-Turpin, 2015; Flores-Díaz *et al.*, 2014; Hobbs *et al.*, 2011; Petursdottir *et al.*, 2013; van Oosten *et al.*, 2014). This latter approach involves an appreciation of how people understand and value the place they live in (Flores-Díaz *et al.*, 2014), encourages citizens to reconnect to their place (van Oosten, 2013) and engages them in a process of "collective sense-making" (Couix & Gonzalo-Turpin, 2015).

Box 6.11 Restoration of Xingu watershed in the Amazon

The Xingu River is one of the Amazon's main tributaries. Its basin, in west-central Brazil, has 51 million hectares and is home to one of the largest conservation areas, the Xingu Indigenous Park, comprising of 24 indigenous groups (Schwartzman *et al.*, 2013). While the river channel is well protected within the Park, high deforestation rates have taken place in recent decades in the Xingu headwaters just outside the Park boundaries – mostly driven by cattle ranching and more recently by soybean production (Schwartzman *et al.*, 2013). Concerned about the degradation of water resources and the threat to the traditional ways of life within the Xingu basin, civil society organizations, indigenous organizations, state and municipal governments and farmers initiated the "Y Ikatu Xingu" campaign (YIX– "Save the Good Water of Xingu," in the Kamaiura language) (Schwartzman *et al.*, 2013).

The objectives of this forest restoration campaign included: conservation of water, fruit and wood production; carbon sequestration; and compliance with Brazilian environmental legislation (Durigan *et al.*, 2013). Forest restoration strategies were flexible and considered farmers' demands, motivations and farm facilities, as well as manpower, infrastructure and inputs. For forest restoration, direct seeding was deemed the appropriate method for tree establishment, and involved a mixture of green manure and seeds of forest species of different successional classes, applied and/or sown with the same tractors and implements used for crop and pasture cultivation (Campos-Filho *et al.*, 2013). This method of restoration was attractive to farmers, due to its low cost and familiarity of farmers and employees with the planting techniques and equipment. Also, since direct seeding requires large volumes of seeds (ca 400,000/ha), this approach stimulated the foundation of the Xingu Seed Network, formed by 420 indigenous and peasants collectors (Urzedo *et al.*, 2016). The Network produces 225 tree species and since 2007 has commercialized 137 tons of native seeds (www.sementesdoxingu.org.br). Five seed houses throughout the territory store seed lots and redistribute seeds to clients of the Y Ikatu Xingu restoration projects. Until now, the Y Ikatu Xingu Campaign has restored 900 ha using direct seeding, 300 ha by planting seedlings, and 1,500 ha by passive restoration (natural regeneration). The Y Ikatu Xingu Campaign is an example of a practical approach to large-scale restoration through law enforcement, shared governance and technological arrangements – ultimately leading to reductions in restoration costs, income generation and social mobilization.

Box 6.12 Landscape restoration and governance

Referring to landscape restoration, van Oosten *et al.* (2014) distinguish three modes of governance that steer decision-making:

- *Landscape governance as a management tool* – with a rather traditional hierarchical system of decision-making based on a central locus of authority, professional knowledge and binding regulation. Responsibilities can be shared among stakeholders, who can be considered co-managers of the system (generally in a well-defined system).

- *Landscape governance as a multi-stakeholder process* – in which attention is paid to new institutional interactions with increasing importance to private actors and soft law approaches, as well as local practices. It is most relevant in complex mosaic landscapes with delicate and politically-oriented decision-making. For example, between the forest and agricultural sector as it can enable better negotiation and conflict mediation.

- *Landscape governance as the creation of an institutional space* – in which actors from different sectors and scales create a new institutional space by creatively combining traditional and locally-embedded institutions, crafting hybrid institutions adapted to the specific socio-ecological characteristics. Such modes are most adapted to landscapes that stretch across administrative boundaries, scales and political entities.

6.4.4 Responses based on research and technology development

Global challenges associated with chronic land degradation – due to increasing populations, lack of fiscal or human resources, or inappropriate management decisions – have attracted numerous researchers from an array of disciplines to study the numerous underlying social, environmental and economic drivers and consequences (Bai *et al.*, 2008; Bojö, 1996; Conacher & Sala, 1998; Taddese, 2001). Most have concluded that appropriate land degradation responses can be developed and could be successful if research, improved local practices and appropriate institutional development activities become more widespread.

At a global level, UN organizations (e.g., UNCCD, UNEP, FAO), other multilateral agencies (e.g., WB, IFAD, WOCAT), research institutions (e.g., universities, and research centres) and government departments have all pursued research on how to avoid land degradation, restore degraded lands and develop human capital. These activities have resulted in numerous peer-reviewed and “grey” research reports and literature – providing excellent sources of information or knowledge on how to avoid and reduce further land degradation. Anthropogenic assets, including technology and infrastructure, are available for guiding improved land resource management (UNCCD, 2014). There has been significant progress towards the development of a conceptual framework for monitoring the progress of the UNCCD in addressing land degradation. For example, UNCCD decision 22/COP.11 has established a monitoring and evaluation approach consisting of: (i) progress indicators; (ii) a conceptual framework that allows the integration of indicators; and (iii) mechanisms for data sourcing and management at the national and/or local level (Low, 2013). Following this, India has developed a “desertification and land degradation atlas” by monitoring land use, processes of land degradation and severity levels between 2003-05 and 2011-13 (Space Applications Centre, 2016).

The spatial distribution of human capital (information, knowledge and skills) and technology have been influenced by socio-economic and technological factors – often leading to an uneven distribution among stakeholders (governments, communities and households). As a result, access to research knowledge and technology for sustainable land management or soil and water conservation and their adoption by land managers has been inconsistent. Therefore, in addition to research focused on soil degradation per se, the adaptive capacity of stakeholders also needs to be explored to determine what additional research and technology transfer investments are needed (UNEP, 2014). A recent assessment report on “unlocking the sustainable potential of land resources” concluded that improved land-use information systems and land-use planning and management are required to minimize the expansion of built-up land on fertile soils, and to invest in the restoration of degraded land (UNEP, 2016). This again points to integrated systems approaches, since efficient land management and major technological innovations (in agriculture) have potential to avoid a shortage of productive land while restoring degraded land (Lambin & Meyfroidt, 2011).

Advancements in technology and greater access to information are significantly increasing efforts to respond to land degradation problems more effectively. With appropriate data sources, new techniques based on land capability assessments can be used to monitor the extent and effects of both climate change and land degradation. Enhanced remote-sensing techniques have also made it possible to monitor the extent to which response options reduce or reverse degradation effects. Remote sensing has been used to monitor the provision of many ecosystem services including: provisioning, regulating, supporting and cultural services. However, determining specific degradation causes generally requires more detailed, field-level biophysical and socio-economic assessments, because of the wide range of factors that can cause any given change (Reed & Stringer, 2015). Furthermore, although several biophysical indicators can be monitored cost-effectively via remote sensing at broad spatial scales, field-based measurements are necessary to accurately interpret the data and establish cause and effect relationships (Reed & Stringer, 2015).

The combination of research, technology development and information transfer – initiated in the 1960s through the Green Revolution – has significantly contributed to increased production in food, feed and fibre for an ever-increasing global population (Khush, 1999). However, even though the revolution successfully enhanced productivity and income from farm-based communities, it unintentionally encouraged ecological destruction through unsustainable production practices – ultimately resulting in negative effects on the farm economy (Shiva, 1991). Therefore, to address sustainability issues while increasing per capita food production, combinations of technology with indigenous, traditional knowledge are needed (Conway & Barbier, 2013). One such example is the sloping agricultural land technology programme which has been very effective and popular in mountainous areas, such as the Loess Plateau of China and denuded uplands in Philippines, by conserving conserve soil and enhancing farm incomes (Sureshwaran *et al.*, 1996; Tacio, 1993; World Bank, 2007). Capacity-building of all stakeholders – from farmers to decision makers – is recognized as an effective means to combat land degradation and to achieve land degradation neutrality targets. This includes: the enhancement of scientific capacities to address key knowledge gaps; awareness-raising among decision makers and the general public; technology and knowledge transfer; and training. Perhaps the most significant need for capacity-building is in land resource management to deal with the complex issues of building efficient land information systems and sustainable institutional infrastructures, especially in developing countries and countries in transition (Enemark & Ahene, 2003). Given its pivotal role, several international organizations (such as FAO) and countless non-governmental organizations support capacity-building to combat land degradation worldwide. Among initiatives to support capacity-building to achieve land degradation neutrality, the Land Degradation Neutrality Target Setting Programme – conducted by the Global Mechanism of the UNCCD – currently supports 110 countries to set voluntary national targets (Orr et al, 2017) (see Chapter 8, Sections 8.2.1.1 and 8.4.3).

6.4.5 Responses based on institutional reforms

Land conservation and restoration policies have been implemented in a number of countries for several decades, leading to a growing body of assessments and comparative studies at different scales. Although many programmes derive from common international and national frameworks, several authors observe that similar legislation and policies can have very different outcomes depending on the existing local institutional arrangements (Hayes & Persha, 2010; He, 2014; Prager *et al.*, 2012; van Oosten *et al.*, 2014).

In recent years, the evolution of conservation or restoration policies beyond the traditional top-down state policies has led to a range of governance regimes and new institutional arrangements, with a transfer of responsibilities towards local governments and non-state actors (Agrawal *et al.*, 2008; Hayes & Persha, 2010). This decentralization can be more or less successful depending on the power transfer, accountability mechanisms and local participation involved (Ribot & Larson, 2005). Although effective stakeholder involvement is often cited as one of the main factors of success (France, 2016; Light, 2000), in practice, it is far from being systematic, often because of a lack of definition of who are the important stakeholders (Couix & Gonzalo-Turpin, 2015), and because formal institutions usually lack the flexibility and openness to cope with the more dynamic and innovative informal organizations. Furthermore, the history of community-based natural resource management suggests that simply understanding the value of local participation is complementary to reforming existing institutions or establishing new institution (e.g., community-based organizations, for example).

Governments, multilateral development banks, private sectors, and donor agencies have advanced various institutional models to engage local communities and others in reforestation, including partnerships with commercial plantations (Barr & Sayer, 2012). Such initiatives are supposed to generate benefits for rural communities, including employment, access to credit, low cost inputs (seeds, fertilizers and so on) and ready markets (Lamb, 2010). However, as many authors warn, diverging interests and power relations embedded in conservation or restoration are often overlooked in such arrangements (Baker *et al.*, 2014; Barr & Sayer, 2012; Bliss & Fischer, 2011; Hayes & Persha, 2010): Who really benefits from the resources? Who is actually able to make the rules? Who monitors and enforces the rules? The equitable distribution of burdens and benefits is probably the main challenge and the greatest obstacle to overcome in inter-institutional reform and decision-making processes.

Not all institutional arrangements for reforestation or restoration programmes are effective in generating greater benefits for local people. For example, reforestation programmes in the Asia Pacific, which are led by administration or corporate interests, have led to displacement of local communities, channelling international funding towards state elites, facilitated corruption or perverse incentives to convert secondary forests in plantations (Barr & Sayer, 2012). Local communities generally have little leverage in negotiating agreements with plantation companies or ensuring accountability (Barr & Sayer, 2012). Inequitable land-rental contracts and out-grower agreements, sometimes even forced onto the farmers, can have very detrimental effects on smallholders. People's involvement can be limited to handing over common lands and wage employment (Saxena, 1997) shaped by local power relations (Barr & Sayer, 2012).

One of the key aspects in institutional reform is guaranteeing tenure rights to local populations (Barr & Sayer 2012; Mansourian & Vallauri 2014; Williams & van Triest 2009). Although many programmes are put forward as community management, they are often limited by tenure uncertainty and non-participatory decision processes. For example, national forestry laws often recognize traditional tenure systems, but those rights are often subordinate to state claims over forest resources and few institutional mechanisms exist to resolve competing claims between state and customary systems (Vandergeest & Peluso, 2006). Conversely, in the Sloping Land Conversion Program in China, the institutional reform that secured long-term property rights over the restored land was found most effective compared to other incentives offered to engage locals in restoration (Grosjean & Kontoleon, 2009). However, formalization of private tenure can exclude the more marginalized populations, such as women or the "poorest of the poor" (Barr & Sayer, 2012). This points to the necessity of developing an approach to resolve competing claims between local communities managing land under customary tenure systems and state agencies relying on national codes, perhaps by at least committing to the principles of free, prior and informed consent of affected communities (Barr & Sayer, 2012).

Several studies show that innovative types of collaborative network governance are emerging that bring together natural resource users, NGOs, concerned citizens, private corporations and various branches of government. Such arrangement can accommodate, numerous initiatives within a large-scale framework (Adams *et al.*, 2016; France, 2016; Petursdottir *et al.*, 2013; Pinto *et al.*, 2014). These forums or advisory committees ensure the representation of the different interests at stake. However, as underlined by Baker *et al.* (2014), there are still limited studies in which these interests are articulated and negotiated. Too many programmes are still focused on end-products and not enough on the developmental process and social learning that such networks enable, to build true adaptive capacity (Pahl-Wostl, 2006; Zedler *et al.*, 2012).

Box 6.13 Community Forest User Group: Reformed institution to manage forests in the hills of Nepal

The practice of forest management in the hills of Nepal shows how institutional reform help to address deforestation and restore degraded forest lands. Until 1957, before forests were nationalized, forests in the hills of Nepal were protected and managed by nearby villagers for generations based on customary practices. Even though the forest nationalization in 1957 had good intention to cease large tracts of forests hold by ruling class, it sent a wrong signal to ordinary villagers in the hills resulting in policy failure and a trigger for rampant deforestation. During the 1960s, the Nepalese government adopted a "command and control" approach to halt deforestation, but failed due to inadequate institutional capacity – leading to continued deforestation and degradation of hill slopes with increased problems of landslides and soil erosion (Pandit & Bevilacqua, 2011). This phenomenon of forest degradation and soil erosion is famously described in the form of "Himalayan Degradation Hypothesis" by Eckholm (1976).

To address the deforestation, forest degradation and soil erosion problems in the hills, by 1978 the Nepalese government reformed forest policy and initiated a new institution to manage hill forests based on a bottom-up and participatory approach, now commonly referred to as "community forest user group". This approach transferred forest-use rights to "forest user groups" and reconnected them with their nearby forests – named as community forests – with a sense of ownership (HMG/N, 1993), allowing "forest user groups" to develop rules (i.e., constitution of community forest user group) to manage the forest based on a collective forest management plan and share the benefits amongst themselves (HMG/ADB/FINNIDA, 1988). With the inception of a new institution, and reformed forest policy in 1978, degraded hills were extensively planted with the mobilization of local users. Due to its success in the hills, community forestry became a nationwide programme since 1993. By 2015, a total of 1,798,733 ha of forests (approximately 30.85% of total forest area in Nepal) have been managed by 18,960 "community forest user groups", benefitting nearly 2,392,755 households (DoF, 2015). As shown in Figure 6.16, community forestry programmes have transformed many degraded hills into productive forests and have either halted or at least reduced deforestation, and associated land degradation. Forest statistics of Nepal indicate that forest cover decreased from about 38% of country's land mass (147,181 Km^2) in 1978/79 to about 37.4% in 1985/86, which then increased to about 38.3% in 1995, and 44.74% (covering 59,624.38 Km^2, of which 40.36% forests and 4.38% shrub lands) in 2015 (DFRS, 2015). Most of this gain in forest cover has been in the hills where community forestry programme has been in operation since 1978; initially as Panchayat, or Panchayat Protected forest, and later as community forestry.

Figure 6 16 Restored degraded hill forest in Nepal (right panel) through community forestry programme. The degraded site is showcased on the left panel. Site: Dandapakhar, Sindhupalchok district. Photo: Courtesy of Fritz Berger on behalf of Nepal Swiss Community Forestry Project (2011).

In 1975 In 2010

6.5 Knowledge gaps and research needs

There currently exists a deep and broad base of knowledge and experience to support sustainable land management and soil and water conservation, biodiversity conservation and restoration practices, as well as a rapidly developing understanding of the importance of policies, institutions and governance responses in providing an enabling environment for effective responses to land degradation and its drivers. There is enormous potential for applying this existing knowledge more widely, given adequate support by decision makers, land managers and the general public. Nonetheless, there remains a number of key areas where significantly enhanced effort - by the research and development communities, farmers and other land managers, planners and decision makers - is required to halt and reverse current land degradation trends.

Further work is needed to:

- Develop analytical methodologies and tools to better understand and quantify the full range of values (nature's contributions to people) people derive from land (and ecosystems), the short- medium- and long-range costs associated with biodiversity loss and degradation, as well as costs and benefits associated with avoiding, mitigating and reversing land degradation;

- Provide knowledge, tools and skills (by the scientific community) on land condition monitoring for land managers and planners - both conventional and ILK-based approaches, including citizen science;
- Bridge, among and within countries, current gaps in knowledge and skills, capacity and resources needed by landowners, communities and governmental land management agencies to effectively halt land degradation and restore degraded lands - through, for example, the development of easily accessible geospatial land information systems, and enhanced North-South, South-South and triangular knowledge-sharing, research and development activities;
- Better understand the conditions under which indigenous and local knowledge and practices, for sustainable land management and restoration, can be used more extensively, and how such knowledge and practice can better inform the development of strategies and specific technologies for sustainably managing croplands, rangelands, forests, wetlands and urban lands;
- Develop policies that encourage sustainable land use at the landscape level, in a coordinated and integrated fashion among development sectors; and
- Better understand which policy instruments, institutional and governance systems are most effective for avoiding, reducing and reversing land degradation under local environmental, social, cultural and economic conditions. Addressing land degradation issues at a local level, by aligning policies and instruments that could generate benefits on multiple scales, is fundamentally important for the success of restoration responses in conserving biodiversity, providing ecosystem services and supporting livelihoods.

6.6 References

Abdelgalil, E. A., & Cohen, S. I. (2001). Policy modelling of the trade-off between agricultural development and land degradation - the Sudan case. *Journal of Policy Modeling*, *23*(8), 847–874.

Aber, J., Pavri, F., & Aber, S. (2012). Environmental Cycles and Feedback. In *Wetland environments: a global perspective*. Chichester: John Wiley & Sons.

Acharya, K. P. (2002). Twenty-four years of community forestry in Nepal. *International Forestry Review*, *4*(2), 149–156. https://doi.org/10.1505/IFOR.4.2.149.17447

Acton, J. P. (1989). *Understanding Superfund*. Santa Monica, CA. Retrieved from https://www.rand.org/pubs/research_briefs/RB9010.html

Adams, W. M., Hodge, I. D., Macgregor, N. A., & Sandbrook, L. C. (2016). Creating restoration landscapes: Partnerships in large-scale conservation in the UK. *Ecology and Society, 21*(3), 1. https://doi.org/10.5751/ES-08498-210301

Adiansyah, J. S., Rosano, M., Vink, S., & Keir, G. (2015). A framework for a sustainable approach to mine tailings management: Disposal strategies. *Journal of Cleaner Production*, *108*, 1–13. https://doi.org/10.1016/j.jclepro.2015.07.139

Adkins, S., & Shabbir, A. (2014). Biology, ecology and management of the invasive parthenium weed (Parthenium hysterophorus L.). *Pest Management Science*, *70*(7), 1023–1029. https://doi.org/10.1002/ps.3708

Adugna, A., Abegaz, A., & Cerdà, A. (2015). Soil erosion assessment and control in Northeast Wollega, Ethiopia. *Solid Earth Discussions*, *7*(4), 3511–3540.

Agrawal, A. (2002). Common Resources and Institutional Sustainability. In E. Ostrom, T. Dietz, N. Dolsak, P. Stern, S. Stonich, & E. Weber (Eds.), *The Drama of the Commons* (pp. 41–86). Washington, DC: National academy Press. Retrieved from www.nap.edu

Agrawal, A., Chhatre, A., & Hardin, R. (2008). Changing governance of world's forests. *Science*, *320*, 1460–1462.

Agrawal, A., & Ostrom, E. (2001). Collective Action, Property Rights, and Decentralization in Resource Use in India and Nepal. *Politics & Society*, *29*(4), 485–514. https://doi.org/10.1177/0032329201029004002

Aide, T. M., Clark, M. L., Grau, H. R., López-Carr, D., Levy, M. A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Núñez, M. J., & Muñiz, M. (2013). Deforestation and Reforestation of Latin America and the Caribbean (2001-2010). *Biotropica*, *45*(2), 262–271. https://doi.org/10.1111/j.1744-7429.2012.00908.x

Aide, T. M., & Grau, H. R. (2004). Globalization, migration and Latin American ecosistems. *Science*, *305*(5692), 1915–1916. https://doi.org/10.1126/science.1103179

Albrecht, A., & Kandji, S. T. (2003). Carbon sequestration in tropical agroforestry systems. *Agriculture, Ecosystems & Environment*, *99*(1), 15–27. https://doi.org/10.1016/S0167-8809(03)00138-5

Alexander, S., & McInnes, R. (2012). *The benefits of wetland restoration. Ramsar Scientific and Technical Briefing Note No. 4*. Retrieved from http://www.ramsar.org/sites/default/files/documents/library/bn4-en.pdf

Allendorf, F. W., & Lundquist, L. L. (2003). Introduction: Population biology, evolution, and control of Invasive Species. *Conservation Biology*, *17*(1), 24–30. https://doi.org/10.1046/j.1523-1739.2003.02365.x

Alterman, R. (1997). The Challenge of Farmland Preservation: Lessons from a Six-Nation Comparison.

Journal of the American Planning Association, *63*(2), 220–243. https://doi.org/10.1080/01944369708975916

Altieri, M. A. (2002). Agroecology: the science of natural resource management for poor farmers in marginal environments. *Agriculture, Ecosystems & Environment*, *93*(1–3), 1–24. https://doi.org/10.1016/S0167-8809(02)00085-3

Altieri, M. A. (2004). Linking ecologists and tradition in the search for sustainable agriculture. *ESA*, *2*(1), 35–42.

Altieri, M. A., Funes-Monzote, F. R., & Petersen, P. (2012). Agroecologically efficient agricultural systems for smallholder farmers: contributions to food sovereignty. *Agronomy for Sustainable Development*, *32*(1), 1–13. https://doi.org/10.1007/s13593-011-0065-6

Altieri, M. A., & Toledo, V. M. (2011). The agroecological revolution in Latin America: rescuing nature, ensuring food sovereignty and empowering peasants. *The Journal of Peasant Studies, 38*(3), 587-612. https://doi.org/10.1080/03066150.2011.582947

Ambrose, R.F., & Lee, S. F. (2004). *An evaluation of compensatory mitigation projects permitted under Clean Water Act Section 401 by the Los Angeles Regional Quality Control Board, 1991-2002. Report prepared for the Los Angeles Regional Water Quality Control Board.* Los Angeles.

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., & Anderson, P. (2017). An overview of the progress and challenges of peatland restoration in Western Europe. *Restoration Ecology*, *25*(2), 271–282. https://doi.org/10.1111/rec.12415

Andrews, S. S., Karlen, D. L., & Cambardella, C. A. (2004). The Soil Management Assessment Framework. *Soil Science Society of America Journal*, *68*(6), 1945. https://doi.org/10.2136/sssaj2004.1945

Anker, Y., Hershkovitz, Y., Bendor, E., & Gasith, A. (2014). Application of Aerial Digital Photography for Macrophyte Cover and Composition Survey in Small Rural Streams. *River Research and Applications*, *30*(7), 925–937. https://doi.org/10.1002/rra

Anker, Y., Rosenthal, E., Shulman, H., & Flexer, A. (2009). Runoff geochemical evolution of the hypersaline Lower Jordan Valley basin. *Israel Journal of Earth Sciences*, *58*(1), 41–61. https://doi.org/10.1560/IJES.58.1.41

Arnalds, O., & Barkarson, B. . (2003). Soil erosion and land use policy in Iceland in relation to sheep grazing and government subsidies. *Environmental Science & Policy*, *6*(1), 105–113. https://doi.org/10.1016/S1462-9011(02)00115-6

Aronson, J., & Alexander, S. (2013). Steering Towards Sustainability Requires More Ecological Restoration. *Natureza & Conservacao*, *11*(2), 127–137.

Aronson, M. F., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., & Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment*, *15*(4), 189–196. https://doi.org/10.1002/fee.1480

Artmann, M. (2014). Assessment of Soil Sealing Management Responses, Strategies, and Targets Toward Ecologically Sustainable Urban Land Use Management. *AMBIO*, *43*(4), 530–541. https://doi.org/10.1007/s13280-014-0511-1

Arts, B., Buizer, M., Horlings, L., Ingram, V., van Oosten, C., & Opdam, P. (2017). Landscape Approaches: A State-of-the-Art Review. *Annual Review of Environment and Resources*, *42*(1), annurev-environ-102016-060932. https://doi.org/10.1146/annurev-environ-102016-060932

Asmelash, F., Bekele, T., & Birhane, E. (2016). The potential role of arbuscular mycorrhizal fungi in the restoration of degraded lands. *Frontiers in Microbiology*, *7*(JUL), 1–15. https://doi.org/10.3389/fmicb.2016.01095

Bae, J. S., Joo, R. W., & Kim, Y.-S. (2012). Forest transition in South Korea: Reality, path and drivers. *Land Use Policy*, *29*(1), 198–207. https://doi.org/10.1016/j.landusepol.2011.06.007

Bai, Z. G., Dent, D. L., Olsson, L., & Schaepman, M. E. (2008). Proxy global assessment of land degradation. *Soil Use and Management*, *24*(3), 223–234. https://doi.org/10.1111/j.1475-2743.2008.00169.x

Bailis, R., Drigo, R., Ghilardi, A., & Masera, O. (2015). The carbon footprint of traditional woodfuels. *Nature Climate Change*, *5*(3), 266–272. https://doi.org/10.1038/nclimate2491

Baker, S., Eckerberg, K., & Zachrisson, A. (2014). Political science and ecological restoration. *Environmental Politics*, *23*(3), 509–524. https://doi.org/10.1080/09644016.2013.835201

Banks-Leite, C., Pardini, R., Tambosi, L. R., Pearse, W. D., Bueno, A. A., Bruscagin, R. T., Condez, T. H., Dixo, M., Igari, A. T., Martensen, A. C., & Metzger, J. P. (2014). Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. *Science*, *345*(6200), 1041–1045. https://doi.org/10.1126/science.1255768

Banning, N. C., Gleeson, D. B., Grigg, A. H., Grant, C. D., Andersen, G. L., Brodie, E. L., & Murphy, D. V. (2011). Soil microbial community successional patterns during forest ecosystem restoration. *Applied and Environmental Microbiology*, *77*(17), 6158–6164. https://doi.org/10.1128/AEM.00764-11

Barr, C. M., & Sayer, J. A. (2012). The political economy of reforestation and forest restoration in Asia-Pacific: Critical issues for REDD+. *Biological Conservation, 154*, 9-19. https://doi.org/10.1016/j.biocon.2012.03.020

Bartelmus, P. (2015). Do We Need Ecosystem Accounts? *Ecological Economics*, *118*, 292–298.

Bartley, R., Corfield, J. P., Abbott, B. N., Hawdon, A. A., Wilkinson, S. N., & Nelson, B. (2010). Impacts of improved grazing land management on sediment yields, Part 1: Hillslope processes. *Journal of Hydrology*, *389*(3–4), 237–248. https://doi.org/10.1016/j.jhydrol.2010.05.002

Battles, J. J., Fahey, T. J., Driscoll, C. T., Blum, J. D., & Johnson, C. E. (2014). Restoring Soil Calcium Reverses Forest Decline. *Environmental Science & Technology Letters*, *1*(1), 15–19. https://doi.org/10.1021/ez400033d

Baylis, K., Peplow, S., Rausser, G., & Simon, L. (2008). Agri-environmental policies in the EU and United States: A comparison. *Ecological Economics*, *65*(4), 753–764. https://doi.org/10.1016/j.ecolecon.2007.07.034

BBOP. (2012). *Part 2: Principles with Criteria and Indicators*. Washington D.C., USA: Business and Biodiversity Offsets Programme. Retrieved http://www.forest-trends.org/documents/files/doc_3078.pdf

Behnke, R. H., Mortimore, M. (2016). *The End of Desertification? Disputing Environmental Change in the Drylands*. Berlin: Springer Earth System Sciences. https://doi.org/10.1007/978-3-642-16014-1

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science*, *325*(5944), 1121–1124. https://doi.org/10.1126/science.1172460

Benefield, C. B., DiTomaso, J. M., Kyser, G. B., Orloff, S. B., Churches, K. R., Marcum, D. B., & Nader, G. A. (1999). Success of mowing to control yellow starthistle depends on timing and plant's branching form. *California Agriculture*, *53*(2), 17–21. https://doi.org/10.3733/ca.v053n02p17

Berkes, F., Colding, J., & Folke, C. (2000). Rediscovvery of Traditional Ecological Knowledge as Adaptive Management. *Ecological Applications*, *10*(5), 1251–1262. https://doi.org/10.1890/1051-0761(2000)010[1251:ROTEKA]2.0.CO;2

Bestelmeyer, B. T., Ellison, A. M., Fraser, W. R., Gorman, K. B., Holbrook, S. J., Laney, C. M., Ohman, M. D., Peters, D. P. C., Pillsbury, F. C., Rassweiler, A., Schmitt, R. J., & Sharma, S. (2011). Analysis of abrupt transitions in ecological systems. *Ecosphere*, *2*(12), art129. https://doi.org/10.1890/ES11-00216.1

Bewket, W. (2007). Soil and water conservation intervention with conventional technologies in northwestern highlands of Ethiopia: Acceptance and adoption by farmers. *Land Use Policy*, *24*(2), 404–416. https://doi.org/10.1016/j.landusepol.2006.05.004

Beymer-Farris, B. A., & Bassett, T. J. (2012). The REDD menace: Resurgent protectionism in Tanzania's mangrove forests. *Global Environmental Change*, *22*(2), 332–341. https://doi.org/10.1016/j.gloenvcha.2011.11.006

Bhattacharya, A., Oppenheim, J., & Stern, N. (2015). Driving Sustainable Development Through Better Infrastructure : Key Elements of a Transformation Program. *Global Economy & Development*. Working Paper 91. Retrieved from https://www.brookings.edu/wp-content/uploads/2016/07/07-sustainable-development-infrastructure-v2.pdf

Bienkowski, P., & van der Steen, E. (2001). Tibes, Trade, and Towns: A New Framework for the Late Iron Age in Southern Jordan and the Negev. *Bulletin of the American School of Oriental Research*, *323*(323), 21–47.

Billet, P. (2008). L'indemnité compensatoire des contraintes environnementales dans les zones d'érosion et certaines zones de protection des aires d'alimentation des captages. *Revue de Droit Rural*, *364*.

Binney, J., & Charlie Zammit. (2010). Australia: The Tasmanian Forest Conservation Fund. In K. Karousakis & Quiller Brooke (Eds.), *Paying for Biodiversity: Enhancing the Cost-Effectiveness of Payments for Ecosystem Services* (pp. 131–160). OECD.

Biological, A. R., Ginocchio, R., & Baker, A. J. M. (2004). Metallophytes in Latin America : a remarkable biological and genetic resource scarcely known and studied in the region. *Revista Chilena de Historia Natural, 77*, 185–194.

Bird, D. N., & Boysen, E. (2007). *The Carbon Sequestration Potential from Afforestation in Ontario*. Research Information Note. Note Number 5. Ontario. Retrieved from http://www.climateontario.ca/MNR_Publications/276909.pdf

Birkhofer, K., Bezemer, T. M., Bloem, J., Bonkowski, M., Christensen, S., Dubois, D., & Ekelund, Fleming Fliessbach, Andreas; Gunst, Lucie and Hedlund, Katarina LU, *et al.* (2008). Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity. *Soil Biology & Biochemistry*, *40*(9), 2297–2308.

Black, A. L., Brown, P. L., Halvorson, A. D., & Siddoway, F. H. (1981). Dryland cropping strategies for efficient water-use to control saline seeps in the northern great plains, u.s.a. *Developments in Agricultural Engineering*, *2*(C), 295–311. https://doi.org/10.1016/B978-0-444-41999-6.50020-3

Black, R., Stephen, R., Bennett, G., Thomas, S. M., & Beddington, J. R. (2011). Mobility can bring opportunities for coping with. *Nature*, *478*, 447–449.

Bliss, J. C., & Fischer, A. P. (2011). Toward a political ecology of ecosystem restoration. In *Human Dimensions of Ecological Restoration* (pp. 135–148). Island Press/Center for Resource Economics.

Boardman, J., Poesen, J., & Evans, R. (2003). Socio-economic factors in soil erosion and conservation. *Environmental Science & Policy*, *6*(1), 1–6. https://doi.org/10.1016/S1462-9011(02)00120-X

Bodiguel, L. (2014). Lutter contre le changement climatique : le nouveau leitmotiv de la politique agricole commune. *Revue de l'Union Européenne*, (580), 414–426.

Boer, B., & Hannam, I. (2015). Developing a Global Soil Regime. *International Journal of Rural Law and Policy*, *2015*, *No.*(1), 1–13.

Bojö, J. (1996). The costs of land degradation in Sub-Saharan Africa. *Ecological Economics*, *16*(2), 161–173. https://doi.org/10.1016/0921-8009(95)00087-9

Bommarco, R., Kleijn, D. & Potts, S. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends Ecol. Evol.*, 28, 230–238.

Bongers, F., Chazdon, R., Poorter, L., & Peña-Claros, M. (2015). The potential of secondary forests. *Science*, *348*(6235).

Botoni, E., & Reij, C. (2009). *La transformation silencieuse de l'environnement et des systèmes de production au Sahel: Impacts des investisements publics et privés dans la gestion des ressouces naturelles*. Centre for International Cooperation, Comité permanent Inter-États de Lutte contre la Sécheresse dans le Sahel. Retrieved from http://www.agrhymet.ne/portailCC/images/pdf/Rapport%20Synthse_Etude_Sahel%20Final.pdf

Boy, G., & Witt, A. (2013). *Invasive alien plants and their management in Africa*. UNEP/GEF Removing Barriers to Invasive Plant Management Project. International Coordination Unit, CABI Africa. Retrieved from https://www.cabi.org/Uploads/CABI/publishing/promotional-materials/african-invasives-book.pdf

Boyer, S., Blakemore, R. J., & Wratten, S. D. (2011). An integrative taxonomic approach to the identification of three new New Zealand endemic earthworm species (Acanthodrilidae, Octochaetidae: Oligochaeta). *Zootaxa*, *32*(2994), 21–32.

Boyer, S., & Wratten, S. D. (2010). The potential of earthworms to restore ecosystem services after opencast mining - A review. *Basic and Applied Ecology*, *11*(3), 196–203. https://doi.org/10.1016/j.baae.2009.12.005

Brancalion, P. H. S., Melo, F. P. L., Tabarelli, M., & Rodrigues, R. R. (2013). Biodiversity persistence in highly human-modified tropical landscapes depends on ecological restoration. *Tropical Conservation Science*, *6*(6), 705–710.

Brancalion, P. H. S., Pinto, S. R., Pugliese, L., Padovezi, A., Ribeiro Rodrigues, R., Calmon, M., Carrascosa, H., Castro, P., & Mesquita, B. (2016). Governance innovations from a multi-stakeholder coalition to implement large-scale Forest Restoration in Brazil. *World Development Perspectives*, *3*, 15–17. https://doi.org/10.1016/J.WDP.2016.11.003

Bredemeier, B., von Haaren, C., Rüter, S., Reich, M., & Meise, T. (2015). Evaluating the nature conservation value of field habitats: A model approach for targeting agri-environmental measures and projecting their effects. *Ecological Modelling*, *295*, 113–122. https://doi.org/10.1016/j.ecolmodel.2014.08.010

Briggs, B. D. J., Hill, D. A., & Gillespie, R. (2009). Habitat banking-how it could work in the UK. *Journal for Nature Conservation*, *17*(2), 114–124. https://doi.org/10.1016/j.jnc.2008.12.006

Briones, R. M. (2010). Addressing Land Degradation: Benefits, Costs, and Policy Directions. *Philippine Journal of Development*, *37*(1), 41–79.

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation*, *17*(5), 925–951. https://doi.org/10.1007/s10531-008-9380-x

Broeckhoven, N., & Cliquet, A. (2015). Gender and ecological restoration: time to connect the dots. *Restoration Ecology*, *23*(6), 729–736. https://doi.org/10.1111/rec.12270

Broegaard, R. B., Vongvisouk, T., & Mertz, O. (2017). Contradictory Land Use Plans and Policies in Laos: Tenure Security and the Threat of Exclusion. *World Development*, *89*, 170–183. https://doi.org/10.1016/j.worlddev.2016.08.008

Brondízio, E. S. (2008). *New Book Highlights the Story of the International Boom in Amazon's Açaí Fruit. Announcing the Publication of The Amazonian Caboclo and the Açaí Palm: Forest Farmers in the Global Market*. Retrieved from https://www.nybg.org/content/uploads/2017/08/AcaiPalmbookrelease6-2-08.pdf

Brown, M. E. (2008). *Famine Early Warning Systems and Remote Sensing Data*. Heidelberg: Springer Verlag.

Brown, M. E., Funk, C., Pedreros, D., Korecha, D., Lemma, M., Rowland, J., Williams, E., & Verdin, J. (2017). A climate trend analysis of Ethiopia: examining subseasonal climate impacts on crops and pasture conditions. *Climatic Change*, *142*(1–2), 169–182. https://doi.org/10.1007/s10584-017-1948-6

Brown, R. R., & Davies, P. (2007). Understanding community receptivity to water re-use: Ku-ring-gai Council case study. *Water Science & Technology*, *55*(4), 283–290.

Brown, S. C., & Veneman, P. L. M. (2001). Effectiveness of compensatory wetland mitigation in Massachusetts, USA. *Wetlands*, *21*(4), 508–518. https://doi.org/10.1672/0277-5212(2001)021[0508:EOCWMI]2.0.CO;2

Brudvig, L. A., Leroux, S. J., Albert, C. H., Bruna, E. M., Davies, K. F., Ewers, R. M., Levey, D. J., Pardini, R., & Resasco, J. (2017). Evaluating conceptual models of landscape change. *Ecography*, *40*(1), 74–84. https://doi.org/10.1111/ecog.02543

Bruner, A. G., Gullison, R. E., Rice, R. E., & da Fonseca, G. A. B. (2001). Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, *291*(5501). Retrieved from http://science.sciencemag.org/content/291/5501/125

Brunette, V., & Germain, R. H. (2003). Forest management in the New York City Watershed. In *XII World Forest Congress*. Quebec, Canada.

Buchholz, T. A., & Madary, D. A. (2016). Stream Restoration in Urban Environments: Concept, Design Principles and Case Studies of Stream Daylighting. In *Sustainable Water Management in Urban Environments*. Retrieved from http://lar.vt.edu/wp-content/uploads/2016/03/Buchholz-Stream-Restoration-Revision-4.compressed.pdf

Buckingham, K., & Hanson, C. (2013). *The Restoration Diagnostic Case Example: China Loess Plateau Case Example: China Loess Plateau*. Retrieved from https://www.wri.org/sites/default/files/WRI_Restoration_Diagnostic_Case_Example_China.pdf

Buckley, Y. M. (2008). The role of research for integrated management of invasive species, invaded landscapes and communities. *Journal of Applied Ecology*, *45*(2), 397–402. https://doi.org/10.1111/j.1365-2664.2008.01471.x

Bull, J. W., Suttle, K. B., Gordon, A., Singh, N. J., & Milner-Gulland, E. J. (2013). Biodiversity offsets in theory and practice. *Oryx*, *47*(3), 369–380. https://doi.org/10.1017/S003060531200172X

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution*, *26*(10), 541–549. https://doi.org/10.1016/j.tree.2011.06.011

Buongiorno, J., & Zhu, S. (2014). Assessing the impact of planted forests on the global forest economy. *New Zealand Journal of Forestry Science*, *44*. https://doi.org/10.1186/1179-5395-44-s1-s2

Bureau, J.-C., & Thoyer, S. (2014). *La politique agricole commune*. La Découverte. Retrieved from http://cairnrevues.ezproxy.univ-ubs.fr/la-politique-agricole-commune--9782707182104.htm

Burney, J. A., Davis, S. J., & Lobell, D. B. (2010). Greenhouse gas mitigation by agricultural intensification. *PNAS*, *107*(26), 12052–12057.

Claus, C. A., Osterhoudt, S., Baker, L., Cortesi, L., Hebdon, C., Zhang, A., & Dove, M. R. (2015). Disaster, Degradation, Dystopia. In R. L. Bryant (Ed.), *The International Handbook of Political Ecology* (pp. 291–304). Edward Elgar Publishing.

Cadman, T., Maraseni, T., Breakey, H., López-Casero, F., & Ma, H. O. (2016). Governance values in the climate change regime: Stakeholder perceptions of REDD+ legitimacy at the national level. *Forests*, *7*(10). https://doi.org/10.3390/f7100212

Cai, Q. (2002). The relationships between soil erosion and human activities on the Loess Plateau. In J. Juren (Ed.), *Sustainable Utilisation of Global Soil and Water resources. Technology and Method of Soil and Water conservation.* (pp. 112–118). Beijing: Proceedings of 12th International Soil Conservation Organisation Conf., Beijing, May 26–31, 2002. Vol. 3. Tsinghua University Press.

Calvo-Alvarado, J., McLennan, B., Sanchez-Azofeifa, A., & Garvin, T. (2009). Deforestation and forest restoration in Guanacaste, Costa Rica: Putting conservation policies in context. *Forest Ecology and Management*, *258*(6), 931–940. https://doi.org/10.1016/j.foreco.2008.10.035

Cambardella, C. A., Delate, K., & Jaynes, D. B. (2015). Water Quality in Organic Systems. *Sustainable Agriculture Research*, *4*(3), 60. https://doi.org/10.5539/sar.v4n3p60

Campos-Filho, E. M., Costa, J. N. M. N. Da, Sousa, O.; Junqueira, R. G. P., Da Costa, J. N. M. N., De Sousa, O. L., & Junqueira, R. G. P. (2013). Mechanized Direct-Seeding of Native Forests in Xingu , Central Brazil. *Journal of Sustainable Forestry*, *32*(7), 702–727. https://doi.org/10.1080/10549811.2013.817341

Campos, J. J., Alpízar, F., Louman, B., Parrotta, J., Porras, I. T., & Support, A. F. ecosystem services (FES) are fundamental for the E. life. (2005). An Integrated Approach to Forest Ecosystem Services. In *Global Forum* (2nd ed., pp. 97–116).

Canadell, J. G., & Raupach, M. R. (2008). Managing Forests for Climate Change Mitigation. *Science*, *320*(5882), 1456–1457. https://doi.org/10.1126/science.1155458

Caprotti, F., Cowley, R., Datta, A., Broto, V. C., Gao, E., Georgeson, L., Herrick, C., Odendaal, N., & Joss, S. (2017). The New Urban Agenda: key opportunities and challenges for policy and practice. *Urban Research & Practice*, *10*(3), 367–378. https://doi.org/10.1080/17535069.2016.1275618

Carvalho, S. B., Velo-Antón, G., Tarroso, P., Portela, A. P., Barata, M., Carranza, S., Moritz, C., & Possingham, H. P. (2017). Spatial conservation prioritization of biodiversity spanning the evolutionary continuum. *Nature Ecology & Evolution*, *1*(6), 151. https://doi.org/10.1038/s41559-017-0151

Cashore, B., & Howlett, M. (2007). Punctuating Which Equilibrium? Understanding Thermostatic Policy Dynamics in Pacific Northwest Forestry. *American Journal of Political Science*, *51*(3), 532–551. https://doi.org/10.1111/j.1540-5907.2007.00266.x

CBD (2011). *Memorandum of Understanding between the Convention on Biological Diversity and the United Conventions to Combat Desertification regarding the Joint Work Plan 2011-2012*. Retrieved from http://www.cbd.int/doc/agreements/agmt-unccd-2011-09-03-mou-web-en.pdf

CGIAR. (2016). World Water Day 2016. Retrieved April 10, 2017, from http://drylandsystems.cgiar.org/tags/world-water-day-2016

Charudattan, R., & Dinoor, A. (2000). Biological control of weeds using plant pathogens: accomplishments and limitations. *Crop Protection*, *19*(8), 691–695. https://doi.org/10.1016/S0261-2194(00)00092-2

Chasek, P., Safriel, U., Shikongo, S., & Fuhrman, V. F. (2015). Operationalizing Zero Net Land Degradation: The next stage in international efforts to combat desertification? *Journal of Arid Environments*, *112*, 5–13. https://doi.org/10.1016/j.jaridenv.2014.05.020

Chazdon, R. L., Broadbent, E. N., Rozendaal, D. M. A., Bongers, F., Zambrano, A. M. A., Aide, T. M., Balvanera, P., Becknell, J. M., Boukili, V., Brancalion, P. H. S., Craven, D., Almeida-Cortez, J. S., Cabral, G. A. L., de Jong, B., Denslow, J. S., Dent, D. H., DeWalt, S. J., Dupuy, J. M., Durán, S. M., Espírito-Santo, M. M., Fandino, M. C., César, R. G., Hall, J. S., Hernández-Stefanoni, J. L., Jakovac, C. C., Junqueira, A. B., Kennard, D., Letcher, S. G., Lohbeck, M., Martínez-Ramos, M., Massoca, P., Meave,

J. A., Mesquita, R., Mora, F., Muñoz, R., Muscarella, R., Nunes, Y. R. F., Ochoa-Gaona, S., Orihuela-Belmonte, E., Peña-Claros, M., Pérez-García, E. A., Piotto, D., Powers, J. S., Rodríguez-Velázquez, J., Romero-Pérez, I. E., Ruíz, J., Saldarriaga, J. G., Sanchez-Azofeifa, A., Schwartz, N. B., Steininger, M. K., Swenson, N. G., Uriarte, M., van Breugel, M., van der Wal, H., Veloso, M. D. M., Vester, H., Vieira, I. C. G., Bentos, T. V., Williamson, G. B., & Poorter, L. (2016). Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Science Advances*, *2*(5), e1501639.

Chazdon, R. L., Harvey, C. A., Komar, O., Griffith, D. M., Ferguson, B. G., Martinez-Ramos, M., Morales, H., Nigh, R., Soto-Pinto, L., van Breugel, M., & Philpott, S. M. (2009). Beyond Reserves: A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. *Biotropica*, *41*(2), 142–153. https://doi.org/10.1111/j.1744-7429.2008.00471.x

Chen, L., Wei, W., Fu, B., & Lu, Y. (2007). Soil and water conservation on the Loess Plateau in China: review and perspective. *Progress in Physical Geography*, *31*(4), 389–403. https://doi.org/10.1177/0309133307081290

Chen, R., Ye, C., Cai, Y., Xing, X., & Chen, Q. (2014). The impact of rural out-migration on land use transition in China: Past, present and trend. *Land Use Policy*, *40*, 101–110. https://doi.org/10.1016/j.landusepol.2013.10.003

Cheng, L., Lu, Q., Wu, B., Yin, C., Bao, Y., & Gong, L. (2016). Estimation of the Costs of Desertification in China: A Critical Review. *Land Degradation & Development*. https://doi.org/10.1002/ldr.2562

Cherubin, M. R., Karlen, D. L., Franco, A. L. C., Cerri, C. E. P., Tormena, C. A., & Cerri, C. C. (2016). A Soil Management Assessment Framework (SMAF) Evaluation of Brazilian Sugarcane Expansion on Soil Quality. *Soil Science Society of America Journal*, *80*(1), 215. https://doi.org/10.2136/sssaj2015.09.0328

Chidumayo, E. N., & Gumbo, D. J. (2013). The environmental impacts of charcoal production in tropical ecosystems of the world: A synthesis. *Energy for Sustainable Development*, *17*(2), 86–94. https://doi.org/10.1016/j.esd.2012.07.004

Chimner, R. A., Cooper, D. J., Wurster, F. C., & Rochefort, L. (2017). An overview of peatland restoration in North America: where are we after 25 years? *Restoration Ecology*, *25*(2), 283–292. https://doi.org/10.1111/rec.12434

Chirwa, P. W., & Mala, W. (2016). Trees in the landscape: towards the promotion and development of traditional and farm forest management in tropical and subtropical regions. *Agroforestry Systems*, *90*(4), 555–561. https://doi.org/10.1007/s10457-016-9987-y

Chodak, M., Gołębiewski, M., Morawska-Płoskonka, J., Kuduk, K., & Niklińska, M. (2015). Soil chemical properties affect the reaction of forest soil bacteria to drought and rewetting stress. *Annals of Microbiology*, *65*(3), 1627–1637. https://doi.org/10.1007/s13213-014-1002-0

CIESIN. (2005). CIAT (Center for International Earth Science Information Network-CIESIN-Columbia University, United Nations Food and Agriculture Programme-FAO, and Centro Internacional de Agricultura Tropical--CIAT): Gridded Population of the World, Version 3 (GPWv3): Po. Palisades: NASA Socioeconomic Data and Applications Center (SEDAC).

Coad, L., Leverington, F., Knights, K., Geldmann, J., Eassom, A., Kapos, V., Kingston, N., de Lima, M., Zamora, C., Burgess, ND., Hockings, M. & Nolte, C. (2015). Measuring impact of protected area management interventions: current and future use of the Global Database of Protected Area Management Effectiveness. Philos Trans. R. Soc. B, *370*(1681). https://doi: 10.1098/rstb.2014.0281.

Cochrane, L. L. (2013). Land Degradation, Faith-Based Organizations, and Sustainability in Senegal. *Culture, Agriculture, Food and Environment*, *35*(2), 112–124. https://doi.org/10.1111/cuag.12015

Coggan, A., Buitelaar, E., Whitten, S., & Bennett, J. (2013). Factors that influence transaction costs in development offsets: Who bears what and why? *Ecological Economics*, *88*, 222–231.

https://doi.org/10.1016/j.ecolecon.2012.12.007

Collas, L., Green, R. E., Ross, A., Wastell, J. H., & Balmford, A. (2017). Urban development, land sharing and land sparing: the importance of considering restoration. *Journal of Applied Ecology*, *54*(6), 1865–1873. https://doi.org/10.1111/1365-2664.12908

Collinge, S. K. (1996). Ecological consequences of habitat fragmentation: Implications for landscape architecture and planning. *Landscape and Urban Planning*, *36*(1), 59–77. https://doi.org/10.1016/s0169-2046(96)00341-6

Conacher, A., & Sala, M. (1998). *Land degradation in Mediterranean environments of the world: nature and extent, causes and solutions*. Wiley.

Connor, J. D., Ward, J. R., & Bryan, B. (2008). Exploring the cost effectiveness of land conservation auctions and payment policies. *Australian Journal of Agricultural and Resource Economics*, *52*(3), 303–319. https://doi.org/10.1111/j.1467-8489.2007.00417.x

Conway, G., & Barbier, E. (2013). After the Green Revolution: Sustainable and equitable agricultural development. *Futures, 20*(6), 651-670. https://doi.org/10.1016/0016-3287(88)90006-7

Corbera, E., & Schroeder, H. (2011). Governing and implementing REDD+. *Environmental Science & Policy*, *14*(2), 89–99. https://doi.org/10.1016/j.envsci.2010.11.002

Corntassel, J., & Bryce, C. (2012). Practicing Sustainable Indigenous Approaches to Cultural Self-Determination. *Brown Journal of World Affairs*, *18*(2), 151–162. Retrieved from http://www.corntassel.net/sustainable_selfdetermination.pdf

Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S. J., Kubiszewski, I., Farber, S., & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global Environmental Change*, *26*(1), 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002

Cottier, T. (2006). From Progressive Liberalization to Progressive Regulation in WTO Law. *Journal of International Economic Law*, *9*(4), 779–821. https://doi.org/10.1093/jiel/jgl029

Couix, N., & Gonzalo-Turpin, H. (2015). Towards a land management approach to ecological restoration to encourage stakeholder participation. *Land Use Policy, 46*, 155-162. https://doi.org/10.1016/j.landusepol.2015.01.025

Cowan, P., Warburton, B., & Fisher, P. (2011). Welfare and ethical issues in invasive species management. *Julius-Kühn-Archiv*, *432*, 44.

Craig, M. D., White, D. A., Stokes, V. L., & Prince, J. (2017). Can postmining revegetation create habitat for a threatened mammal? *Ecological Management & Restoration*, *18*(2), 149–155. https://doi.org/10.1111/emr.12258

Cristescu, R. H., Rhodes, J., Frére, C., & Banks, P. B. (2013). Is restoring flora the same as restoring fauna? Lessons learned from koalas and mining rehabilitation. *Journal of Applied Ecology*, *50*(2), 423–431. https://doi.org/10.1111/1365-2664.12046

Cullen, L., Schmink, M., Padua, C. V, & Morato, M. I. R. (2001). Agroforestry benefit zones: A tool for the conservation and management of Atlantic forest fragments, Sao Paulo, Brazil. *Natural Areas Journal*, *21*(4), 346–356.

Curran, M., Hellweg, S., & Beck, J. (2014). Is there any empirical support for biodiversity offset policy ? *Ecological Applications*, *24*(4), 617–632.

Cuypers, S., & Randier, C. (2009). L'application juridique de la convention sur la protection des Alpes : la situation en Allemagne, Autriche, France, Italie et Slovénie. *Revue Européenne de Droit de l'Environnement*, *13*(1), 3–32. https://doi.org/10.3406/reden.2009.2071

D. Schaible, G., & P. Aillery, M. (2012). *Water Conservation in Irrigated Agriculture: Trends and Challenges in the Face of Emerging Demands*. *Economic Research Service*. Retrieved from http://www.ers.usda.gov/media/884158/eib99.pdf

Dai, L., Wang, Y., Su, D., Zhou, L., Yu, D., Lewis, B. J., & Qi, L. (2011). Major Forest Types and the Evolution of Sustainable Forestry in China. *Environmental Management*, *48*(6), 1066–1078. https://doi.org/10.1007/s00267-011-9706-4

Daily, G. C., Kareiva, P. M., Polasky, S., Ricketts, T. H., & Tallis, H. (2011). Mainstreaming natural capital into decisions. In S. P. Kareiva, H. Tallis, T.H. Ricketts, G.C. Daily Polasky (Ed.), *Natural capital: theory and practice of mapping ecosystem services.* (pp. 3–14). Oxford: Oxford University Press.

Dal Ferro, N., Cocco, E., Lazzaro, B., Berti, A., & Morari, F. (2016). Assessing the role of agri-environmental measures to enhance the environment in the Veneto Region, Italy, with a model-based approach. *Agriculture, Ecosystems & Environment*, *232*, 312–325. https://doi.org/10.1016/j.agee.2016.08.010

Daley, D. M., & Layton, D. F. (2004). Policy Implementation and the Environmental Protection Agency: What Factors Influence Remediation at Superfund Sites? *Policy Studies Journal*, *32*(3), 375–392. https://doi.org/10.1111/j.1541-0072.2004.00071.x

Davidson, N. C. (2014). How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research*, *65*(10), 934–941. https://doi.org/10.1071/MF14173

DeFries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*, *3*(3), 178–181. Retrieved from http://dx.doi.org/10.1038/ngeo756

Delfosse, E. S. (2005). Risk and ethics in biological control. *Biological Control, 35*(3), 319-329.

Deng, L., Shangguan, Z., & Sweeney, S. (2014). “Grain for Green” driven land use change and carbon sequestration on the Loess Plateau, China. *Scientific Reports*, *4*, 7039. https://doi.org/10.1038/srep07039

Denkstatt. (2016). *Natural Capital Accounting: The Coca-Cola Water Replenishment Programme.* Vienna. Retrieved from http://naturalcapitalcoalition.org/wp-content/uploads/2016/07/Denkstatt_Natural_Capital_Accounting.pdf

DFRS. (2015). *State of Nepal’s Forests. Forest Resource Assessment (FRA)*. Kathmandu, Nepal.

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. R., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K. M., Figueroa, V. E., Duraiappah, A., Fischer, M., Hill, R., Koetz, T., Leadley, P., Lyver, P., Mace, G. M., Martin-Lopez, B., Okumura, M., Pacheco, D., Pascual, U., Pérez, E. S., Reyers, B., Roth, E., Saito, O., Scholes, R., Sharma, N., Tallis, H., Thaman, R., Watson, R., Yahara, T., Hamid, Z. A., Akosim, C., Al-Hafedh, Y., Allahverdiyev, R., Amankwah, E., Asah, T. S., Asfaw, Z., Bartus, G., Brooks, A. L., Caillaux, J., Dalle, G., Darnaedi, D., Driver, A., Erpul, G., Escobar-Eyzaguirre, P., Failler, P., Fouda, A. M. M., Fu, B., Gundimeda, H., Hashimoto, S., Homer, F., Lavorel, S., Lichtenstein, G., Mala, W. A., Mandivenyi, W., Matczak, P., Mbizvo, C., Mehrdadi, M., Metzger, J. P., Mikissa, J. B., Moller, H., Mooney, H. A., Mumby, P., Nagendra, H., Nesshover, C., Oteng-Yeboah, A. A., Pataki, G., Roué, M., Rubis, J., Schultz, M., Smith, P., Sumaila, R., Takeuchi, K., Thomas, S., Verma, M., Yeo-Chang, Y., & Zlatanova, D. (2015). The IPBES Conceptual Framework — connecting nature and people. *Current Opinion in Environmental Sustainability*. https://doi.org/10.1016/j.cosust.2014.11.002

Dimobe, K., Ouédraogo, A., Soma, S., Goetze, D., Porembski, S., & Thiombiano, A. (2015). Identification of driving factors of land degradation and deforestation in the Wildlife Reserve of Bontioli (Burkina Faso, West Africa). *Global Ecology and Conservation*, *4*, 559–571. https://doi.org/10.1016/j.gecco.2015.10.006

Dissart, J.-C. (2006). Protection des espaces agricoles et naturels: une analyse des outils américains et français. *Économie Rurale*, *291*, 6–25.

DiTomaso, J. M. (2000). Invasive weeds in rangelands: Species, impacts, and management. *Weed Science*, *48*(2), 255–265. https://doi.org/10.1614/0043-1745(2000)048[0255:IWIRSI]2.0.CO;2

DiTomaso, J. M., Brookes, M. L., Allen, E. B., Minnich, R., Rice, P. M., & Kyser, G. B. (2006). Control of Invasive Weeds with Prescribed Burning 1. *Weed Technology*, *20*(2), 535–548. https://doi.org/10.1614/WT-05-086R1.1

DoF. (2015). *Community forestry national database: MIS database*. Kathmandu, Nepal: Department of Forests (DoF), Community Forest Division.

DOI. (2012). The Department of the Interior's Economic Contributions, FY 2011-2012. Department of Interior, USA. Retrieved from http://www.doi.gov/americasgreatoutdoors/loader.cfm?csModule = security/getfile&pageid=308931

Dooley, E., Roberts, E., & Wunder, S. (2015). Land degradation neutrality unders the SDGs : National and international implementation of the land degradation neutral world target. *Environemntal Law Network International*, *1/2*, 2–9. Retrieved from http://www.elni.org/fileadmin/elni/dokumente/Archiv/2015/elni_2015_1_2_land_degradation_neutrality.pdf

Duff, S. N., Stonehouse, Hilts, D., & Blackburn, S. (1991). Soil conservation behavior and attitudes among Ontario farmers toward alternative government policy responses. *Journal of Soil and Water Conservation*, *46*(3), 2150219.

Duggin, J. ., & Gentle, C. . (1998). Experimental evidence on the importance of disturbance intensity for invasion of Lantana camara L. in dry rainforest–open forest ecotones in north-eastern NSW, Australia. *Forest Ecology and Management*, *109*(1), 279–292. https://doi.org/10.1016/S0378-1127(98)00252-7

Dumanski, J. (2015). Evolving concepts and opportunities in soil conservation. *International Soil and Water Conservation Research*, *3*(1), 1–14. https://doi.org/10.1016/j.iswcr.2015.04.002

Durigan, G., Guerin, N., & da Costa, J. N. (2013). Ecological restoration of Xingu Basin headwaters: motivations, engagement, challenges and perspectives. *Philosophical Transactions of the Royal Society B-Biological Sciences*, *368*(1619). https://doi.org/10.1098/rstb.2012.0165

Eckholm, E. (1976). Losing Ground. *Environment: Science and Policy for Sustainable Development*, *18*(3), 6–11. https://doi.org/10.1080/00139157.1976.9930747

ECSA, & WFP. (2014). *Comprehensive Food Security and Vulnerability Analysis (CFSVA)*. Retrieved from http://documents.wfp.org/stellent/groups/public/documents/ena/wfp265490.pdf?_ga=2.26177213.1027608398.1518102244-1864233058.1518102244

Edens, B., & Graveland, C. (2014). Experimental valuation of Dutch water resources according to SNA and SEEA. *Water Resources and Economics*, *7*, 66–81. https://doi.org/10.1016/J.WRE.2014.10.003

Edwards, P. E. T., Sutton-Grier, A. E., & Coyle, G. E. (2013). Investing in nature: Restoring coastal habitat blue infrastructure and green job creation. *Marine Policy*, *38*, 65–71. https://doi.org/10.1016/j.marpol.2012.05.020

EEA. (2016). *Environmental Indicator Report 2016*. https://doi.org/10.2800/4874

EEMP. (2013). *Loess Plateau Watershed Rehabilitation Project.* Environmental Education Media Project. Retrieved from http://eempc.org/loess-plateau-watershed-rehabilitation-project/

Egan, J., & Mortensen, D. (2012). A comparison of land-sharing and land-sparing strategies for plant richness conservation in agricultural landscapes. *Ecol Appl.*, *22*(2), 459–471.

Eigenraam, M., Strappazzon, L., Lansdell, N., Beverly, C., & Stoneham, G. (2007). Designing frameworks to deliver unknown information to support market-based instruments. *Agricultural Economics*, *37*(s1), 261–269. https://doi.org/10.1111/j.1574-0862.2007.00250.x

ELD Initiative. (2015). *Report for policy and decision makers: Reaping economic and environmental benefits from sustainable land management.* The Economics of Land Degradation. Retrieved from http://www.eld-initiative.org/fileadmin/pdf/ELD-pm-report_08_web_72dpi.pdf

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, *65*(4), 663–674. https://doi.org/10.1016/j.ecolecon.2008.03.011

Ens, E., Fisher, J., & Costello, O. (2015). *Indigenous people and invasive species: Perceptions, challenges, management and uses*. IUCN Commission on Ecosystem Management Community Booklet. Retrieved from https://www.iucn.org/backup_iucn/cmsdata.iucn.org/downloads/ipis_booklet_v6_low_rez_final.pdf

Entec. (2009). *Integrated pollution prevention & control (IPPC)*.

ESPON (2012). *EU-LUPA: European Land Use Patterns. Volume XIII: Policy options and recommendations*. Part C Scientific Report, v30. Retrieved from https://www.espon.eu/sites/default/files/attachments/Volume_XIII_Policies_301112.pdf

EC. (2012). *Guidelines on best practice to limit, mitigate or compensate soil sealing*. Luxembourg: Publications Office of the European Union. https://doi.org/10.2779/75498

EEA. (2011). Urban soil sealing in Europe. European Environment Agency. Retrieved April 4, 2017, from http://www.eea.europa.eu/articles/urban-soil-sealing-in-europe

EU. (2013). *Mapping and Assessment of Ecosystems and their Services*. *Agriculture, Ecosystems & Environment*. European Union. https://doi.org/10.2779/12398

EU. (2014). *Study on ELD effectiveness, scope and exceptions reference*. European Union. https://doi.org/10.2779/70236

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics*, *34*, 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

FAIS. (2012). *World Food Aid Flows*. UN World Food Program. https://doi.org/http://www.wfp.org/fais/

Fan, M., Li, Y., & Li, W. (2015). Solving one problem by creating a bigger one: The consequences of ecological resettlement for grassland restoration and poverty alleviation in Northwestern China. *Land Use Policy*, *42*, 124–130. https://doi.org/10.1016/j.landusepol.2014.07.011

FAO. (2011). *Community-based fire management* (FAO Forestry Paper No. 166). Rome: Food and Agriculture Organization of the United Nations. Retrieved from http://www.fao.org/docrep/015/i2495e/i2495e00.htm

FAO (2012). *Voluntary Guidelines on the Governance of Tenure: At a glance*. Rome: Food and Agriculture Organization of the United Nations. Retrieved from http://www.fao.org/docrep/016/i3016e/i3016e.pdf

FAO. (2014). *World reference base for soil resources 2014. International soil classification system for naming soils and creating legends for soil maps*. World Soil Resources Reports No. 106. Rome: Food and Agriculture Organization of the United Nation. https://doi.org/10.1017/S0014479706394902

FAO. (2015). *Conservation Agriculture Adoption Worldwide*. Retrieved April 10, 2017, from http://www.fao.org/ag/ca/6c.html

FAO. (2017). *Landscapes for life: Approaches to landscape management for sustainable food and* *agriculture*. Rome: Food and Agriculture Organization of the United Nations. Retrieved from http://www.fao.org/3/i8324en/i8324en.pdf

FAO & ITPS. (2015). *Status of the World's Soil Resources (SWSR) – Main Report*. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils, Rome, Italy.

FAO. (2016). AQUASTAT Main Database. Retrieved April 10, 2017, from http://www.fao.org/nr/water/aquastat/data/query/index.html

FAO, & UNEP. (1995). *Our Land Our Future*. Food and Agricultural Organization of the United Nations & the United Nations Environment Programme.

Fargione, J., Hill, J., Tilman, D., Polasky, S., & Hawthorne, P. (2008). Land clearing and the biofuel carbon debt. *Science*, *319*(5867), 1235–1238. https://doi.org/10.1126/science.1152747

Farooq, M., & Siddique, K. H. M. (2015). Conservation Agriculture: Concepts, Brief History, and Impacts on Agricultural Systems. In *Conservation Agriculture* (p. 3–17.). Springer.

Ferris, J., & Siikamaki, J. (2010). *Conservation Reserve Program and Wetland Reserve Program*. *Resources for the Future*. OECD. Retrieved from http://www.rff.org/files/sharepoint/WorkImages/Download/RFF-BCK-ORRG_CRP_and_WRP.pdf

Fine, P. V. A. (2002). The invasibility of tropical forests by exotic plants. *Journal of Tropical Ecology*, *18*(5), 687–705. https://doi.org/10.1017/S0266467402002456

Finlayson, C. M. (2012). Forty years of wetland conservation and wise use. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *22*(2), 139–143. https://doi.org/10.1002/aqc.2233

Finlayson, C. M., Davidson, N., Pritchard, D., Milton, G. R., & MacKay, H. (2011). The Ramsar Convention and Ecosystem-Based Approaches to the Wise Use and Sustainable Development of Wetlands The Ramsar Convention and Ecosystem-Based Approaches to the Wise Use and Sustainable Development of Wetlands. *Journal of International Wildlife Law & Policy*, *14*(3–4), 176–198. https://doi.org/10.1080/13880292.2011.626704

Fisher, J. (2013). Decline of availability of traditional food sources in indigenous Australia. In S. Das Gupta (Ed.), *Food & environmental security--imperatives of indigenous knowledge systems* (pp. 383–389). Jodhpur, Rajasthan: Agrobios (India).

Fisher, J. L. (2011). *Weed mapping and management of the City of Stirling Coastal Zone Perth, Stirling Natural Environment Coastcare (SNEC)*. Perth.

Fisher, M., Chaudhury, M., & McCusker, B. (2010). Do forests help rural households adapt to climate variability? Evidence from Southern Malawi. *World Development, 38*(9), 1241-1250. https://doi.org/10.1016/j.worlddev.2010.03.005

Fisher J. L. (2015). *Ecosystem Approach to Invasive Alien Species Management Sri Lanka*. Colombo.

Fisher J. L. (2016). *Restoration of the City of Stirling Coastal Zone incorporating ecological and economic effectiveness of Restoration, prepared for City of Stirling*. Perth.

Fisheries & Forestry. (2013). *Grazing BMP self-assessment Grazing land management Northern Australian module*. The State of Queensland.

Flores-Díaz, A. C., Castillo, A., Sánchez-Matías, M., & Maass, M. (2014). Local values and decisions: views and constraints for riparian management in western Mexico. *Knowledge and Management of Aquatic Ecosystems*, *414*. https://doi.org/10.1051/kmae/2014017

Forest People Program & Program. (2010). *Customary sustainable use of biodiversity by indigenous peoples and local communities: Examples, challenges, community initiatives and recommendations relating to CBD Article 10(c)*. Retrieved from http://www.forestpeoples.org/customary-sustainable-use-studies

France, R. L. (2016). From land to sea: Governance-management lessons from terrestrial restoration research useful for developing and expanding social-ecological marine restoration. *Ocean and Coastal Management*, *133*, 64–71. https://doi.org/10.1016/j.ocecoaman.2016.08.022

Franzluebbers, A. J., Sawchik, J., & Taboada, M. A. (2014). Agronomic and environmental impacts of pasture-crop rotations in temperate North and South America. *Agriculture, Ecosystems and Environment*, *190*, 18–26. https://doi.org/10.1016/j.agee.2013.09.017

Frey, B. S., & Jegen, R. (2001). Motivation crowding theory a survey of empirical evidence. *Journal of Economic Surveys*, *15*(5), 589–611. https://doi.org/10.3929/ETHZ-B-000225616

Friedrich, T., Derpsch, R., & Kassam, A. (2012). Overview of the Global Spread of Conservation Agriculture. *FACTS Reports*, *Special Issue 6*.

Frouz, J., Dvorščík, P., Vávrová, A., Doušová, O., Kadochová, Š., & Matějíček, L. (2015). Development of canopy cover and woody vegetation biomass on reclaimed and unreclaimed post-mining sites. *Ecological Engineering*, *84*, 233–239. https://doi.org/10.1016/j.ecoleng.2015.09.027

Frouz, J., Livečková, M., Albrechtová, J., Chroňáková, A., Cajthaml, T., Pižl, V., Háněl, L., Starý, J., Baldrian, P., Lhotáková, Z., Šimáčková, H., & Cepáková, Š. (2013). Is the effect of trees on soil properties mediated by soil fauna? A case study from post-mining sites. *Forest Ecology and Management*, *309*, 87–95. https://doi.org/10.1016/j.foreco.2013.02.013

Fu, B. (1989). Soil erosion and its control in the loess plateau of China. *Soil Use and Management*, *5*(2), 76–82. https://doi.org/10.1111/j.1475-2743.1989.tb00765.x

Funk, C., Sanay, G., Asfaw, A., Korecha, D., Choularton, R., Verdin, J., Eilerts, G., & Michaelsen, J. (2005). *Recent Drought Tendencies in Ethiopia and Equatorial-Subtropical Eastern Africa*. Washington DC: Famine Early Warning System Network, USAID.

Gaertner, M., Fisher, J., Sharma, G., & Esler, K. (2012). Insights into invasion and restoration ecology: Time to collaborate towards a holistic approach to tackle biological invasions. *NeoBiota*, *12*, 57–76. https://doi.org/10.3897/neobiota.12.2123

Gaertner, M., Wilson, J. R. U., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., & Richardson, D. M. (2017). Non-native species in urban environments: patterns, processes, impacts and challenges. *Biological Invasions*, *19*(12), 3461–3469. https://doi.org/10.1007/s10530-017-1598-7

Gao, Z., Sun, B., del Barrio, G., Li, X., Wang, H., Bai, L., Wang, B., Zhang, W., & Ieee. (2014). Land degradation assessment by applying relative rue in Inner Mongolia, China, 2001-2010. *2014 IEEE International Geoscience and Remote Sensing Symposium (IGARSS)*, 1449–1452.

Gardner, J. H., & Bell, D. T. (2007). Bauxite mining restoration by Alcoa World Alumina Australia in Western Australia: Social, political, historical, and environmental contexts. *Restoration Ecology*, *15*(SUPPL. 4), 3–10. https://doi.org/10.1111/j.1526-100X.2007.00287.x

Gardner, R.C., Barchiesi, S., Beltrame, C., Finlayson, C.M., Galewski, T., Harrison, I., Paganini, M., Perennou, C., Pritchard, D.E., Rosenqvist, A., and Walpole, M. 2015. (2015). State of the World's Wetlands and their Services to People: A compilation of recent analyses. *Ramsar Briefing Note*, *no. 7*(June), 1–20. https://doi.org/10.2139/ssrn.2589447

Garrett, R. D., Lambin, E. F., & Naylor, R. L. (2013). The new economic geography of land use change: Supply chain configurations and land use in the Brazilian Amazon. *Land Use Policy*, *34*, 265–275. https://doi.org/10.1016/j.landusepol.2013.03.011

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliess bach, A., Buchmann, N., Mader, P., Stoize, M., Smith, P., Scialabba, N. E., & Niggli, U. (2012). Enhanced top soil carbon stocks under organic farming. *Proceedings of the National Academy of Sciences*, *109*, 18226–18231.

Gebreselassie, S., Kirui, O. K., & Mirzabaev, A. (2016). Economics of Land Degradation and Improvement in Ethiopia. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 400–430). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_14

Geist, C., & Galatowitsch, S. M. (1999). Reciprocal Model for Meeting Ecological and Human Needs in Restoration Projects. *Conservation Biology*, *13*(5), 970–979. https://doi.org/10.1046/j.1523-1739.1999.98074.x

Geist, H. J., & Lambin, E. F. (2002). Proximate Causes and Underlying Driving Forces of Tropical Deforestation. *BioScience*, *52*(2), 143. https://doi.org/10.1641/0006-3568(2002)052[0143:PCAUDF]2.0.CO;2

Geldermann, J., & Rentz, O. (2004). The reference installation approach for the techno-economic assessment of emission abatement options and the determination of BAT according to the IPPC-directive. *Journal of Cleaner Production*, *12*(4), 389–402. https://doi.org/10.1016/S0959-6526(03)00032-5

Gennaio, M.-P., Hersperger, A. M., & Bürgi, M. (2009). Containing urban sprawl—Evaluating effectiveness of urban growth boundaries set by the Swiss Land Use Plan. *Land Use Policy*, *26*(2), 224–232. https://doi.org/10.1016/j.landusepol.2008.02.010

Genovesi, P. (2005). Eradications of invasive alien species in Europe: a review. In *Issues in Bioinvasion Science* (pp. 127–133). Berlin/Heidelberg: Springer-Verlag. https://doi.org/10.1007/1-4020-3870-4_12

Ghosheh, H. Z. (2005, September). Constraints in implementing biological weed control: A review. *Weed Biology and Management, 5*(3), 83-92. https://doi.org/10.1111/j.1445-6664.2005.00163.x

Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., & Walker, N. F. (2015). Brazil's Soy Moratorium. *Science*, *347*(6220), 377–378. https://doi.org/10.1126/science.aaa0181

Gibson, C. A., & Marks, S. A. (1995). Transforming Rural Hunters into Conservationsits: An Assessment of Community-Based Wildlife Management Programmes in Africa. *World Development*, *23*(6), 941–957.

Giesy, J. P., Dobson, S., & Solomon, K. R. (2000). Ecotoxicological Risk Assessment for Roundup® Herbicide. In G. W. Ware (Ed.) *Reviews of Environmental Contamination and Toxicology*, *167.* New York: Springer. https://doi.org/10.1007/978-1-4612-1156-3_2

Gill, S. E., Handley, J. F., Ennos, A. R., & Pauleit, S. (2007). Adapting cities for climate change: the role of the green infrastructure. *Built Environment*, *33*(1), 115–133. https://doi.org/10.2148/benv.33.1.115

Glen, A. S., Atkinson, R., Campbell, K. J., Hagen, E., Holmes, N. D., Keitt, B. S., Parkes, J. P., Saunders, A., Sawyer, J., & Torres, H. (2013). Eradicating multiple invasive species on inhabited islands: the next big step in island restoration? *Biological Invasions*, *15*(12), 2589–2603. https://doi.org/10.1007/s10530-013-0495-y

GLF. (2014). *Global Landscapes Forum*. Retrieved January 31, 2017, from http://www.landscapes.org/glf-2014/

Gliessman, S. R. (2014). *Agroecology: the ecology of sustainable food systems*. CRC Press.

Godfrey, S., Labhasetwar, P., Wate, S., & Jimenez, B. (2010). Safe greywater reuse to augment water supply and provide sanitation in semi-arid areas of rural India. *Water Science and Technology*, *62*(6). Retrieved from http://wst.iwaponline.com/content/62/6/1296

GOE. (2015). *Joint Government and Humanitarian Partners' Document* . Addis Ababa, Ethiopia: Government of Ethiopia and Partners.

Goesch, T. (2001). Delivery Charges for Water: Their Impact on Interregional Trade in Water Rights. *Australian Commodities: Forecasts and Issues*, *8*(4), 626–634.

Goldstein, J. H., Caldarone, G., Duarte, T. K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S., & Daily, G. C. (2012). Integrating ecosystem-service tradeoffs into land-use decisions. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(19), 7565–7570. https://doi.org/10.1073/pnas.1201040109

Gómez-Aíza, L., Martínez-Ballesté, A., Álvarez-Balderas, L., Lombardero-Goldaracena, A., García-Meneses, P. M., Caso-Chávez, M., & Conde-Álvarez, C. (2017). Can wildlife management units reduce land use/land cover change and climate change vulnerability? Conditions to encourage this capacity in Mexican municipalities. *Land Use Policy*, *64*, 317–326. https://doi.org/10.1016/j.landusepol.2017.03.004

Graham, L. L. B., Giesen, W., & Page, S. E. (2017). A common-sense approach to tropical peat swamp forest restoration in Southeast Asia. *Restoration Ecology*, *25*(2), 312–321. https://doi.org/10.1111/rec.12465

Grant, C., & Koch, J. (2007). Decommissioning Western Australia's First Bauxite Mine: Co-evolving vegetation restoration techniques and targets. *Ecological Management & Restoration*, *8*(2), 92–105. https://doi.org/10.1111/j.1442-8903.2007.00346.x

Grau, H. R., Aide, T. M., Zimmerman, J. K., Thomlinson, J. R., Helmer, E., & XioMing, Z. (2003). The ecological consequences of socioeconomic and land-use changes in postagricultural Puerto Rico. *BioScience*, *53*(12), 1159–1168.

Graves, A. R., Morris, J., Deeks, L. K., Rickson, R. J., Kibblewhite, M. G., Harris, J. A., Farewell, T. S., & Truckle, I. (2015). The Total Costs of Soil Degradation in England and Wales. *Ecological Economics*, *119*, 399–413.

Groot, R. De, Fisher, B., Christie, M., Aronson, J., Braat, L., Gowdy, J., Haines-young, R., Maltby, E., Neuville, A., Polasky, S., Portela, R., Ring, I., Blignaut, J., Brondízio, E., Costanza, R., Jax, K., Kadekodi, G. K., May, P. H., Mcneely, J., & Shmelev, S. (2010). Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In P. Kumar (Ed.) *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.* London: Routledge. https://doi.org/10.4324/9781849775489

Grosjean, P., & Kontoleon, A. (2009). How sustainable are sustainable development programs? The case of the Sloping Land Conversion Program in China. *World Development*, *37*(1), 268–285.

Guerry, A., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G. C., Griffin, R., & Ruckelshaus, M. (2015). Natural capital and ecosystem services informing decisions: From promise to practice. *PNAS*, *112*(24), 7348–7355.

Günal, H., Korucu, T., Birkas, M., Özgöz, E., & Halbac-Cotoara-Zamfir, R. (2015). Threats to Sustainability of Soil Functions in Central and Southeast Europe. *Sustainability*, *7*(2), 2161–2188. https://doi.org/10.3390/su7022161

Guo, M., Song, W., & Buhain, J. (2015). Bioenergy and biofuels: History, status, and perspective. *Renewable and Sustainable Energy Reviews*, *42*, 712–725. https://doi.org/10.1016/j.rser.2014.10.013

Hagen, K., & Stiles, R. (2010). *Urban Biodiversity and Design*. Wiley Online Library.

Haglund, E., Ndjeunga, J., Snook, L., & Pasternak, D. (2011). Dry land tree management for improved household livelihoods: Farmer managed natural regeneration in Niger. *Journal of Environmental Management*, *92*(7), 1696–1705. https://doi.org/10.1016/j.jenvman.2011.01.027

Hagy, J. D., Boynton, W. R., Keefe, C. W., & Wood, K. V. (2004). Hypoxia in Chesapeake Bay, 1950–2001:

Long-term change in relation to nutrient loading and river flow. *Estuaries*, *27*(4), 634–658. https://doi.org/10.1007/BF02907650

Hahn, T., McDermott, C., Ituarte-Lima, C., Schultz, M., Green, T., & Tuvendal, M. (2015). Purposes and degrees of commodification: Economic instruments for biodiversity and ecosystem services need not rely on markets or monetary valuation. *Ecosystem Services*, *16*, 74–82. https://doi.org/10.1016/j.ecoser.2015.10.012

Hajkowicz, S., Higgins, A., Williams, K., Faith, D. P., & Burton, M. (2007). Optimisation and the selection of conservation contracts. *The Australian Journal of Agricultural and Resource Economics*, *51*(1), 39–56. https://doi.org/10.1111/j.1467-8489.2007.00345.x

Hall, J., Arheimer, B., Borga, M., Brázdil, R., Claps, P., Kiss, A., Kjeldsen, T. R., Kriaučiūnienė, J., Kundzewicz, Z. W., Lang, M., Llasat, M. C., Macdonald, N., McIntyre, N., Mediero, L., Merz, B., Merz, R., Molnar, P., Montanari, A., Neuhold, C., Parajka, J., Perdigão, R. A. P., Plavcová, L., Rogger, M., Salinas, J. L., Sauquet, E., Schär, C., Szolgay, J., Viglione, A., & Blöschl, G. (2014). Understanding flood regime changes in Europe: a state-of-the-art assessment. *Hydrology and Earth System Sciences*, *18*(7), 2735–2772. https://doi.org/10.5194/hess-18-2735-2014

Hammac, W. A., Stott, D. E., Karlen, D. L., & Cambardella, C. A. (2016). Crop, Tillage, and Landscape Effects on Near-Surface Soil Quality Indices in Indiana. *Soil Science Society of America Journal*, *80*, 1638–1652.

Hanley, N., Banerjee, S., Lennox, G. D., & Armsworth, P. R. (2012). How should we incentivize private landowners to “produce” more biodiversity? *Oxford Review of Economic Policy*, *28*(1), 93–113. https://doi.org/10.1093/oxrep/grs002

Haregeweyn, N., Berhe, A., Tsunekawa, A., Tsubo, M., & Meshesha, D. T. (2012). Integrated Watershed Management as an Effective Approach to Curb Land Degradation: A Case Study of the Enabered Watershed in Northern Ethiopia. *Environmental Management*, *50*(6), 1219–1233. https://doi.org/10.1007/s00267-012-9952-0

Harris, J.A., Birch, P., Palmer, J. P. (1996). *Land restoration and reclamation: principles and practice*. Addison Wesley Longman Ltd.

Harris, J. (2009). Soil microbial communities and restoration ecology: facilitators or followers? *Science*, *325*(5940), 573–574.

Harris, J. A. (2003). Measurements of the soil microbial community for estimating the success of restoration. *European Journal of Soil Science*, *54*(4), 801–808. https://doi.org/10.1046/j.1365-2389.2003.00559.x

Harris, J. A., & Birch, P. (1989). Soil microbial activity in opencast coal mine restorations. *Soil Use and Management*, *5*(4), 1006–1009. https://doi.org/10.1111/j.1475-2743.1989.tb00778.x

Harris, J. A., Birch, P., & Short, K. C. (1989). Changes in the microbial community and physico-chemical characteristics of topsoils stockpiled during opencast mining. *Soil Use and Management*, *5*(4), 161–168. https://doi.org/10.1111/j.1475-2743.1989.tb00778.x

Harris, J. *et al.* (2006). Ecological restoration and global climate change. *Restoration Ecology*, *14*(2), 170–176.

Hartmann, W. E., Wendt, D. C., Saftner, M. A., Marcus, J., & Momper, S. L. (2014). Advancing Community-Based Research with Urban American Indian Populations: Multidisciplinary Perspectives. *American Journal of Community Psychology*, *54*(1–2), 72–80. https://doi.org/10.1007/s10464-014-9643-5

Hasenheit, M., Gerdes, H., Kiresiewa, Z., & Beekman, V. (2016). *Summary report on the social, economic and environmental impacts of the bioeconomy*. BioSTEP. Retrieved from https://www.ecologic.eu/sites/files/publication/2016/2801-social-economic-environmental-impacts-bioeconomy-del2-2.pdf

Hassan, A. M., & Lee, H. (2015). Toward the sustainable development of urban areas: An overview of global trends in trials and policies. *Land Use Policy*, *48*, 199–212. https://doi.org/10.1016/j.landusepol.2015.04.029

Hayes, T., & Persha, L. (2010). Nesting local forestry initiatives: Revisiting community forest management in a REDD+ world. *Forest Policy and Economics, 12*(8). 545-553. https://doi.org/10.1016/j.forpol.2010.07.003

He, J. (2014). Governing forest restoration: Local case studies of sloping land conversion program in Southwest China. *Forest Policy and Economics, 46*, 30-38. https://doi.org/10.1016/j.forpol.2014.05.004

Heap, I. (2014). Herbicide Resistant Weeds. In D. Pimentel & R. Peshin (Eds.) *Integrated Pest Management* (pp. 281–301). Dordrecht: Springer Netherlands. https://doi.org/10.1007/978-94-007-7796-5_12

Hein, L., Bagstad, K., Edens, B., Obst, C., de Jong, R., & Lesschen, J. P. (2016). Defining Ecosystem Assets for Natural Capital Accounting. *PLOS ONE*, *11*(11), e0164460. https://doi.org/10.1371/journal.pone.0164460

Hessel, R., Reed, M. S., Geeson, N., Ritsema, C. J., van Lynden, G., Karavitis, C. A., Schwilch, G., Jetten, V., Burger, P., van der Werff ten Bosch, M. J., Verzandvoort, S., van den Elsen, E., & Witsenburg, K. (2014). From Framework to Action: The DESIRE Approach to Combat Desertification. *Environmental Management*, *54*(5), 935–950. https://doi.org/10.1007/s00267-014-0346-3

Hiers, J. K., Jackson, S. T., Hobbs, R. J., Bernhardt, E. S., & Valentine, L. E. (2016). The Precision Problem in Conservation and Restoration. *Trends in Ecology and Evolution*, *31*(11), 820–830. https://doi.org/10.1016/j.tree.2016.08.001

Hill, R., Pert, P. L., Davies, J., Robinson, C. J., Walsh, F., & Falco-Mammone, F. (2013). Indigenous Land Management in Australia: Extent, Scope, Diversity, Barriers and Success Factors. Cairns: CSIRO Ecosystem Sciences. Retrieved from http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.645.4100&rep=rep1&type=pdf

HMG/ADB/FINNIDA. (1988). *Master plan for the forestry sector, Nepal : main report*. Kathmandu, Nepal.

HMG/N. (1993). Forest Act 1993 (in Nepali). Kathmandu, Nepal: His Majesty's Government of Nepal (HMG/N), Ministry of Forests and Soil Conservation.

Hobbs, R. J., Hallett, L. M., Ehrlich, P. R., & Mooney, H. A. (2011). Intervention Ecology: Applying Ecological Science in the Twenty-first Century. *BioScience*, *61*(6), 442–450. https://doi.org/10.1525/bio.2011.61.6.6

Hobbs, R. J., & Harris, J. a. (2001). Restoration Ecology : Repairing the Earth ' s Ecosystems in the New Millennium. *Restoration Ecology*, *9*(2), 239–246. https://doi.org/10.1046/j.1526-100x.2001.009002239.x

Hoeksema, J. D., Chaudhary, V. B., Gehring, C. A., Johnson, N. C., Karst, J., Koide, R. T., Pringle, A., Zabinski, C., Bever, J. D., Moore, J. C., Wilson, G. W. T., Klironomos, J. N., & Umbanhowar, J. (2010). A meta-analysis of context-dependency in plant response to inoculation with mycorrhizal fungi. *Ecology Letters*, *13*(3), 394–407. https://doi.org/10.1111/j.1461-0248.2009.01430.x

Hogarth, P. J. (2007). *The biology of mangroves and seagrasses*. (Oxford University Press, Ed.). Retrieved from https://global.oup.com/academic/product/the-biology-of-mangroves-and-seagrasses-9780198716556?cc=us&lang=en&

Holl, K. D. (2017). Restoring tropical forests from the bottom up. *Science*, *355*(6324), 455–456. https://doi.org/10.1126/science.aam5432

Hooijer, A., Page, S., Canadell, J. G., Silvius, M., Kwadijk, J., Wosten, H., & Jauhiainen, J. (2010). Current and future CO2 emissions from drained peatlands in Southeast Asia. *Biogeosciences*, *7*, 1505–1514.

Horwitz, P., & Finlayson, C. M. (2011). Wetlands as Settings for Human Health: Incorporating Ecosystem Services and Health Impact Assessment into Water Resource Management. *BioScience*, *61*(9), 678–688. https://doi.org/10.1525/bio.2011.61.9.6

Hostetler, M., Allen, W., & Meurk, C. (2011). Conserving urban biodiversity? Creating green infrastructure is only the first step. *Landscape and Urban Planning*, *100*(4), 369–371. https://doi.org/10.1016/j.landurbplan.2011.01.011

Hough, P., & Robertson, M. (2009). Mitigation under Section 404 of the Clean Water Act: Where it comes from, what it means. *Wetlands Ecology and Management*, *17*(1), 15–33. https://doi.org/10.1007/s11273-008-9093-7

Howell, D. M., Gupta, S. Das, Pinno, B. D., & M.D., M. (2016). Reclaimed soils, fertilizer, and bioavailable nutrients: Determining similarity with natural benchmarks over time. *Canadian Journal of Soil Science*, *97*(2), 149–158.

Howieson, J., Calmy, H., Ballard, N., Skinner, P., WO’Hara, G., Skinner, L., Ruthrof, K. X., Swift, R., Ballard, V., St Hardy, G. E., & McHenry, M. P. (2017). Bread from stones: Post-mining land use change from phosphate mining to farmland. *The Extractive Industries and Society*, *4*(2), 290–299. https://doi.org/10.1016/J.EXIS.2016.11.001

Hu, S., Niu, Z., Chen, Y., Li, L., & Zhang, H. (2017). Global wetlands: Potential distribution, wetland loss, and status. *Science of the Total Environment*, *586*, 319–327. https://doi.org/10.1016/j.scitotenv.2017.02.001

Hulme, P. E. (2006). Beyond control: wider implications for the management of biological invasions. *Journal of Applied Ecology*, *43*(5), 835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

Hulme, P. E. (2009). Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, *46*(1), 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Huot, H., Séré, G., Charbonnier, P., Simonnot, M.-O., & Morel, J. L. (2015). Lysimeter monitoring as assessment of the potential for revegetation to manage former iron industry settling ponds. *Science of The Total Environment*, *526*, 29–40. https://doi.org/10.1016/j.scitotenv.2015.04.025

Hurtt, G. C., Frolking, S., Fearon, M. G., Moore, B., Shevliakova, E., Malyshev, S., Pacala, S. W., & Houghton, R. A. (2006). The underpinnings of land-use history: Three centuries of global gridded land-use transitions, wood-harvest activity, and resulting secondary lands. *Global Change Biology*, *12*(7), 1208–1229. https://doi.org/10.1111/j.1365-2486.2006.01150.x

Huttunen, S., & Peltomaa, J. (2016). Agri-environmental policies and “good farming” in cultivation practices at Finnish farms. *Journal of Rural Studies*, *44*, 217–226. https://doi.org/10.1016/j.jrurstud.2016.02.004

Industrial Economics Inc. (2012). *The Economic Impacts of Ecological Restoration in Massachusetts*. Prepared for the Massachusetts Department of Fish and Game, Division of Ecological Restoration. Retrieved from http://www.mass.gov/eea/docs/dfg/der/pdf/economic-impacts-ma-der.pdf

International Agency for Research on Cancer. (2015). *IARC Monographs Volume 112: evaluation of five organophosphate insecticides and herbicides*. World Health Organization. Retrieved from https://www.iarc.fr/en/media-centre/iarcnews/pdf/MonographVolume112.pdf

IWA. (2015). *Wise Use of Water With Smart Technologies*. Retrieved February 2, 2017, from http://www.iwa-network.org/wise-use-of-water-with-smart-technologies/

IWA. (2016). *Can Alternative Water Resources Become Viable in Low-Income Countries?* Retrieved

February 2, 2017, from http://www.iwa-network.org/can-alternative-water-resources-become-viable-in-low-income-countries/

IPCC. (2014). *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands Methodological Guidance on Lands with Wet and Drained Soils, and Constructed Wetlands for Wastewater Treatment*. T. Hiraishi, T. Krug, K. Tanabe, N. Srivastava, B. Jamsranjav, M. Fukuda, & T. Troxler (Eds.). Switzerland: Intergovernmental Panel on Climate Change. Retrieved from http://www.ipcc-nggip.iges.or.jp

ITTO. (2002). *ITTO Guidelines for the Restoration, Management and Rehabilitation of Degraded and Secondary Tropical Forests*. International Tropical Timber Organization & International Union for the Conservation of Nature. Retrieved from http://cmsdata.iucn.org/downloads/summary_report_flr_national_workshops.pdf

ITTO. (2009). *ITTO/IUCN guidelines for the conservation and sustainable use of biodiversity in tropical timber production forests*. ITTO Policy Development Series No. 17. A joint publication of the International Tropical Timber Organization and the International Union for the Conservation of Nature. Retrieved from https://www.cbd.int/forest/doc/itto-iucn-biodiversity-guidelines-tropical-forests-2009-en.pdf

ITTO. (2016). *Criteria and indicators for the sustainable management of tropical forests*. ITTO Policy Development Series No. 21. Yokohama: International Tropical Timber Organization. Retrieved from http://www.itto.int/direct/topics/topics_pdf_download/topics_id=4872&no=1&disp=inline

IUCN. (2014). *Biodiversity Offsets Technical Study Paper*. Gland: International Union for the Conservation of Nature. Retrieved from https://portals.iucn.org/library/sites/library/files/documents/2014-044.pdf

Izquierdo, A. E., Grau, H. R., & Aide, T. M. (2011). Implications of Rural-Urban Migration for Conservation of the Atlantic Forest and Urban Growth in Misiones, Argentina (1970-2030). *Ambio*, *40*(3), 298–309.

Jat, R. A., Sahrawat, K. L., Kassam, A. H., & Friedrich, T. (2014). Conservation Agriculture for Sustainable and Resilient Agriculture: Global Status, Prospects and Challenges. In R. A. Jat, K. S. Sahrawat, & A. H. Kassam (Eds.), *Conservation Agriculture: Global Prospects and Challenges* (pp. 1–25). Boston, MA: CABI International.

Jin-xing, Z., Chun-yun, Z., Jing-ming, Z., Xiao-hui, W., & Zhou-hong, L. (2002). Landslide disaster in the loess area of China. *Journal of Forestry Research*, *13*(2), 157–161. https://doi.org/10.1007/BF02857244

Johnson, C. E., Driscoll, C. T., Blum, J. D., Fahey, T. J., & Battles, J. J. (2014). Soil Chemical Dynamics after Calcium Silicate Addition to a Northern Hardwood Forest. *Soil Science Society of America Journal*, *78*(4), 1458. https://doi.org/10.2136/sssaj2014.03.0114

Jones, G. M., Gutiérrez, R. J., Tempel, D. J., Whitmore, S. A., Berigan, W. J., & Peery, M. Z. (2016). Megafires: an emerging threat to old-forest species. *Frontiers in Ecology and the Environment*, *14*(6), 300–306. https://doi.org/10.1002/fee.1298

Jones, H. P., & Schmitz, O. J. (2009). Rapid recovery of damaged ecosystems. *PLoS ONE*, *4*(5). https://doi.org/10.1371/journal.pone.0005653

Józefowska, A., Pietrzykowski, M., Woś, B., Cajthaml, T., & Frouz, J. (2017). The effects of tree species and substrate on carbon sequestration and chemical and biological properties in reforested post-mining soils. *Geoderma*, *292*, 9–16.

Kairis, O., Kosmas, C., Karavitis, C., Ritsema, C., Salvati, L., Acikalin, S., Alcalá, M., Alfama, P., Atlhopheng, J., Barrera, J., Belgacem, A., Solé-Benet, A., Brito, J., Chaker, M., Chanda, R., Coelho, C., Darkoh, M., Diamantis, I., Ermolaeva, O., Fassouli, V., Fei, W., Feng, J., Fernandez, F., Ferreira, A., Gokceoglu, C., Gonzalez, D., Gungor, H., Hessel, R., Juying, J., Khatteli, H., Khitrov, N., Kounalaki, A., Laouina, A., Lollino, P., Lopes, M., Magole, L., Medina, L., Mendoza, M., Morais, P., Mulale, K., Ocakoglu, F., Ouessar, M., Ovalle, C., Perez, C., Perkins, J., Pliakas, F., Polemio, M., Pozo, A., Prat, C., Qinke, Y., Ramos, A., Ramos, J., Riquelme, J., Romanenkov, V., Rui, L., Santaloia, F., Sebego, R., Sghaier, M., Silva, N., Sizemskaya, M., Soares, J., Sonmez, H., Taamallah, H., Tezcan, L., Torri, D., Ungaro, F., Valente, S., de Vente, J., Zagal, E., Zeiliguer, A., Zhonging, W., & Ziogas, A. (2014). Evaluation and Selection of Indicators for Land Degradation and Desertification Monitoring: Types of Degradation, Causes, and Implications for Management. *Environmental Management*, *54*(5), 971–982. https://doi.org/10.1007/s00267-013-0110-0

Karavanas, A., Chaloulakou, A., & Spyrellis, N. (2009). Evaluation of the implementation of best available techniques in IPPC context: an environmental performance indicators approach. *Journal of Cleaner Production*, *17*(4), 480–486. https://doi.org/10.1016/j.jclepro.2008.08.016

Karlen, D., & Rice, C. (2015). Soil Degradation: Will Humankind Ever Learn? *Sustainability*, *7*(9), 12490–12501. https://doi.org/10.3390/su70912490

Keeley, J. E. (2006). Fire Management Impacts on Invasive Plants in the Western United States. *Conservation Biology*, *20*(2), 375–384. https://doi.org/10.1111/j.1523-1739.2006.00339.x

Keitt, B., Campbell, K., Saunders, A., & Clout, M. (2011). The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems. In *Island invasives: eradication and management* (pp. 74–77). Gland, Switzerland: IUCN. Retrieved from http://www.issg.org/pdf/publications/Island_Invasives/pdfHQprint/1Keitt.pdf

Kemp, W. M., Boynton, W. R., Adolf, J. E., Boesch, D. F., Boicourt, W. C., Brush, G., Cornwell, J. C., Fisher, T. R., Glibert, P. M., Hagy, J. D., Harding, L. W., Houde, E. D., Kimmel, D. G., Miller, W. D., Newell, R. I. E., Roman, M. R., Smith, E. M., & Stevenson, J. C. (2005). Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology Progress Series. *Marine Ecology Progress Series*, *303*, 1–29. https://doi.org/10.3354/meps303001

Khan, M., & Law, F. (2005). Adverse effects of pesticides and related chemicals on enzyme and hormone systems of fish, amphibians and reptiles: a review. *Proceedings of the Pakistan Academy of Sciences*, *42*(4), 315–323.

Khush, G. S. (1999). Green revolution: preparing for the 21st century. *Genome*, *42*(4), 646–655. https://doi.org/10.1139/gen-42-4-646

Kiage, L. M., Liu, K.-B., Walker, N. D., Lam, N., & Huh, O. K. (2007). Recent land-cover/use change associated with land degradation in the Lake Baringo catchment, Kenya, East Africa: evidence from Landsat TM and ETM+. *International Journal of Remote Sensing*, *28*(19), 4285–4309. https://doi.org/10.1080/01431160701241753

Kidron, G. J., Ying, W., Starinsky, A., & Herzberg, M. (2017). Drought effect on biocrust resilience: High-speed winds result in crust burial and crust rupture and flaking. *Science of the Total Environment*, *579*, 848–859. https://doi.org/10.1016/j.scitotenv.2016.11.016

Kihslinger, R. L. (2008). Success of Wetland Mitigation Projects. *National Wetlands Newsletter*, *30*(2001), 14–17.

Kirui, O. K. (2016). Economics of Land Degradation and Improvement in Tanzania and Malawi. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.) *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 609–649). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_20

Kissinger, G., Brasser, A., & Gross, L. (2013). Scoping study. *Reducing Risk: Landscape Approaches to Sustainable Sourcing.* Washington, DC: Landscapes for People, Food and Nature Initiative. Retrieved from https://peoplefoodandnature.org/wp-content/uploads/2013/04/ReducingRisk_SynthesisReport_2013.pdf

Kittinger, J. N., Bambico, T. M., Minton, D., Miller, A., Mejia, M., Kalei, N., Wong, B., & Glazier, E. W.

(2016). Restoring ecosystems, restoring community: socioeconomic and cultural dimensions of a community-based coral reef restoration project. *Regional Environmental Change*, *16*(2), 301–313. https://doi.org/10.1007/s10113-013-0572-x

Kjellstrom, T., Lodh, M., McMichael, T., Ranmuthugala, G., Shrestha, R., & Kingsland, S. (2006). Air and Water Pollution: Burden and Strategies for Control. In *Disease Control Priorities in Developing Countrie* (2nd ed.). Washington (DC): World Bank.

Kløve, B., Allan, A., Bertrand, G., Druzynska, E., Ertürk, A., Goldscheider, N., Henry, S., Karakaya, N., Karjalainen, T. P., Koundouri, P., Kupfersberger, H., Kvœrner, J., Lundberg, A., Muotka, T., Preda, E., Pulido-Velazquez, M., & Schipper, P. (2011). Groundwater dependent ecosystems. Part II. Ecosystem services and management in Europe under risk of climate change and land use intensification. *Environmental Science & Policy*, *14*(7), 782–793. https://doi.org/10.1016/j.envsci.2011.04.005

Knowler, D., & Bradshaw, B. (2007). Farmers' adoption of conservation agriculture: A review and synthesis of recent research. *Food Policy*, *32*(1), 25–48. https://doi.org/10.1016/j.foodpol.2006.01.003

Koch, B. J., Febria, C. M., Cooke, R. M., Hosen, J. D., Baker, M. E., Colson, A. R., Filoso, S., Hayhoe, K., Loperfido, J. V., Stoner, A. M. K., & Palmer, M. A. (2015). Suburban watershed nitrogen retention: Estimating the effectiveness of stormwater management structures. *Elementa: Science of the Anthropocene*, *3*, 63. https://doi.org/10.12952/journal.elementa.000063

Koch, J. M., & Hobbs, R. J. (2007). Synthesis: Is Alcoa successfully restoring a jarrah forest ecosystem after bauxite mining in Western Australia? *Restoration Ecology*, *15*(SUPPL. 4), 137–144. https://doi.org/10.1111/j.1526-100X.2007.00301.x

Kohli, R. K., Batish, D. R., Singh, H. P., & Dogra, K. S. (2006). Status, invasiveness and environmental threats of three tropical American invasive weeds (Parthenium hysterophorus L., Ageratum conyzoides L., Lantana camara L.) in India. *Biological Invasions*, *8*(7), 1501–1510. https://doi.org/10.1007/s10530-005-5842-1

Komba, C., & Muchapondwa, E. (2016). An analysis of factors affecting household willingness to participate in the REDD+ programme in Tanzania. *Climate and Development*, 1–14. https://doi.org/10.1080/17565529.2016.1145098

Kong, T. M., Austin, D. E., Kellner, K., & Orr, B. J. (2014). The interplay of knowledge, attitude and practice of livestock farmers' land management against desertification in the South African Kalahari. *Journal of Arid Environments*, *105*, 12–21. https://doi.org/10.1016/j.jaridenv.2014.02.002

Koning, N., & Smaling, E. (2005). Environmental crisis or "lie of the land"? The debate on soil degradation in Africa. *Land Use Policy*, *22*(1), 3–11. https://doi.org/10.1016/j.landusepol.2003.08.003

Kosmas, C., & Kelly, C. (2012). Responses to LEDD in cropland: general. Retrieved January 31, 2017, from http://leddris.aegean.gr/responses-to-ledd-5895/160-responses-to-ledd-in-cropland-general.html

Kosoy, N., & Corbera, E. (2010). Payments for ecosystem services as commodity fetishism. *Ecological Economics*, *69*(6), 1228–1236. https://doi.org/10.1016/j.ecolecon.2009.11.002

Krause, T., & Loft, L. (2013). Benefit Distribution and Equity in Ecuador's Socio Bosque Program. *Society & Natural Resources*, *26*(10), 1170–1184. https://doi.org/10.1080/08941920.2013.797529

Kroeger, T. (2013). The quest for the ?optimal? payment for environmental services program: Ambition meets reality, with useful lessons. *Forest Policy and Economics*, *37*, 65–74. https://doi.org/10.1016/j.forpol.2012.06.007

Kumar, R., McInnes, R., Everard, M., Gardner, R., Kulindwa, K. A. A., & Infante Mata, D. (2017). *Integrating multiple wetland values into decision-making* (Ramsar Policy Brief 2). Retrieved from https://www.ramsar.org/sites/default/files/documents/library/rpb_values_of_wetlands_e.pdf

Kumaresan, D., Cross, A. T., Moreira-Grez, B., Kariman, K., Nevill, P., Stevens, J., Allcock, R. J. N., O'Donnell, A. G., Dixon, K. W., & Whiteley, A. S. (2017). Microbial Functional Capacity Is Preserved Within Engineered Soil Formulations Used In Mine Site Restoration. *Scientific Reports*, *7*(1), 564. https://doi.org/10.1038/s41598-017-00650-6

Lahmar, R. (2010). Adoption of conservation agriculture in Europe: Lessons of the KASSA project. *Land Use Policy*, *27*(1), 4–10. https://doi.org/10.1016/j.landusepol.2008.02.001

Lahmar, R., Bationo, B. A., Dan Lamso, N., Gu??ro, Y., & Tittonell, P. (2012). Tailoring conservation agriculture technologies to West Africa semi-arid zones: Building on traditional local practices for soil restoration. *Field Crops Research*, *132*, 158–167. https://doi.org/10.1016/j.fcr.2011.09.013

Lal, R. (2000). Carbon sequestration in drylands. *Annals of Arid Zone*, *39*(1), 1–10.

Lal, R. (2002). Soil carbon dynamics in cropland and rangeland. In *Environmental Pollution, 116*(3), 353-362. https://doi.org/10.1016/S0269-7491(01)00211-1

Lal, R. (2015a). A system approach to conservation agriculture. *Journal of Soil and Water Conservation*, *70*(4), 82A–88A.

Lal, R. (2015b). Sequestering carbon and increasing productivity by conservation agriculture. *Journal of Soil and Water Conservation*, *70*(3), 55A–62A.

Lal, R., Safriel, U., & Boer, B. (2012). *Zero Net Land Degradation: A New Sustainable Development Goal for Rio+20.* United Nations Convention to Combat Desertification.

Lamb, D. (2010). *Regreening the bare hills: tropical forest restoration in the Asia-Pacific region*. New York: Springer.

Lamb, D., Erskine, P. D., & Parrotta, J. A. (2005). Restoration of degraded tropical forest landscapes. *Science*, *310*(5754), 1628–1632. https://doi.org/10.1126/science.1111773

Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences of the United States of America*, *108*(9), 3465–3472. https://doi.org/10.1073/pnas.1100480108

Lambin, E. F., Meyfroidt, P., Rueda, X., Blackman, A., Börner, J., Cerutti, P. O., Dietsch, T., Jungmann, L., Lamarque, P., Lister, J., Walker, N. F., & Wunder, S. (2014). Effectiveness and synergies of policy instruments for land use governance in tropical regions. *Global Environmental Change*, *28*(1), 129–140. https://doi.org/10.1016/j.gloenvcha.2014.06.007

Land, M., Granéli, W., Grimvall, A., Hoffmann, C. C., Mitsch, W. J., Tonderski, K. S., & Verhoeven, J. T. A. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence*, *5*(1), 9. https://doi.org/10.1186/s13750-016-0060-0

Lasco, R. D., Delfino, R. J. P., & Espaldon, M. L. O. (2014). Agroforestry systems: helping smallholders adapt to climate risks while mitigating climate change. *Wiley Interdisciplinary Reviews: Climate Change*, *5*(6), 825–833. https://doi.org/10.1002/wcc.301

Latacz-Lohmann, U., & der Hamsvoort, C. Van. (1997). Auctioning Conservation Contracts: A Theoretical Analysis and an Application. *American Journal of Agricultural Economics*, *79*(2), 407. https://doi.org/10.2307/1244139

Latacz-Lohmann, U., & Schilizzi, S. (2007). Quantifying the Benefits of Conservation Auctions: Evidence from an Economic Experiment. *EuroChoices*, *6*(3), 32–39. https://doi.org/doi:10.1111/j.1746-692X.2007.00073.x

Latawiec, A. E., Strassburg, B. B. N., Silva, D., Alves-Pinto, H. N., Feltran-Barbieri, R., Castro, A., Iribarrem, A., Rangel, M. C., Kalif, K. A. B., Gardner, T., & Beduschi, F. (2017). Improving land management in Brazil: A perspective from producers. *Agriculture, Ecosystems & Environment*, *240*, 276–286.

https://doi.org/10.1016/j.agee.2017.01.043

Laurance, W. F. (2009). Conserving the hottest of the hotspots. *Biological Conservation*, *142*(6), 1137. https://doi.org/10.1016/j.biocon.2008.10.011

Lavigne Delville, P. (2010). Conceptions des droits fonciers, récits de politiques publiques et controverses. Les Plans fonciers ruraux en Afrique de l'Ouest. In Karthala (Ed.), *Les politiques d'enregistrement des droits fonciers. Du cadre légal aux pratiques locales* (pp. 69–103).

Law, E. A., & Wilson, K. A. (2015). Providing Context for the Land-Sharing and Land-Sparing Debate. *Conservation Letters*, *8*(6), 404–413. https://doi.org/10.1111/conl.12168

Lecomte, R. (2008). De la nécessité d'une législation spécifique à la protection des sols; l'exemple des sites contaminés et les limites de l'obligation de remise en état. *Droit de l'Environnement*, *158*, 11–15.

Legge, S., Murphy, S., Kingswood, R., Maher, B., & Swan, D. (2011). EcoFire: restoring the biodiversity values of the Kimberley region by managing fire. *Ecological Management & Restoration*, *12*(2), 84–92. https://doi.org/10.1111/j.1442-8903.2011.00595.x

Lehman, R. M., Cambardella, C. A., Stott, D. E., Acosta-Martinez, V., Manter, D. K., Buyer, J. S., Maul, J. E., Smith, J. L., Collins, H. P., Halvorson, J. J., Kermer, R. J., Lundgren, J. G., Ducey, T. F., Jin, V. L., & Karlen, D. L. (2015). Understanding and enhancing soil biological health: The solution for reversing soil degradation. *Sustainability*, *7*, 988–1027.

Lei, K., Pan, H., & Lin, C. (2016). A landscape approach towards ecological restoration and sustainable development of mining areas. *Ecological Engineering*, *90*, 320–325. https://doi.org/https://doi.org/10.1016/j.ecoleng.2016.01.080

Leibfried, S., Huber, E., Lange, M., Levy, J. D., Nullmeier, F., & Stephens, J. D. (2015). *The Oxford handbook of transformations of the state*. Oxford: Oxford University Press.

Lemenih, M. (2004). *Effects of land use changes on soil quality and native flora degradation and restoration in the highlands of Ethiopia*. New York. Retrieved from http://urn.kb.se/resolve?urn=urn:nbn:se:slu:epsilon-273

Lestrelin, G., Castella, J.-C., & Bourgoin, J. (2012). Territorialising Sustainable Development: The Politics of Land-use Planning in Laos. *Journal of Contemporary Asia*, *42*(4), 581–602. https://doi.org/10.1080/00472336.2012.706745

Leung, B., Lodge, D. M., Finnoff, D., Shogren, J. F., Lewis, M. A., & Lamberti, G. (2002). An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. *Proceedings of the Royal Society of London B: Biological Sciences*, *269*(1508). Retrieved from http://rspb.royalsocietypublishing.org/content/269/1508/2407.short

Levidow, L., Lindgaard-Jørgensen, P., Nilsson, Å., Skenhall, S. A., & Assimacopoulos, D. (2016). Process eco-innovation: Assessing meso-level eco-efficiency in industrial water-service systems. *Journal of Cleaner Production*, *110*, 54–65. https://doi.org/10.1016/j.jclepro.2014.12.086

Lewis III, R. R. (2000). Ecologically based goal setting in mangrove forest and tidal marsh restoration. *Ecological Engineering*, *15*(3–4), 191–198. https://doi.org/10.1016/S0925-8574(00)00070-7

Liang, D., Fu, B., Lu, Y., Liu, Y., Gao, G., Li, Y., & Li, Z. (2012). Effects of retired steep land afforestation on soil properties: A case study in the Loess Plateau of China. *Acta Agriculturae Scandinavica*, *62*(6), 547–555.

Light, A. (2000). Restoration, the Value of Participation, and the Risks of Professionalization. In P. Gobster & B. Hull (Eds.), *Restoring nature: Perspectives from the social sciences and humanities* (pp. 163–181). Island Press.

Liniger, H., Cahill, D., Thomas, D., van Lynden, G., & Schwilch, G. (2002). Categorization of SWC technologies and approaches–a global need. In *Proceedings of International Soil Conservation Organization (ISCO) Conference*. Retrieved from http://tucson.ars.ag.gov/isco/isco12/VolumeIII/CategorizationofSWCTechnologies.pdf

Liniger, H., & Critchley, W. (Eds.). (2007). *Where the Land is Greener: Case Studies and Anlysis of Soil and Water Conservation Initiatives Worldwide*. WOCAT - World Overview of Conservation Approaches and Technologies. CTA, FAO, UNEP and CDE. Retrieved from https://www.wocat.net/library/media/27/

Liu, G.Q., Ni, W. J. (2002). On Some Problems of Vegetation Rehabilitation in the Loess Plateau. In J. Juren (Ed.), *Sustainable Utilisation of Global Soil and Water resources. Technology and Method of Soil and Water conservation.* (pp. 217–222). Beijing: Proceedings of 12th International Soil Conservation Organisation Conf., Beijing, May 26–31, 2002. Vol. 3. Tsinghua University Press.

Liu, H., Bauer, L. S., Zhao, T., Gao, R., & Poland, T. M. (2016). Seasonal abundance and development of the Asian longhorned beetle and natural enemy prevalence in different forest types in China. *Biological Control*, *103*, 154–164. https://doi.org/10.1016/j.biocontrol.2016.08.010

Liu, J. D., & Hiller, B. T. (2016). A continuing inquiry into ecosystem restoration: examples from China's Loess plateau and locations worldwide and their emerging implications. In I. Chabay, M. Frick, & J. Helgeson (Eds.), *Land Restoration: Reclaiming Landscapes for a Sustainable Future* (pp. 361–382). Academic Press, Elsevier.

Liu, X., Ou, J., Li, X., & Ai, B. (2013). Combining system dynamics and hybrid particle swarm optimization for land use allocation. *Ecological Modelling*, *257*, 11–24. https://doi.org/10.1016/j.ecolmodel.2013.02.027

Long, J., Tecle, A., & Burnette, B. (2003). Cultural Foundations for Ecological Restoration on the White Mountain Apache Reservation. *Conservation Ecology*, *8*(1), art4. https://doi.org/10.5751/ES-00591-080104

Low, P.S (2013). *White Paper I: Economic and Social Impacts of Desertification, Land Degradation and Drought.* UNCCD 2nd Scientific Conference, prepared with the contributions of an international group of scientists. Bonn: United Nations Convention to Combat Desertification. Retrieved from http://knowledge.unccd.int/sites/default/files/inline-files/unccd_white_paper_1.pdf

Lu, Y., Waldmann, N., Nadel, D., & Marco, S. (2017). Increased sedimentation following the Neolithic Revolution in the Southern Levant. *Global and Planetary Change*, *152*(April), 199–208. https://doi.org/10.1016/j.gloplacha.2017.04.003

Lukas, M. C. (2014). Eroding battlefields: Land degradation in Java reconsidered. *Geoforum*, *56*, 87–100. https://doi.org/10.1016/j.geoforum.2014.06.010

Lund, J. F., Sungusia, E., Mabele, M. B., & Scheba, A. (2017). Promising Change, Delivering Continuity: REDD+ as Conservation Fad. *World Development*, *89*(13), 124–139. https://doi.org/10.1016/j.worlddev.2016.08.005

Luo, J.N., Fan, Y.D., Shi, Peijun, Lu, W.J., Zhou, J.-H. (2003). Information-comparable method of monitoring the intensity of dust storm by multisource data of remote sensing. *Journal of Natural Disasters*, *12*(2), 28–34.

M.E., R., & Zhu X.M. (1994). Anthropogenic influences on changes in the sediment load of the Yellow River, China, During the Holocene. *The Holocene*, *4*, 314–320.

MacDicken, K. G., Sola, P., Hall, J. E., Sabogal, C., Tadoum, M., & de Wasseige, C. (2015). Global progress toward sustainable forest management. *Forest Ecology and Management*, *352*, 47–56. https://doi.org/10.1016/j.foreco.2015.02.005

Macedo, M. N., DeFries, R. S., Morton, D. C., Stickler, C. M., Galford, G. L., & Shimabukuro, Y. E. (2012). Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(4), 1341–1346. https://doi.org/10.1073/pnas.1111374109

Mack, R. N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M., & Bazzaz, F. A. (2000). Biotic invasions: Causes, epidemiology,global consequence, and control. *Ecological Applications*, *10*, 689–710. https://doi.org/10.1890/1051-0761(2000)010[0689:BICEGC]2.0.CO;2

MAFRI. (2008). *Soil Management Guide*. Retrieved from https://www.gov.mb.ca/agriculture/environment/soil-management/soil-management-guide/

Maginnis, S., & Jackson, W. (2003). The role of planted forests in forest landscape restoration. In *UNFF Intersessional Experts Meeting on the Role of Planted Forests in Sustainable Forest management,*. New Zealand. Retrieved from http://www.globalrestorationnetwork.org/uploads/files/LiteratureAttachments/337_the-role-of-planted-forests-in-forest-landscape-restoration.pdf

Maltby, E. (2009). The Changing Wetland Paradigm. In E. Maltby & T. Barker (Eds.), *The Wetlands Handbook* (pp. 3–42). Oxford, UK: Wiley Blackwell.

Maltby, E., & Acreman, M. C. (2011). Ecosystem services of wetlands: pathfinder for a new paradigm. *Hydrological Sciences Journal*, *56*(8), 1341–1359. https://doi.org/10.1080/02626667.2011.631014

Mansourian, S., & Vallauri, D. (2014). Restoring Forest Landscapes: Important Lessons Learnt. *Environmental Management*, *53*(2), 241–251. https://doi.org/10.1007/s00267-013-0213-7

Marais, C., & Wannenburgh, A. M. (2008). Restoration of water resources (natural capital) through the clearing of invasive alien plants from riparian areas in South Africa - Costs and water benefits. *South African Journal of Botany*, *74*(3), 526–537. https://doi.org/10.1016/j.sajb.2008.01.175

Markwei, C., Ndlovu, L., Robinson, E., & Shah, W. P. (2008). *International Assessment of Agricultural Knowledge, Science, and Technology for Development (IAASTD). Sub-Saharan Africa Summary for Decision Makers.* Retrieved from https://www.fanrpan.org/archive/documents/d00522/IAASTD_summary_decision_makers.pdf

Marois, D. E., & Mitsch, W. J. (2015). Coastal protection from tsunamis and cyclones provided by mangrove wetlands – a review. *International Journal of Biodiversity Science, Ecosystem Services & Management*, *11*(1), 71–83. https://doi.org/10.1080/21513732.2014.997292

Martínez-Navarro, B., Belmaker, M., & Bar-Yosef, O. (2012). The Bovid assemblage (Bovidae, Mammalia) from the Early Pleistocene site of 'Ubeidiya, Israel: Biochronological and paleoecological implications for the fossil and lithic bearing strata. *Quaternary International*, *267*(April 2016), 78–97. https://doi.org/10.1016/j.quaint.2012.02.041

Mbaga-Semgalawe, Z., & Folmer, H. (2000). Household adoption behaviour of improved soil conservation: the case of the North Pare and West Usambara Mountains of Tanzania. *Land Use Policy*, *17*(4), 321–336. https://doi.org/10.1016/S0264-8377(00)00033-8

Mbow, C., Smith, P., Skole, D., Duguma, L., & Bustamante, M. (2014). Achieving mitigation and adaptation to climate change through sustainable agroforestry practices in Africa. *Current Opinion in Environmental Sustainability*, *6*, 8–14. https://doi.org/10.1016/j.cosust.2013.09.002

McCrackin, M. L., Jones, H. P., Jones, P. C., & Moreno-Mateos, D. (2017). Recovery of lakes and coastal marine ecosystems from eutrophication: A global meta-analysis. *Limnology and Oceanography*, *62*(2), 507–518. https://doi.org/10.1002/lno.10441

McGeoch, M. A., Genovesi, P., Bellingham, P. J., Costello, M. J., McGrannachan, C., & Sheppard, A. (2016). Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion.

Biological Invasions, *18*(2), 299–314. https://doi.org/10.1007/s10530-015-1013-1

McGuire, D. (2014). FAO's Forest and Landscape Restoration Mechanism. In J. Chavez-Tafur & R. Zagt (Eds.), *Towards productive landscapes*. Wageningen, Netherlands: Tropenbos International. Retrieved from http://www.worldagroforestry.org/region/sea/publications/detail?pubID=3289

McKenney, B. A., & Kiesecker, J. M. (2010). Policy development for biodiversity offsets: A review of offset frameworks. *Environmental Management*, *45*(1), 165–176. https://doi.org/10.1007/s00267-009-9396-3

Mckinney, M. L. (2002). Urbanization, Biodiversity and Conservation. *BioScience*, *52*(10), 883–890.

McKinney, R. A., Raposa, K. B., & Kutcher, T. E. (2010). Use of urban marine habitats by foraging wading birds. *Urban Ecosystems*, *13*(2), 191–208. https://doi.org/10.1007/s11252-009-0111-1

McLean, K. G. (2010). *Advance Guard: Climate Change Impacts, Adaptation, Mitigation and Indigenous Peoples – A Compendium of Case Studies.* Darwin: United Nations University, Traditional Knowledge Initiative. Retrieved from https://tfm.unu.edu/publications/books/2010-advance-guard-climate-change-compendium.html#overview

McNeeley, S. M., Even, T. L., Gioia, J. B. M., Knapp, C. N., & Beeton, T. A. (2017). Expanding vulnerability assessment for public lands: The social complement to ecological approaches. *Climate Risk Management, 16*, 106-119. https://doi.org/10.1016/j.crm.2017.01.005

McWilliam, W., Brown, R., Eagles, P., & Seasons, M. (2015). Evaluation of planning policy for protecting green infrastructure from loss and degradation due to residential encroachment. *Land Use Policy*, *47*, 459–467. https://doi.org/10.1016/j.landusepol.2015.05.006

MEA. (2005a). *Ecosystems and human well-being: Synthesis*. Washington, DC: Island Press.

MEA. (2005b). *Ecosystems and Human Well-Being: Current State and Trends: Findings of the Condition and Trends Working Group*. Millennium Ecosystem Assessment. Washington, DC: Island Press.

Mechlem, K. (2006). Harmonizing Trade in Agriculture and Human Rights: Options for the Integration of the Right to Food into the Agreement on Agriculture. *Max Planck Yearbook of United Nations Law Online*, *10*(1), 127–190. https://doi.org/10.1163/138946306783559968

Meir, A., & Tsoar, H. (1996). International Borders and Range Ecology : The Case of Bedouin Transborder Grazing. *Human Ecology*, *24*(1), 39–64.

Mekdaschi-Studer, R., & Liniger, H. (2013). *Water harvesting Guidelines to Good Practice*. Centre for Development and Environment (CDE), Bern; Rainwater Harvesting Implementation Network (RAIN), Amsterdam; MetaMeta, Wageningen; The International Fund for Agricultural Development (IFAD), Rome. Retrieved from http://www.fao.org/sustainable-forest-management/toolbox/tools/tool-detail/en/c/280390/

Mekouar, A. M. (2006). Le texte révisé de la convention africaine sur la conservation de la nature et des ressources naturelles: petite histoire d'une grande rénovation. *FAO Legal Papers Online*.

Meli, P., Rey Benayas, J. M., Balvanera, P., & Martínez Ramos, M. (2014). Restoration Enhances Wetland Biodiversity and Ecosystem Service Supply, but Results Are Context-Dependent: A Meta-Analysis. *PLoS ONE*, *9*(4), e93507. Retrieved from http://dx.doi.org/10.1371%252Fjournal.pone.0093507

Melo, F. P. L., Arroyo-Rodriguez, V., Fahrig, L., Martinez-Ramos, M., & Tabarelli, M. (2013). On the hope for biodiversity-friendly tropical landscapes. *Trends in Ecology & Evolution*, *28*(8), 462–468. https://doi.org/10.1016/j.tree.2013.01.001

Melo, F. P. L., Pinto, S. R. R., Brancalion, P. H. S., Castro, P. S., Rodrigues, R. R., Aronson, J., & Tabarelli, M. (2013). Priority setting for scaling-up tropical forest restoration projects: Early lessons from the Atlantic Forest Restoration Pact. *Environmental Science & Policy*, *33*, 395–404.

https://doi.org/10.1016/j.envsci.2013.07.013

Mendez, M. O., & Maier, R. M. (2008). Phytoremediation of mine tailings in temperate and arid environments. *Reviews in Environmental Science and Biotechnology, 7*(1), 47-59. https://doi.org/10.1007/s11157-007-9125-4

Mendoza, G. A., & Prabhu, R. (2003). Qualitative multi-criteria approaches to assessing indicators of sustainable forest resource management. *Forest Ecology and Management*, *174*(1–3), 329–343. https://doi.org/10.1016/s0378-1127(02)00044-0

Mertz, O., & Mertens, C. F. (2017). Land Sparing and Land Sharing Policies in Developing Countries – Drivers and Linkages to Scientific Debates. *World Development, 98*, 523-535. https://doi.org/10.1016/j.worlddev.2017.05.002

Messing, R., & Wright, M. (2006). Biological control of invasive species: solution or pollution? *Frontiers in Ecology and Environment*, *4*(3), 132–140. https://doi.org/10.1890/1540-9295(2006)004[0132:BCOISS]2.0.CO;2

Milder, J. C., Scherr, S. J., Bracer, C., & others. (2010). Trends and future potential of payment for ecosystem services to alleviate rural poverty in developing countries. *Ecology and Society*, *15*(2), 4.

Miller, R. W., Hauer, R. J., & Werner, L. P. (2015). *Urban Forestry, Planning and Managing Urban Greenspaces*. Waveland Press.

Minnemeyer, S., Laestadius, L., Sizer, N., Saint-Laurent, C., & Potapov, P. (2011). *Atlas of Forest and Landscape Restoration Opportunities*. Washington (DC): World Resources Institute.

Mirzabaev, A., Goedecke, J., Dubovyk, O., Djanibekov, U., & Aw-Hassan, A. (2016). Economics of land degradation in Central Asia. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 261–290). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_7

Mirzabaev, A., Nkonya, E., & von Braun, J. (2015). Economics of sustainable land management. *Current Opinion in Environmental Sustainability*, *15*, 9–19. https://doi.org/10.1016/j.cosust.2015.07.004

Mishra, P. . (2002). *Indigenous Technical Knowledge (ITK) on Soil & Water Conservation in Rainfed Areas*. Retrieved from http://www.crida.in/DRM2-Winter School/PKM.pdf

Mitsch, W. J., Wu, X., Nairn, R., Weihe, P., Wang, N., Deal, R., Boucher, C., Mitsch, W. J., Wu, X., Nairn, R. W., Weihe, P. E., Wang, N., Deal, R., & Boucher, C. E. (1998). Creating and restoring wetlands: a whole-ecosystem experiment in self-design. *BioScience*, *48*(12), 1019–1030. Retrieved from http://www.jstor.org/stable/1313458

Moebius-Clune, B. N., Moebius-Clune, D. J., Gugino, B. K., Idowu, O. J., Schindelbeck, R. R., Ristow, A. J., van Es, H. M., Thies, J. E., Shayler, H. A., McBride, M. B., Kurtz, K. S. M., Wolfe, D. W., & Abawi, G. S. (2016). *Comprehensive Assessment of Soil Health – The Cornell Framework.* Edition 3.2. Geneva, New York: Cornell University. Retrieved from http://css.cornell.edu/extension/soil-health/manual.pdf

Montagnini, F., & Nair, P. K. R. (2004). Carbon sequestration: An underexploited environmental benefit of agroforestry systems. In *Agroforestry Systems, 61*, 281. https://doi.org/10.1023/B:AGFO.0000029005.92691.79

Montanarella, L., & Vargas, R. (2012). Global governance of soil resources as a necessary condition for sustainable development. *Current Opinion in Environmental Sustainability*, *4*(5), 559–564. https://doi.org/10.1016/j.cosust.2012.06.007

Monty, F., Murti, R., & Furuta, N. (2016). *Helping nature help us: Transforming disaster risk reduction through ecosystem management | PreventionWeb.net*. Gland, Switzerland . Retrieved from http://www.preventionweb.net/publications/view/51240

Moore, P. D. (2000). Plant ecology: Alien invaders. *Nature*, *403*(6769), 492–493. https://doi.org/10.1038/35000686

Morales, C., Brzovic, F., Dascal, G., Aranibar, Z., Mora, L., & Morera, R. (2011). *Measuring the economic value of land degradation / desertification considering the effects of climate change. A study for Latin America and the Caribbean*. Montpellier. Retrieved from http://www.csf-desertification.org/files/pdf/seminaire-juin-2011/session-1/S1-Morales%20CSFD_juin_2011.pdf

Moreno-Mateos, D., Power, M. E., Comín, F. A., & Yockteng, R. (2012). Structural and Functional Loss in Restored Wetland Ecosystems. *PLoS Biology*, *10*(1), e1001247. https://doi.org/10.1371/journal.pbio.1001247

Moussa, B., Nkonya, E., Meyer, S., Kato, E., Johnson, T., & Hawkins, J. (2016). Economics of Land Degradation and Improvement in Niger. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 499–539). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_17

Mukul, S., & Saha, N. (2017). Conservation Benefits of Tropical Multifunctional Land-Uses in and Around a Forest Protected Area of Bangladesh. *Land*, *6*(1), 2. https://doi.org/10.3390/land6010002

Mulinge, W., Gicheru, P., Murithi, F., Maingi, P., Kihiu, E., Kirui, O. K., & Mirzabaev, A. (2016). Economics of Land Degradation and Improvement in Kenya. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 470–498). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_16

Müller, N., & Kamada, M. (2011). URBIO: An introduction to the International Network in Urban Biodiversity and Design. *Landscape and Ecological Engineering*, *7*(1), 1–8. https://doi.org/10.1007/s11355-010-0139-7

Murthy, K. I., Dutta, S., Varghese, V., Joshi, P. P., Kumar, P., Murthy α, I. K., Dutta σ, S., Varghese ρ, V., & Joshi Ѡ, P. P. (2016). Impact of Agroforestry Systems on Ecological and Socio- Economic Systems: A Review. *Global Journal of Science Frontier Research: H Environment & Earth Science, 16*(5). Retrieved from http://www.journalofscience.org/index.php/GJSFR/article/viewFile/1934/1795

Myers, R. L. (2006). *Living with Fire: Sustaining Ecosystems and Livelihoods through Integrated Fire Management*. Tallahassee, FL: The Nature Conservancy.

Myers, R. . (1993). *Slope Stabilization Erosion Control Using Vegetation: A Manual of Practice for Coastal Bluff*. Department of Ecology State of Washington. Retrieved from https://fortress.wa.gov/ecy/publications/documents/9330.pdf

Mythili, G., & Goedecke, J. (2016). Economics of Land Degradation in India. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 431–469). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_15

Naiman, R. J., Decamps, H., & Pollock, M. (1993). The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications*, *3*(2), 209–212. https://doi.org/10.2307/1941822

Nair, K. S. S. (2007). *Tropical Forest Insect Pests*. Cambridge University Press. https://doi.org/10.1017/CBO9780511542695

Nair, P. K. R. (1993). *An introduction to agroforestry*. Kluwer Academic Publishers in cooperation with International Centre for Research in Agroforestry. Retrieved from https://www.worldagroforestry.org/Units/Library/Books/PDFs/32_An_introduction_to_agroforestry.pdf?n=161

Nair, P. K. R., Kumar, B. M., & Nair, V. D. (2009). Agroforestry as a strategy for carbon sequestration.

Journal of Plant Nutrition and Soil Science, 172(1), 10-23. https://doi.org/10.1002/jpln.200800030

Nair, P. K. R., Nair, V. D., Mohan Kumar, B., & Showalter, J. M. (2010). Chapter Five – Carbon Sequestration in Agroforestry Systems. In *Advances in Agronomy, 108*, 237–307. https://doi.org/10.1016/S0065-2113(10)08005-3

Narayan, S., Beck, M. W., Reguero, B. G., Losada, I. J., Van Wesenbeeck, B., Pontee, N., Sanchirico, J. N., Ingram, J. C., Lange, G. M., & Burks-Copes, K. A. (2016). The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PLoS ONE*, *11*(5), 1–17. https://doi.org/10.1371/journal.pone.0154735

National Research Council. (2001). *Compensating for wetland losses under the Clean Water Act*. Washington D.C., USA: National Academy Press.

Natural Capital Declaration. (2015). *Towards Including Natural Resource Risks in Cost of Capital*. Retrieved from http://www.unepfi.org/fileadmin/documents/NCD-NaturalResourceRisksScopingStudy.pdf

NCC. (2016). *Natural Capital Protocol*. Natural Capital Coalition. Retrieved from http://naturalcapitalcoalition.org/protocol/

Nel, J., Colvin, C., Maitre, D. Le, Smith, J., & Haines, I. (2013). *South Africa's Strategic Water Source Areas*.

Ngendakumana, S., & Bachange, E. (2013). *Rethinking Rights and Interests of Local Communities in REDD*. *ISRN Forestry*. Retrieved from http://downloads.hindawi.com/journals/isrn.forestry/2013/830902.pdf

Nguyen, D. C., Dao, A. D., Kim, T.-I., & Han, M. (2013). A Sustainability Assessment of the Rainwater Harvesting System for Drinking Water Supply: A Case Study of Cukhe Village, Hanoi, Vietnam. *Environmental Engineering Research*, *18*(2), 109–114. https://doi.org/10.4491/eer.2013.18.2.109

Nielsen-Pincus, M., & Cassandra Moseley. (2010). *Economic and Employment Impacts of Forest and Watershed Restoration in Oregon*. Ecosystem Workforce Program. Working Paper 24. Oregon: Institute for a sustainable environment. Retrieved from http://ewp.uoregon.edu/sites/ewp.uoregon.edu/files/downloads/WP24.pdf

Nielsen, T. T., & Adriansen, H. K. (2005). Government policies and land degradation in the Middle East. *Land Degradation & Development*, *16*(2), 151–161. https://doi.org/10.1002/ldr.677

Nkonya, E., Johnson, T., Kwon, H. Y., & Kato, E. (2016). Economics of land degradation in Sub-Saharan Africa. In E. Nkonya, A. Mirzabaev, & J. von Braun (Eds.), *Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development* (pp. 251–259). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-19168-3_7

Nordhaus, W. (2014). Estimates of the Social Cost of Carbon: Concepts and Results from the DICE-2013R Model and Alternative Approaches. *Journal of the Association of Environmental and Resource Economists*, *1*(1/2), 273–312. https://doi.org/10.1086/676035

Norgaard, R. B. (2010). Ecosystem services: From eye-opening metaphor to complexity blinder. *Ecological Economics*, *69*(6), 1219–1227. https://doi.org/10.1016/j.ecolecon.2009.11.009

Norton, D. A. (1998). Indigenous Biodiversity Conservation and Plantation Forestry : Options for the future. *N. Z. Forestry*, (August), 34–39.

Ntshotsho, P., Prozesky, H. E., Esler, K. J., & Reyers, B. (2015a). What drives the use of scientific evidence in decision making? The case of the South African Working for Water program. *Biological Conservation*, *184*(0), 136–144. https://doi.org/http://dx.doi.org/10.1016/j.biocon.2015.01.021

Ntshotsho, P., Forsyth, G., Maitre, D. Le, Sitas, N., & Yapi, T. (2015b). Two decades of managing invasive alien plants : exploring Working for Water success stories : natural environment. *CSIR Science Scope*, *8*(2), 100–101. Retrieved from https://journals.co.za/content/csir_sci/8/2/EJC179976

OECD. (2015). *OECD Environmental Performance Reviews: Brazil 2015* (OECD Environmental Performance Reviews). *OECD Environmental Performance Reviews*. Paris: OECD Publishing. https://doi.org/10.1787/9789264240094-en

OECD. (2016). *Biodiversity Offsets: Effective Design and Implementation*. Paris: OECD Publishing. https://doi.org/10.1787/9789264222519-en

ONS. (2017). *Principles of Natural Capital Accounting*. London: Office for National Statistics. Retrieved from https://www.ons.gov.uk/economy/environmentalaccounts/methodologies/principlesofnaturalcapitalaccounting

Orr, B. J., Cowie, A. L., Castillo Sanchez, V. M., Chasek, P., Crossman, N. D., Erlewein, A., Louwagie, G., Maron, M., Metternicht, G. I., Minelli, S., Tengberg, A. E., Walter, S., & Welton, S. (2017). *Scientific Conceptual Framework for Land Degradation Neutrality*. Bonn: A Report of the Science-Policy Interface. Advance copy. United Nations Convention to Combat Desertification.

Orth, R., Moore, K., Marion, S., Wilcox, D., & Parrish, D. (2012). Seed addition facilitates eelgrass recovery in a coastal bay system. *Marine Ecology Progress Series*, *448*, 177–195. https://doi.org/10.3354/meps09522

Ostrom, E. (1990). *Governing the Commons: The Evolution of Institutions for Collective Action*. New York: Cambridge University Press.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., & Policansky, D. (1999). Revisiting the commons: local lessons, global challenges. *Science*, *284*(5412), 278–282. https://doi.org/10.1126/science.284.5412.278

Ouédraogo, M.-M., Degré, A., Debouche, C., & Lisein, J. (2014). The evaluation of unmanned aerial system-based photogrammetry and terrestrial laser scanning to generate DEMs of agricultural watersheds. *Geomorphology*, *214*, 339–355. https://doi.org/10.1016/j.geomorph.2014.02.016

Ouyang, Z., Zheng, H., Xiao, Y., Polasky, S., Liu, J., Xu, W., Wang, Q., Zhang, L., Xiao, Y., Rao, E., Jiang, L., Lu, F., Wang, X., Yang, G., Gong, S., Wu, B., Zeng, Y., Yang, W., & Daily, G. C. (2016). Improvements in ecosystem services from investments in natural capital. *Science*, *352*(6292), 1455–1460.

Pagdee, A., Kim, Y., & Daugherty, P. J. (2006). What Makes Community Forest Management Successful: A Meta-Study From Community Forests Throughout the World. *Society & Natural Resources*, *19*(1), 33–52. Journal Article. https://doi.org/10.1080/08941920500323260

Pagiola, S. (1996). Price policy and returns to soil conservation in semi-arid Kenya. *Environmental & Resource Economics*, *8*(3), 255–271. Retrieved from http://search.proquest.com/docview/15921640?accountid=14681

Pahl-Wostl, C. (2006). The importance of social learning in restoring the multifunctionality of rivers and floodplains. *Ecology and Society*, *11*(1).

Paleari, S. (2017). Is the European Union protecting soil? A critical analysis of Community environmental policy and law. *Land Use Policy*, *64*, 163–173. https://doi.org/10.1016/j.landusepol.2017.02.007

Pandit, R., & Bevilacqua, E. (2011). Forest users and environmental impacts of community forestry in the hills of Nepal. *Forest Policy and Economics*, *13*(5), 345–352. https://doi.org/10.1016/j.forpol.2011.03.009

Pandit, R., & Laband, D. N. (2010). Energy savings from tree shade. *Ecological Economics*, *69*(6), 1324–1329. https://doi.org/10.1016/J.ECOLECON.2010.01.009

Pandit, R., Polyakov, M., & Sadler, R. (2014). Valuing public and private urban tree canopy cover. *Australian Journal of Agricultural and Resource Economics*, *58*(3), 453–470. https://doi.org/10.1111/1467-8489.12037

Pannell, D. J., McFarlane, D. J., & Ferdowsian, R. (2001). Rethinking the externality issue for dryland salinity in Western Australia. *The Australian Journal of Agricultural and Resource Economics*, *45*(3), 459–475. https://doi.org/10.1111/1467-8489.00152

Paquette, A., & Messier, C. (2010). The role of plantations in managing the world's forests in the Anthropocene. *Frontiers in Ecology and the Environment*, *8*(1), 27–34. https://doi.org/10.1890/080116

Park, M. S., & Youn, Y.-C. (2017). Reforestation policy integration by the multiple sectors toward forest transition in the Republic of Korea. *Forest Policy and Economics*, *76*, 45–55. https://doi.org/10.1016/j.forpol.2016.05.019

Parrotta, J. A., & Knowles, O. H. (2001). Restoring tropical forests on lands mined for bauxite: Examples from the Brazilian Amazon. *Ecological Engineering*, *17*(2), 219–239. https://doi.org/10.1016/S0925-8574(00)00141-5

Parrotta, J. A., & Trosper, R. L. (2012). *Traditional Forest-Related Knowledge. Sustaining Communities, Ecosystems and Biocultural Diversity*. Netherlands: Springer.

Parrotta, J. A., de Pryck, J. D., Obiri, B. D., Padoch, C., Powell, B., Sandbrook, C., Agarwal, B., Ickowitz, A., Jeary, K., Serban, A., Sunderland, T. C. H., & Tran, N.T. (2015). The historical, environmental and socio-economic context of forests and tree-based systems for food security and nutrition. In S. Vira, B., Wildburger, C., Mansourian (Ed.), *Forests and Food: Addressing Hunger and Nutrition Across Sustainable Landscapes* (pp. 73-136). Cambridge UK: Open Book Publishers.

Payn, T., Carnus, J.-M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., Orazio, C., Rodriguez, L., Silva, L. N., & Wingfield, M. J. (2015). Changes in planted forests and future global implications. *Forest Ecology and Management*, *352*, 57–67. https://doi.org/10.1016/j.foreco.2015.06.021

Pearce, D., Putz, F. E., & Vanclay, J. K. (2003). Sustainable forestry in the tropics: panacea or folly? *Forest Ecology and Management*, *172*(2–3), 229–247. https://doi.org/10.1016/s0378-1127(01)00798-8

Pelletier, J., Gélinas, N., & Skutsch, M. (2016). The place of community forest management in the REDD+ landscape. *Forests*, *7*(8), 1–24. https://doi.org/10.3390/f7080170

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., Craft, C., Fourqurean, J. W., Kauffman, J. B., Marb?, N., Megonigal, P., Pidgeon, E., Herr, D., Gordon, D., & Baldera, A. (2012). Estimating Global ?Blue Carbon? Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems. *PLoS ONE*, *7*(9), e43542. https://doi.org/10.1371/journal.pone.0043542

Peng, H., & Coster, J. (2007). The Loess Plateau: Finding a place for forests. *Journal of Forestry*, *105*(8), 409–413.

Pert, P. L., Ens, E. J., Locke, J., Clarke, P. A., Packer, J. M., & Turpin, G. (2015). An online spatial database of Australian Indigenous Biocultural Knowledge for contemporary natural and cultural resource management. *Science of The Total Environment*, *534*, 110–121. https://doi.org/10.1016/j.scitotenv.2015.01.073

Pest Management Regulatory Agency. (2017). *Re-evaluation Decision RVD2017-01, Glyphosate*. Retrieved from https://www.canada.ca/en/health-canada/services/consumer-product-safety/reports-publications/pesticides-pest-management/decisions-updates/registration-decision/2017/glyphosate-rvd-2017-01.html

Petursdottir, T., Arnalds, O., Baker, S., Montanarella, L., & Aradóttir, Á. L. (2013). A social-ecological system approach to analyze stakeholders' interactions within a large-scale rangeland restoration program. *Ecology and Society, 18*(2), 29. https://doi.org/10.5751/ES-05399-180229

Peyraud, J. L., Taboada, M., & Delaby, L. (2014). Integrated crop and livestock systems in Western Europe and South America: A review. *European Journal of Agronomy*, *57*, 31–42.

https://doi.org/10.1016/j.eja.2014.02.005

Phalan, B., Onial, M., Balmford, A., & Green, R. E. (2011). Reconciling food production and biodiversity conservation: Land sharing and land sparing compared. *Science*, *333*(6047), 1289–1291. https://doi.org/10.1126/science.1208742

Phelps, J., Webb, E. L., & Agrawal, A. (2010). Land use. Does REDD+ threaten to recentralize forest governance? *Science*, *328*(5976), 312–313. https://doi.org/10.1126/science.1187774

Piguet, E. (2012). Migration: The drivers of human migration. *Nature Climate Change*, *2*(6), 400–401. https://doi.org/10.1038/nclimate1559

Pilli, R., Grassi, G., Kurz, W. A., Viñas, R. A., & Guerrero, N. H. (2016). Modelling forest carbon stock changes as affected by harvest and natural disturbances. I. Comparison with countries' estimates for forest management. *Carbon Balance and Management*, *11*(1), 5. https://doi.org/10.1186/s13021-016-0047-8

Pinto, S., Melo, F., Tabarelli, M., Padovesi, A., Mesquita, C., de Mattos Scaramuzza, C., Castro, P., Carrascosa, H., Calmon, M., Rodrigues, R., César, R., & Brancalion, P. (2014). Governing and Delivering a Biome-Wide Restoration Initiative: The Case of Atlantic Forest Restoration Pact in Brazil. *Forests*, *5*(9), 2212–2229. https://doi.org/10.3390/f5092212

Pittock, J., Finlayson, M., Arthington, A. H., Roux, D., Matthews, J. H., Biggs, H., Blom, E., Flitcroft, R., Froend, R., Harrison, I., Hermoso, V., Junk, W., Kumar, R., Linke, S., Nel, J., Nunes, C., Cunha, D., Pattnaik, A., Pollard, S., Rast, W., Thieme, M., Turak, E., Turpie, J., Van Niekerk, L., Willems, D., & Viers, J. (2015). 'Managing freshwater, river, wetland and estuarine protected areas. In S. F. and I. P. G. L. Worboys, M. Lockwood, A. Kothari (Ed.), *Protected Area Governance and Management,* (pp. 569–608). Canberra. Retrieved from http://sru.nmmu.ac.za/sru/media/Store/Pittock-et-al_2015_Managing-fresh-water-protected-areas_Chapter-19.pdf

Plançon, C. (2009). Droit, foncier et développement: les enjeux de la notion de propriété étude de cas au Sénégal. *Revue Tiers Monde*, 837–851.

Pluess, T., Jarošík, V., Pyšek, P., Cannon, R., Pergl, J., Breukers, A., & Bacher, S. (2012). Which Factors Affect the Success or Failure of Eradication Campaigns against Alien Species? *PLoS ONE*, *7*(10), e48157. https://doi.org/10.1371/journal.pone.0048157

Poirier, S. (2010). Change, Resistance, Accommodation and Engagement in Indigenous Contexts: A Comparative (Canada–Australia) Perspective. *Anthropological Forum*, *20*(1), 41–60. https://doi.org/10.1080/00664670903524202

Pollin, R., Garrett-Peltier, H., Heintz, J., & Scharber, H. (2008). Green Recovery. Retrieved from http://www.peri.umass.edu/fileadmin/pdf/other_publication_types/green_economics/peri_report.pdf

Polyakov, M., Fogarty, J., Zhang, F., Pandit, R., & Pannell, D. J. (2017). The value of restoring urban drains to living streams. *Water Resources and Economics*, *17*, 42–55. https://doi.org/10.1016/j.wre.2016.03.002

Ponisio, L. C., M'Gonigle, L. K., Mace, K. C., Palomino, J., de Valpine, P., & Kremen, C. (2015). Diversification practices reduce organic to conventional yield gap. *Proceedings. Biological Sciences*, *282*(1799), 20141396. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/25621333

Poorter, L., Bongers, F., Aide, T. M., Almeyda Zambrano, A. M., Balvanera, P., Becknell, J. M., Boukili, V., Brancalion, P. H. S., Broadbent, E. N., Chazdon, R. L., Craven, D., de Almeida-Cortez, J. S., Cabral, G. A. L., de Jong, B. H. J., Denslow, J. S., Dent, D. H., DeWalt, S. J., Dupuy, J. M., Durán, S. M., Espírito-Santo, M. M., Fandino, M. C., César, R. G., Hall, J. S., Hernández-Stefanoni, J. L., Jakovac, C. C., Junqueira, A. B., Kennard, D., Letcher, S. G., Licona, J.-C., Lohbeck, M., Marín-Spiotta, E., Martínez-Ramos, M., Massoca, P., Meave, J. A., Mesquita, R., Mora, F., Muñoz, R., Muscarella, R., Nunes, Y. R.

F., Ochoa-Gaona, S., de Oliveira, A. A., Orihuela-Belmonte, E., Peña-Claros, M., Pérez-García, E. A., Piotto, D., Powers, J. S., Rodríguez-Velázquez, J., Romero-Pérez, I. E., Ruíz, J., Saldarriaga, J. G., Sanchez-Azofeifa, A., Schwartz, N. B., Steininger, M. K., Swenson, N. G., Toledo, M., Uriarte, M., van Breugel, M., van der Wal, H., Veloso, M. D. M., Vester, H. F. M., Vicentini, A., Vieira, I. C. G., Bentos, T. V., Williamson, G. B., & Rozendaal, D. M. A. (2016). Biomass resilience of Neotropical secondary forests. *Nature*, *530*(7589), 211–214. https://doi.org/10.1038/nature16512

Porras, I., Chacón-cascante, A., Barton, D. N., & Tobar, D. (2014). *Ecosystems for sale. Land prices and payments for ecosystem services in Costa Rica*. Retrieved from http://pubs.iied.org/pdfs/16577IIED.pdf?

Poudel, M., Thwaites, R., Race, D., & Dahal, G. R. (2014). REDD+ and community forestry: implications for local communities and forest management- a case study from Nepal. *International Forestry Review*, *16*(1), 39–54. https://doi.org/10.1505/146554814811031251

Powell, B., Ickowitz, A., McMullin, S., Jamnadass, R., Padoch, C., Pinedo-Vasquez, M., Sunderland, T., & Who, F. a O. and. (2013). The role of forests, trees and wild biodiversity for nutrition-sensitive food systems and landscapes. *FAO and WHO*, (January), 1–25. Retrieved from http://www.fao.org/fileadmin/user_upload/agn/pdf/2pages_Powelletal.pdf

Prach, K., Lencová, K., Řehounková, K., Dvořáková, H., Jírová, A., Konvalinková, P., Mudrák, O., Novák, J., & Trnková, R. (2013). Spontaneous vegetation succession at different central European mining sites: A comparison across seres. *Environmental Science and Pollution Research*, *20*(11), 7680–7685. https://doi.org/10.1007/s11356-013-1563-7

Prager, K., Prazan, J., & Penov, I. (2012). Soil Conservation in Transition Countries: The Role of Institutions. *Environmental Policy and Governance, 22*(1), 55-73. https://doi.org/10.1002/eet.592

Preston, C. (2004). Herbicide resistance in weeds endowed by enhanced detoxification: complications for management. *Weed Science*, *52*(3), 448–453. https://doi.org/10.1614/P2002-168B

Pretty, J. ., Morison, J. I. ., & Hine, R. . (2003). Reducing food poverty by increasing agricultural sustainability in developing countries. *Agriculture, Ecosystems & Environment*, *95*(1), 217–234. https://doi.org/10.1016/S0167-8809(02)00087-7

Pretty, J., Brett, C., Gee, D., Hine, R., Mason, C., Morison, J., Rayment, M., Van Der Bijl, G., & Dobbs, T. (2001). Policy Challenges and Priorities for Internalizing the Externalities of Modern Agriculture. *Journal of Environmental Planning and Management*, *44*(2), 263–283. https://doi.org/10.1080/09640560123782

Pretty, J., Toulmin, C., & Williams, S. (2011). Sustainable intensification in African agriculture. *International Journal of Agricultural Sustainability*, *9*(1), 5–24. https://doi.org/10.3763/ijas.2010.0583

Price Waterhouse Coopers. (2011). *The Economic Impacts of the Oil and Natural Gas Industry on the U.S. Economy in 2009: Employment, Labor Income and Value Added 2011*.

Prieur, M. (2011). Instruments internationaux et évaluation environnementale de la biodiversité : enjeux et obstacles. *Revue Juridique de L'environnement*, *5*, 7–28.

Prokop, G., Jobstman, H., & Schönbauer, A. (2011). *Report on best practices for limiting soil sealing and mitigating in affects*. Bruxelles.

Qadir, M., Quillerou, E., Nangia, V., Murtaza, G., Singh, M., Thomas, R. J., Drechsel, P., & Noble, A. D. (2014). Economics of salt-induced land degradation and restoration. *Natural Resources Forum*, *38*(4), 282–295. https://doi.org/10.1111/1477-8947.12054

Quillérou, E., & Fraser, R. (2010). Adverse Selection in the Environmental Stewardship Scheme: Does the Higher Level Stewardship Scheme Design Reduce Adverse Selection? *Journal of Agricultural*

Economics, *61*(2), 369–380. https://doi.org/10.1111/j.1477-9552.2010.00240.x

Quillérou, E., Fraser, R., & Fraser, I. (2011). Farmer Compensation and its Consequences for Environmental Benefit Provision in the Higher Level Stewardship Scheme. *Journal of Agricultural Economics*, *62*(2), 330–339. https://doi.org/10.1111/j.1477-9552.2011.00291.x

Rai, R. K., Scarborough, H., Subedi, N., & Lamichhane, B. (2012). Invasive plants - Do they devastate or diversify rural livelihoods? Rural farmers' perception of three invasive plants in Nepal. *Journal for Nature Conservation*, *20*(3), 170–176. https://doi.org/10.1016/j.jnc.2012.01.003

RAMSAR. (2012). An Integrated Framework and guidelines for avoiding , mitigating and compensating for wetland losses. In *11th Meeting of the Conference of the Parties to the Convention on Wetlands (Ramsar, Iran, 1971)* (pp. 1–30). The Ramsar Convention on Wetlands.

Rancourt, M.-È., Bellavance, F., & Goentzel, J. (2014). Market analysis and transportation procurement for food aid in Ethiopia. *Socio-Economic Planning Sciences*, *48*(3), 198–219.

Rappaport, D. I., Tambosi, L. R., & Metzger, J. P. (2015). A landscape triage approach: combining spatial and temporal dynamics to prioritize restoration and conservation. *Journal of Applied Ecology*, *52*(3), 590–601. https://doi.org/10.1111/1365-2664.12405

Recuero Virto, L., Weber, J. L., & Jeantil, M. (2018). Natural Capital Accounts and Public Policy Decisions: Findings From a Survey. *Ecological Economics*, *144*(August 2017), 244–259. https://doi.org/10.1016/j.ecolecon.2017.08.011

Redford, K. H., Padoch, C., & Sunderland, T. (2013). Fads, Funding, and Forgetting in Three Decades of Conservation. *Conservation Biology*, *27*(3), 437–438. https://doi.org/10.1111/cobi.12071

Reed, M. S., Buenemann, M., Atlhopheng, J., Akhtar-Schuster, M., Bachmann, F., Bastin, G., Bigas, H., Chanda, R., Dougill, A. J., Essahli, W., Evely, A. C., Fleskens, L., Geeson, N., Glass, J. H., Hessel, R., Holden, J., Ioris, A. A. R., Kruger, B., Liniger, H. P., Mphinyane, W., Nainggolan, D., Perkins, J., Raymond, C. M., Ritsema, C. J., Schwilch, G., Sebego, R., Seely, M., Stringer, L. C., Thomas, R., Twomlow, S., & Verzandvoort, S. (2011). Cross-scale monitoring and assessment of land degradation and sustainable land management: A methodological framework for knowledge management. *Land Degradation & Development*, *22*(2), 261–271. https://doi.org/10.1002/ldr.1087

Reed, M. S., & Dougill, A. J. (2008). *Participatory Land Degradation Assessment*. In C. Lee & T. Schaaf, (Eds.), *The Future of Drylands* (pp. 719-729). Dordrecht: Springer.

Reed, M. S., & Stringer, L. (2015). Climate change and desertification: Anticipating, assessing & adapting to future change in drylands- Impulse Report. In *3rd UNCCD Scientific Conference*. Cancun, Mexico: Agropolis International and Groupe CCEE. Retrieved from http://www.unccd.int/en/programmes/Science/Conferences/Documents/3sc_unccd_impulse-report.pdf

Reed, M. S., Stringer, L. C., Dougill, A. J., Perkins, J. S., Atlhopheng, J. R., Mulale, K., & Favretto, N. (2015). Reorienting land degradation towards sustainable land management: Linking sustainable livelihoods with ecosystem services in rangeland systems. *Journal of Environmental Management*, *151*, 472–485. https://doi.org/10.1016/j.jenvman.2014.11.010

Reeson, A. F., & Tisdell, J. G. (2008). Institutions, motivations and public goods: An experimental test of motivational crowding. *Journal of Economic Behavior and Organization*, *68*(1), 273–281. https://doi.org/10.1016/j.jebo.2008.04.002

Reicosky, D. C. (2015). Conservation tillage is not conservation agriculture. *Journal of Soil and Water Conservation*, *70*(5), 103A–108A. https://doi.org/10.2489/jswc.70.5.103A

Reid, H., & Swiderska, K. (2008). *Biodiversity, climate change and poverty : exploring the links*. International Institute for Environment and Development. Retrieved from http://pubs.iied.org/pdfs/17034IIED.pdf

Reij, C., Gray Tappan, & Smale, M. (2009). *Agroenvironmental Transformation in the Sahel*. IFPRI Discussion Paper 00914. International Food Policy Research Institute. Retrieved from http://cdm15738.contentdm.oclc.org/utils/getfile/collection/p15738coll2/id/15847/filename/15848.pdf

Reij, C. P., & Smaling, E. M. A. (2008). Analyzing successes in agriculture and land management in Sub-Saharan Africa: Is macro-level gloom obscuring positive micro-level change? *Land Use Policy*, *25*(3), 410–420. https://doi.org/10.1016/j.landusepol.2007.10.001

Reij, C., Tappan, G., & Belemvire, A. (2005). Changing land management practices and vegetation on the Central Plateau of Burkina Faso (1968-2002). *Journal of Arid Environments*, *63*(3), 642–659. https://doi.org/10.1016/j.jaridenv.2005.03.010

Reinikainen, J., Sorvari, J., & Tikkanen, S. (2016). Finnish policy approach and measures for the promotion of sustainability in contaminated land management. *Journal of Environmental Management*, *184*, 108–119. https://doi.org/10.1016/j.jenvman.2016.08.046

Rejmánek, M., & Pitcairn, M. J. (2002). When is eradication of exotic pest plants a realistic goal? In C. R. Vietch & M. N. Clout (Eds.) *Turning the Tide: The Eradication of Invasive Species* (pp. 249–253). Gland: International Union for the Conservation of Nature.

Remme, R., Edens, B., Schröter, M., & Hein, L. (2015). Ecosystem accounting aims to provide a better understanding of ecosystem contributions to the economy in a spatially explicit way. *Ecological Economics*, *112*, 116–128.

Remme, R., Schröter, M., & Hein, L. (2014). Developing spatial biophysical accounting for multiple ecosystem services. *Ecosystem Services*, *10*, 6–18.

Renaud, F. G., Nehren, U., Sudmeier-Rieux, K., & Estrella, M. (2016). Developments and Opportunities for Ecosystem-Based Disaster Risk Reduction and Climate Change Adaptation. In F. Renaud (Ed.), *Ecosystem-Based Disaster Risk Reduction and Adaptation in Practice* (pp. 1–20). https://doi.org/10.1007/978-3-319-43633-3_1

Requier-Desjardins, M., Adhikari, B., & Sperlich, S. (2011). Some notes on the economic assessment of land degradation. *Land Degradation & Development*, *22*(2), 285–298. https://doi.org/10.1002/ldr.1056

Ribot, J. C., & Larson, A. M. (2005). *Democratic Decentralization Through a Natural Resource Lens: Cases from Africa, Asia and Latin America*. Oxon, UK: Routledge.

Richardson, D.M., van Wilgen, B. W. (2004). Invasive alien plants in South Africa : how well do we understand the ecological impacts? : working for water. *South African Journal of Science, 100*. South African Association for the Advancement of Science. Retrieved from https://journals.co.za/content/sajsci/100/1-2/EJC96214

Rights and Resources Initiative. (2015). *Who Owns the World's Land? A global baseline of formally recognized indigenous and community land rights*. Washington D.C., USA: Rights and Resources Initiative.

Robbins, P. (2012). *Political Ecology: A Critical Introduction.* (2nd ed.). Chichester: Wiley.

Robertson, M. M. (2004). The neoliberalization of ecosystem services: wetland mitigation banking and problems in environmental governance. *Geoforum*, *35*(3), 361–373. https://doi.org/10.1016/J.GEOFORUM.2003.06.002

Robertson, M. M. (2006). The nature that capital can see: science, state, and market in the commodification of ecosystem services. *Environment and Planning D: Society and Space, 24*(3), 367-387.

Robinson, D, A., Panagos, P., Borrelli, P., Jones, A., Montanarella, L., Tye, A., & Obst, C. G. (2017). Soil Natural Capital in Europe; A Framework for State and Change Assessment. *Scientific Reports*, *7*(6706).

Robinson, D. A., Fraser, I., Dominati, E. J., Davidsdottir, B., Jonsson, J. O. G., Jones, L., Jones, S. B., Tuller, M., Lebron, I., Bristow, K. L., Souza, D. M., Banwart, S., & Clothier, B. (2014). On the value of soil resources in the context of natural capital and ecosystem service delivery. *Soil Sci. Soc. Am. J.*, *78*, 685–700.

Rodrigues, R. R., Gandolfi, S., Nave, A. G., Aronson, J., Barreto, T. E., Vidal, C. Y., & Brancalion, P. H. S. (2011). Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. *Forest Ecology and Management*, *261*(10), 1605–1613. https://doi.org/10.1016/j.foreco.2010.07.005

Roger-Estrade, J., Anger, C., Bertrand, M., & Richard, G. (2010, December 1). Tillage and soil ecology: Partners for sustainable agriculture. *Soil and Tillage Research, 111*(1), 33-40. https://doi.org/10.1016/j.still.2010.08.010

Rokia, S., Séré, G., Schwartz, C., Deeb, M., Fournier, F., Nehls, T., Damas, O., & Vidal-Beaudet, L. (2014). Modelling agronomic properties of Technosols constructed with urban wastes. *Waste Management*, *34*(11), 2155–2162. https://doi.org/10.1016/j.wasman.2013.12.016

Roy, S., Byrne, J., & Pickering, C. (2012). A systematic quantitative review of urban tree benefits, costs, and assessment methods across cities in different climatic zones. *Urban Forestry and Urban Greening*, *11*(4), 351–363. https://doi.org/10.1016/j.ufug.2012.06.006

Ruckelshaus, M., McKenzie, E., Tallis, H., Guerry, A., Daily, G., Kareiva, P., Polasky, S., Ricketts, T., Bhagabati, N., Wood, S. A., & Bernhardt, J. (2015). Notes from the field: Lessons learned from using ecosystem service: approaches to inform real-world decisions. *Ecological Economics*, *115*, 11–21.

Ruhl, J. B., & Salzman, J. E. (2006). The Effects of Wetland Mitigation Banking on People. *National Wetlands Newsletter*, *28*(2). Retrieved from https://papers.ssrn.com/sol3/papers.cfm?abstract_id=878331

Russell-Smith, J., Yates, C., Edwards, A., Allan, G. E., Cook, G. D., Cooke, P., Craig, R., Heath, B., & Smith, R. (2003). Contemporary fire regimes of northern Australia, 1997–2001: change since Aboriginal occupancy, challenges for sustainable management. *International Journal of Wildland Fire*, *12*(4), 283. https://doi.org/10.1071/WF03015

Russi, D., Margue, H., Oppermann, R., & Keenleyside, C. (2016). Result-based agri-environment measures: Market-based instruments, incentives or rewards? The case of Baden-Württemberg. *Land Use Policy*, *54*, 69–77. https://doi.org/10.1016/j.landusepol.2016.01.012

Russi, D., Ten Brink, P., Farmer, A., & Badura, T. (2013). *The Economics of Ecosystems and Biodiversity for Water and Wetlands*. London, Brussels, Gland . Retrieved from www.ieep.eu.

Saez de Bikuña, K., Hauschild, M. Z., Pilegaard, K., & Ibrom, A. (2017). Environmental performance of gasified willow from different lands including land-use changes. *GCB Bioenergy*, *9*(4), 756–769. https://doi.org/10.1111/gcbb.12378

Saito, Y., Yang, Z., & Hori, K. (2001). The Huanghe (Yellow River) and Changjiang (Yangtze River) deltas: A review on their characteristics, evolution and sediment discharge during the Holocene. *Geomorphology*, *41*(2), 219–231. https://doi.org/10.1016/S0169-555X(01)00118-0

Salton, J. C., Mercante, F. M., Tomazi, M., Zanatta, J. A., Concenço, G., Silva, W. M., & Retore, M. (2014). Integrated crop-livestock system in tropical Brazil: Toward a sustainable production system. *Agriculture, Ecosystems and Environment*, *190*(January 2013), 70–79. https://doi.org/10.1016/j.agee.2013.09.023

Sato, C. F., & Lindenmayer, D. B. (2017). Meeting the Global Ecosystem Collapse Challenge. *Conservation*

Letters, *11*(1), e12348. https://doi.org/10.1111/conl.12348

Sattler, C., & Matzdorf, B. (2013). Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries. *Ecosystem Services*, *6*, 1–132.

Sattler, C., & Nagel, U. J. (2010). Factors affecting farmers' acceptance of conservation measures—A case study from north-eastern Germany. *Land Use Policy*, *27*(1), 70–77. https://doi.org/10.1016/j.landusepol.2008.02.002

Saxena, N. C. (1997). *The saga of participatory forest management in India*. CIFOR Special Publication. Center for International Forestry Research. Retrieved from http://www.cifor.org/publications/pdf_files/Books/SP-Saga.pdf

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., Venter, M., Boedhihartono, A. K., Day, M., Garcia, C., van Oosten, C., & Buck, L. E. (2013). Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proceedings of the National Academy of Sciences*, *110*(21), 8349–8356. https://doi.org/10.1073/pnas.1210595110

Scherr, S. J. (2000). A downward spiral? Research evidence on the relationship between poverty and natural resource degradation. *Food Policy*, *25*(4), 479–498.

Scherr, S. J., Shames, S., Gross, L., Borges, M. A., Bos, G., & Brasser, A. (2017). *Business for Sustainable Landscapes: An action agenda for sustainable development*. Washington, D.C.: EcoAgriculture Partners and IUCN, on behalf of the Landscapes for People, Food and Nature Initiative.Retrieved from http://peoplefoodandnature.org/wp-content/uploads/2017/05/Business-for-Sustainable-Landscapes-An-Action-Agenda-for-Sustainable-Development-May-2017.pdf

Schlaepfer, M. A., Sherman, P. W., Blossey, B., & Runge, M. C. (2005). Introduced species as evolutionary traps. *Ecology Letters*, *8*(3), 241–246. https://doi.org/10.1111/j.1461-0248.2005.00730.x

Schwartzman, S., Boas, A. V., Ono, K. Y., Fonseca, M. G., Doblas, J., Zimmerman, B., Junqueira, P., Jerozolimski, A., Salazar, M., Junqueira, R. P., & Torres, M. (2013). The natural and social history of the indigenous lands and protected areas corridor of the Xingu River basin. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *368*(1619), 20120164. Retrieved from http://rstb.royalsocietypublishing.org/content/368/1619/20120164.short

Šebelíková, L., Řehounková, K., & Prach, K. (2016). Spontaneous revegetation vs. forestry reclamation in post-mining sand pits. *Environmental Science and Pollution Research*, *23*(14), 13598–13605. https://doi.org/10.1007/s11356-015-5330-9

Seifan, M. (2009). Long-term effects of anthropogenic activities on semi-arid sand dunes. *Journal of Arid Environments*, *73*(3), 332–337. https://doi.org/10.1016/j.jaridenv.2008.10.009

Senay, G. B., & Verdin, J. (2003). Characterization of Yield Reduction in Ethiopia Using a GIS-Based Crop Water Balance Model. *Canadian Journal of Remote Sensing*, *29*(6), 687–692.

SERI, S. for E. R. I. S. & P. W. G. (2004). *The SER International Primer on Ecological Restoration*. Tucson. Retrieved from http://c.ymcdn.com/sites/www.ser.org/resource/resmgr/custompages/publications/SER_Primer/ser_primer.pdf?hhSearchTerms=%22ser+and+international+and+primer%22

Seufert, P. (2013). The FAO Voluntary Guidelines on the Responsible Governance of Tenure of Land, Fisheries and Forests,. *Globalizations, 10*(1), 181–186.

Sewell, A., Bouma, J., & Van Der Esch, S. (2016). *Investigating the challenges and opportunities for scaling up Ecosystem Restoration*. The Hague: PBL Netherlands Envionmental Assessment Agency. Retrieved from http://www.pbl.nl/sites/default/files/cms/publicaties/pbl-2016-investigating-the-challenges-and-opportunities-for-scaling-up-ecosystem-restoration_2356.pdf

Shabbir, A., Dhileepan, K., O'Donnell, C., & Adkins, S. W. (2013). Complementing biological control with plant suppression: Implications for improved management of parthenium weed (Parthenium hysterophorus L.). *Biological Control*, *64*(3), 270–275. https://doi.org/10.1016/j.biocontrol.2012.11.014

Shackleton, R. T., Le Maitre, D. C., Pasiecznik, N. M., & Richardson, D. M. (2014). Prosopis: a global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. *AoB PLANTS*, *6*(0), plu027-plu027. https://doi.org/10.1093/aobpla/plu027

Shackleton, S., Kirby, D., & Gambiza, J. (2011). Invasive plants – friends or foes? Contribution of prickly pear (Opuntia ficus-indica) to livelihoods in Makana Municipality, Eastern Cape, South Africa. *Development Southern Africa*, *28*(2), 177–193. Journal Article. Retrieved from http://www.tandfonline.com/doi/abs/10.1080/0376835X.2011.570065

Shames, S., Hill-Clarvis, M., & Kissinger, G. (2014). Financing strategies for integrated landscape investment: synthesis report. In S. Shames (Ed.) *Financing Strategies for Integrated Landscape Investment*. Washington, DC: EcoAgriculture Partners, on behalf of the Landscapes for People, Food and Nature Initiative.

Sheley, R. L., Jacobs, J. S., Carpinelli, M. F., & Technology, W. (1998). Distribution , Biology , and Management of Diffuse Knapweed (Centaurea diffusa) and Spotted Knapweed (Centaurea maculosa). *Weed Technology*, *12*(2), 353–362. https://doi.org/10.1007/s10021-001-0101-5

Shen, J. (2015). *Study on The Impact of Urbanization on Stream Structure, River Network Connectivity And Storage Capacity in Pudong New Area, Shanghai*. East China Normal University.

Shepherd, C. J. (2010). Mobilizing Local Knowledge and Asserting Culture The Cultural Politics of In Situ Conservation of Agricultural Biodiversity. *Current Anthropology*, *51*(5), 629–654. https://doi.org/10.1086/656424

Shi, Y. (2002). Charcteristics of late Quaternary monsoonal glaciations on the Tibetan plateau and in East Asia. *Quaternary International*, *97–98*, 79–91.

Shiferaw, B., & Holden, S. T. (2000). Policy instruments for sustainable land management: the case of highland smallholders in Ethiopia. *Agricultural Economics*, *22*(3), 217–232. https://doi.org/10.1111/j.1574-0862.2000.tb00071.x

Shiferaw, B., Okello, J., & Reddy, R. V. (2009). Adoption and adaptation of natural resource management innovations in smallholder agriculture: reflections on key lessons and best practices. *Environment, Development and Sustainability*, *11*(3), 601–619.

Shiva, V. (1991). *The Violence of the Green Revolution: Third World Agriculture, Ecology, and Politics*. London: Zed Books, New York: Room 400, Penang: Third World Network.

Shropshire, R., & Wagner, B. (2009). *An Estimation of Montana's Restoration Economy*. Montana: Department of Labor and Industry. Retrieved from https://deq.mt.gov/Portals/112/Land/FedSuperFund/Documents/sst/RestorationEconomyRPT9-17-09.pdf

Sidibé, Y., Myint, M., & Westerberg, V. (2014). *An economic valuation of agroforestry and land restoration in the Kelka Forest, Mali. Assessing the socio-economic and environmental dimensions of land degradation*. Nairobi: Economics of Land Degradation Initiative and International Union for the Conservation of Nature. Retrieved from www.eld-initiative.org

Siebielec, G., Lazar, S., Kaufmann, C., & Jaensch, S. (2010). *Handbook for measures enhancing soil function performance and compensating soil loss during urbanization process URBAN SMS Soil Management Strategy*. Retrieved from http://www.umweltbundesamt.at/fileadmin/inhalte/urbansms/pdf_files/final_results/16_Handbook_measures_enhancing_soil_function_performance.pdf

Sietz, D., & Van Dijk, H. (2015). Land-based adaptation to global change: What drives soil and water conservation in western Africa? *Global Environmental Change*, *33*, 131–141. https://doi.org/10.1016/j.gloenvcha.2015.05.001

Simberloff, D. (2009). We can eliminate invasions or live with them. Successful management projects. *Biological Invasions*, *11*(1), 149–157. https://doi.org/10.1007/s10530-008-9317-z

Singh, A. (2015). Land and water management planning for increasing farm income in irrigated dry areas. *Land Use Policy*, *42*, 244–250. https://doi.org/10.1016/j.landusepol.2014.08.006

Singh, A. (2016). Evaluating the effect of different management policies on the long-term sustainability of irrigated agriculture. *Land Use Policy*, *54*, 499–507. https://doi.org/10.1016/j.landusepol.2016.03.012

Sirina, N., Rohmer, S., & Fortyguina, E. (2013). Environmental Impact Assessment: Challenges in Use of Methods and Tools in France and in Russia. *Key Engineering Materials*, *572*, 40–43. https://doi.org/10.4028/www.scientific.net/KEM.572.40

Skirycz, A., Castilho, A., Chaparro, C., Carvalho, N., George, T., & Siqueira, J. O. (2014). Canga biodiversity , a matter of mining. *Front. Plant Sci*, *5*(653), 1–9. https://doi.org/10.3389/fpls.2014.00653

Sklenicka, P., Molnarova, K. J., Salek, M., Simova, P., Vlasak, J., Sekac, P., & Janovska, V. (2015). Owner or tenant: Who adopts better soil conservation practices? *Land Use Policy*, *47*, 253–261. https://doi.org/10.1016/j.landusepol.2015.04.017

Smith, L. M., Effland, W. R., Behrman, K. D., & Johnson, M.-V. V. (2015). Assessing the Effects of USDA Conservation Programs on Ecosystem Services Provided by Wetlands. *National Wetlands Newsletter*, *37*(5), 10–13.

Spake, R., Ezard, T. H. G., Martin, P. A., Newton, A. C., & Doncaster, C. P. (2015). A meta-analysis of functional group responses to forest recovery outside of the tropics. *Conservation Biology*, *29*(6), 1695–1703. https://doi.org/10.1111/cobi.12548

Spargo, A., & Doley, D. (2016). Selective coal mine overburden treatment with topsoil and compost to optimise pasture or native vegetation establishment. *J Environ Manage.*, *1*(152), 342–350.

Spash, C. L. (2015). Bulldozing biodiversity: The economics of offsets and trading-in Nature. *Biological Conservation*, *192*, 541–551. https://doi.org/10.1016/J.BIOCON.2015.07.037

Sri Lanka National Invasive Alien Species Committee. (2015). *National Strategy and Action Plan for Invasive Alien Species Control in Sri Lanka*. Colombo, Sri Lanka.

Stanturf, J. A., Palik, B. J., & Dumroese, R. K. (2014). Contemporary forest restoration: A review emphasizing function. *Forest Ecology and Management*, *331*, 292–323. https://doi.org/10.1016/j.foreco.2014.07.029

Steichen, P. (2010). La prévention et la réparation de la pollution des sites; le cadre juridique communautaire. In Litec LexisNexis (Ed.), *ites et sols pollués: enjeux d'un droit, droit en jeu(x)* (pp. 123–140).

Stephens, S. L., Agee, J. K., Fulé, P. Z., North, M. P., Romme, W. H., Swetnam, T. W., & Turner, M. G. (2013). Managing Forests and Fire in Changing Climates. *Science*, *342*(6154), 41–42. https://doi.org/10.1126/science.1240294

STK4SD, T. S. and T. K. for S. D. C. (2015). Climate change and land degradation: Bridging knowledge and stakeholders. In *UNCCD 3rd scientific conference*. Cancun: UNCCD. Retrieved from http://3sc.unccd.int/documents-outputs/outputs/climate-change-and-land-degradation-bridging-knowledge-and-stakeholders-outcomes-from-the-unccd-3rd-scientific-conference

Stockholm International, & Water Institute. (2010). Eutrophication Dynamics of Lakes and Reservoirs in

China: Nutrient Management Strategy. In *Responding to Global Changes: The Water Quality Challenge – Prevention, Wise Use and Abatement*. Stockholm: World Water Week. Retrieved from http://www.worldwaterweek.org/documents/WWW_PDF/2010/Final_programme_2010_web.pdf

SRC. (2016). *Mobilizing Traditional Knowledge, Innovations and Practices in rotational farming for sustainable development*. Stockholm: Stockholm Resilience Centre. Retrieved from http://swed.bio/reports/report/mobilizing-traditional-knowledge-innovations-and-practices-in-rotational-farming-for-sustainable-development/

SRC. (2016). *Mobilizing indigenous knowledge, innovations and practices of the Kalanguyas farming systems in Tinoc, Ifuago, the Philippines.* Stockholm: Stockholm Resilience Centre. Retrieved from http://swed.bio/reports/report/mobilizing-indigenous-knowledge-innovations-and-practices-of-the-kalanguyas-farming-systems-in-tinoc-ifuago-the-philippines/

Stoneham, G., Chaudhri, V., Ha, A., & Strappazzon, L. (2003). Auctions for conservation contracts: an empirical examination of Victoria's BushTender trial. *The Australian Journal of Agricultural and Resource Economics*, *47*(4), 477–500. https://doi.org/10.1111/j.1467-8489.2003.t01-1-00224.x

Strassburg, B. B. N., Latawiec, A. E., Barioni, L. G., Nobre, C. A., da Silva, V. P., Valentim, J. F., Vianna, M., & Assad, E. D. (2014). When enough should be enough: Improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. *Global Environmental Change*, *28*(1), 84–97. https://doi.org/10.1016/j.gloenvcha.2014.06.001

Suárez, A., Williams-Linera, G., Trejo, C., Valdez-Hernández, J. I., Cetina-Alcalá, V. M., & Vibrans, H. (2012). Local knowledge helps select species for forest restoration in a tropical dry forest of central Veracruz, Mexico. *Agroforestry Systems*, *85*(1), 35–55. https://doi.org/10.1007/s10457-011-9437-9

Sullivan, S. (2014). *The natural capital myth; or will accounting save the world?* LCSV Working Paper Series No 3. Manchester : The Leverhulme Centre for the Study of Value. Retrieved from http://thestudyofvalue.org/wp-content/uploads/2013/11/WP3-Sullivan-2014-Natural-Capital-Myth.pdf

Sumarga, E., Hein, L., Edens, B., & Suwarno, A. (2015). Mapping monetary values of ecosystem services in support of developing ecosystem accounts. *Ecosystem Services*, *12*, 71–83.

Sureshwaran, S., Londhe, S., & Frazier, P. (1996). A logit model for evaluating farmer participation in soil conservation programs: Sloping agricultural land technology on upland farms in the Philippines. *Journal of Sustainable Agriculture, 7*(4), 57-69. https://doi.org/10.1300/J064v07n04_07

Szulczewska, B., Giedych, R., Borowski, J., Kuchcik, M., Sikorski, P., Mazurkiewicz, A., & Stańczyk, T. (2014). How much green is needed for a vital neighbourhood? In search for empirical evidence. *Land Use Policy*, *38*, 330–345. https://doi.org/10.1016/j.landusepol.2013.11.006

Tacconi, L. (2012). Redefining payments for environmental services. *Ecological Economics*, *73*, 29–36. https://doi.org/10.1016/j.ecolecon.2011.09.028

Tacio, H. (1993). Sloping agricultural land technology (SALT): a sustainable agroforestry scheme for the uplands. *Agroforestry Systems, 22*(2), 145-152.

Taddese, G. (2001). Land degradation: A challenge to Ethiopia. *Environmental Management*, *27*(6), 815–824. https://doi.org/10.1007/s002670010190

Tambosi, L. R., Martensen, A. C., Ribeiro, M. C., & Metzger, J. P. (2014). A Framework to Optimize Biodiversity Restoration Efforts Based on Habitat Amount and Landscape Connectivity. *Restoration Ecology*, *22*(2), 169–177. https://doi.org/10.1111/rec.12049

TEEB. (2012). *The Economics of Ecosystems & Biodiversity*. Retrieved from http://ec.europa.eu/environment/nature/biodiversity/economics/.

Tempel, D. J., Gutiérrez, R. J., Battles, J. J., Fry, D. L., Su, Y., Guo, Q., Reetz, M. J., Whitmore, S. A., Jones, G.

M., Collins, B. M., Stephens, S. L., Kelly, M., Berigan, W., & Peery, M. Z. (2015). Evaluating short- and long-term impacts of fuels treatments and wildfire on an old-forest species. *Ecosphere*, *6*(April), 1–19. https://doi.org/10.1890/ES15-00234.1

Ten Kate, K., Bishop, J., & Bayon, R. (2004). *Biodiversity offsets: Views, experience, and the business case*. IUCN, Gland, Switzerland and Cambridge, UK and Insight Investment, London, UK. Retrieved from http://biodiversityeconomics.org/offsets/execsum.htm

The Natural Resource Management Council. (2010). *Australia's Strategy for the National Reserve System 2009-2030*.

The World Bank. (2007). *Project Performance Assessment Report for Second Loess Plateau Watershed Rehabilitation Project and Xiaolangdi Multipurpose Project I & II and Tarim Basin II Project. (Report No. 41122).* Washington, DS. Retrieved from http://documents.worldbank.org/curated/en/812911468218977436/pdf/41122.pdf

Thoren, R. I., Atwater, J., & Berube, P. (2012). A model for analyzing water reuse and resource recovery potential in urban areas. *Canadian Journal of Civil Engineering*, *39*(11), 1202–1209. https://doi.org/10.1139/l2012-109

Tischew, S., & Kirmer, A. (2007). Implementation of basic studies in the ecological restoration of surface-mined land. *Restoration Ecology*, *15*(2), 321–325. https://doi.org/10.1111/j.1526-100X.2007.00217.x

Tittonell, P. (2014). Ecological intensification of agriculture—sustainable by nature. *Curr. Opinion Environ. Sustain.*, 8, 53–61.

Toth, G. (2010). *Agri-environmental soil quality indicator in the European Perspective*. Ispra, Italy: European Commission, Joint Research Centre. Retrieved from http://www.oecd.org/tad/sustainable-agriculture/44808284.pdf

Tougiani, A., Guero, C., & Rinaudo, T. (2009). Community mobilisation for improved livelihoods through tree crop management in Niger. *GeoJournal*, *74*(5), 377–389. https://doi.org/10.1007/s10708-008-9228-7

Travis, M. J., Wiel-Shafran, A., Weisbrod, N., Adar, E., & Gross, A. (2010). Greywater reuse for irrigation: Effect on soil properties. *Science of the Total Environment*, *408*(12), 2501–2508. https://doi.org/10.1016/j.scitotenv.2010.03.005

Tropek, R., Kadlec, T., Karesova, P., Spitzer, L., Kocarek, P., Malenovsky, P., Banar, P., Tuf, I. H., Hejda, M., & Konvicka, M. (2010). Spontaneous succession in limestone quarries as an effective restoration tool for endangered arthropods and plants. *Journal of Applied Ecology*, *47*, 139–147.

Trucost. (2013). *Natural Capital at Risk. The Top 100 Externalities of Business.* London. Retrieved from https://www.trucost.com/publication/natural-capital-risk-top-100-externalities-business/

Tschakert, P. (2007). Views from the vulnerable: Understanding climatic and other stressors in the Sahel. *Global Environmental Change, 17*(3-4), 381-396. https://doi.org/10.1016/j.gloenvcha.2006.11.008

Tsunekawa, A., Liu, G., Yamanaka, N., & Du, S. (2014). *Restoration and Development of the Degraded Loess Plateau, China.* Tokyo: Springer. https://doi.org/10.1007/978-4-431-54481-4

Tully, K., Sullivan, C., Weil, R., & Sanchez, P. (2015). The State of soil degradation in sub-Saharan Africa: Baselines, trajectories, and solutions. *Sustainability*, *7*(6), 6523–6552. https://doi.org/10.3390/su7066523

Turbé, A., De Toni, A., Benito, P., Lavelle, P. P., Ruiz, N., Van der Putten, W. H., Labouze, E., & Mudgal, S. S. (2010). *Soil biodiversity: functions, threats and tools for policy makers*. Bio Intelligence Service, IRD, and NIOO, Report for European Commission (DG Environment). Retrieved from http://ec.europa.eu/environment/archives/soil/pdf/biodiversity_report.pdf

Turnhout, E., Bloomfield, B., Hulme, M., Vogel, J., & Wynne, B. (2012). Listen to the voices of experience. *Nature*, *488*(7412), 454–455. https://doi.org/10.1038/488454a

Turnhout, E., Waterton, C., Neves, K., & Buizer, M. (2013). Rethinking biodiversity: from goods and services to "living with". *Conservation Letters*, *6*(3), 154–161. https://doi.org/10.1111/j.1755-263X.2012.00307.x

UK NEA. (2014). *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. UK: UNEP-WCMC, LWEC. Retrieved from http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

UN. (2014). *System of Environmental-Economic Accounting 2012 – Central Framework*. The World Bank, United Nations, European Commission, Food and Agricultural Organization of the United Nations, International Monetary Fund, Organisation for Economic Co-operation and Development. New York, USA: United Nations.

UN Water. (2015). *United Nations World Water Development Report (WWDR) 2015: Water for a Sustainable World*. Paris: United Nations Educational, Scientific and Cultural Organization, Retrieved from https://sustainabledevelopment.un.org/content/documents/1711Water%20for%20a%20Sustainable%20World.pdf

UNCCD. (2012). *Zero Net Land Degradation*. Bonn: United Nations Convention to Combat Desertification. Retrieved from http://www.unccd.int/Lists/SiteDocumentLibrary/Rio+20/UNCCD_PolicyBrief_ZeroNetLandDegradation.pdf

UNCCD. (2014). *Land degradation neutrality. Resilience at local, national and regional levels.* Bonn: United Nations Convention to Combat Desertification. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.729.8110&rep=rep1&type=pdf

Undersander, D., Albert, B., Cosgrove, D., Johnson, D., & Peterson, P. (2014). Pastures for profit: A guide to rotational grazing, *A3529*, 1–39. Retrieved from https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1097378.pdf

UNECE/FAO. (2016). *Forest Products Annual Market Review, 2015-2016*. UNITED NATIONS PUBLICATIONS. Retrieved from http://www.unece.org/fileadmin/DAM/timber/publications/fpamr2016.pdf

UNEP. (2014). *Assessing Global Land Use: Balancing Consumption with Sustainable Supply. A Report of the Working Group on Land and Soils of the International Resource Panel*. S. Bringezu, H. Schütz, W. Pengue, M. O´Brien, F. Garcia, R. Sims, R. Howarth, L. Kauppi, M. Swilling, & J. Herrick (Eds.). United Nations Environment Programme. Retrieved from http://www.resourcepanel.org/file/193/download?token=TwLqal30

UNEP-WCMC. (2016a). *Exploring Approaches for Constructing Species Accounts in the Context of SEEA-EEA*. Cambridge: UNEP World Conservation Monitoring Centre. Retrieved from http://naturalcapitalcoalition.org/wp-content/uploads/2016/08/Exploring_Approaches_for_constructing_Species_Accounts_in_the_context_of_the_SEEA-EEA_FINAL-1.pdf

UNEP-WCMC. (2016b). *Feasibility Study for Biodiversity Accounting in Uganda*. Cambridge: UNEP World Conservation Monitoring Centre. Retrieved from https://www.unep-wcmc.org/system/comfy/cms/files/files/000/000/804/original/Feasibility_Study_for_Biodiversity_Accounting_in_Uganda_FINAL.pdf

UNDESA. (2017). *Framework for the Development of Environment Statistics (FDES 2013)*. New York: United Nations, Department of Economic and Social Affairs (UNDESA), Statistics Division. Retrieved from https://unstats.un.org/unsd/environment/FDES/FDES-2015-supporting-tools/FDES.pdf

Uprety, Y., Asselin, H., Dhakal, A., & Julien, N. (2012). Traditional use of medicinal plants in the boreal forest of Canada: review and perspectives. *Journal of Ethnobiology and Ethnomedicine*, *8*(1), 7. https://doi.org/10.1186/1746-4269-8-7

Urzedo, D. I., Vidal, E., Sills, E. O., Pina-Rodrigues, F. C. M., & Junqueira, R. G. P. (2016). Tropical forest seeds in the household economy: effects of market participation among three sociocultural groups in the Upper Xingu region of the Brazilian Amazon. *Environmental Conservation*, *43*(1), 13–23. https://doi.org/10.1017/s0376892915000247

USAID. (2008). *USAID'S Biodiversity Conservation and Foretry Programs, FY 2007*. Retrieved from http://redd.unfccc.int/uploads/2_113_redd_20081107_us.pdf

USDA Farm Services Agency. (2011). The Environmental Benefits of the Conservation Reserve Program (CRP), 2–3.

Valatin, G., & Price, C. (2014). How Cost-Effective Is Forestry for Climate Change Mitigation? In *Challenges and Opportunities for the World's Forests in the 21st Century* (Vol. 81, pp. 297–339). https://doi.org/10.1007/978-94-007-7076-8

Valentin, C., Agus, F., Alamban, R., Boosaner, A., Bricquet, J. P., Chaplot, V., de Guzman, T., de Rouw, A., Janeau, J. L., Orange, D., Phachomphonh, K., Do Duy Phai, Podwojewski, P., Ribolzi, O., Silvera, N., Subagyono, K., Thiébaux, J. P., Tran Duc Toan, & Vadari, T. (2008). Runoff and sediment losses from 27 upland catchments in Southeast Asia: Impact of rapid land use changes and conservation practices. *Agriculture, Ecosystems and Environment, 128*(4), 225-238. https://doi.org/10.1016/j.agee.2008.06.004

van Calster, G. (2005). Will the EC get a finger in each pie? EC law and policy developments in soil protection and brownfields redevelopment. *Journal of Environmental Law*, *16*(1), 3–17.

van der Bij, A. U., Pawlett, M., Harris, J. A., Ritz, K., & van Diggelen, R. (2017). Soil microbial community assembly precedes vegetation development after drastic techniques to mitigate effects of nitrogen deposition. *Biological Conservation*, *212*, 476–483. https://doi.org/10.1016/J.BIOCON.2016.09.008

van Der Werf, G. R., Randerson, J. T., Giglio, L., Collatz, G. J., Mu, M., Kasibhatla, P. S., Morton, D. C., Defries, R. S., Jin, Y., & Van Leeuwen, T. T. (2010). Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997-2009). *Atmospheric Chemistry and Physics, 10*, 11707-11735. https://doi.org/10.5194/acp-10-11707-2010

van Klinken, R. D. (2012). Prosopis spp.—mesquite. In M. Julien, R. McFadyen, & J. Cullen (Eds.), *Biological control of weeds in Australia* (pp 477-485). Melbourne: CSIRO Publishing.

van Klinken, R. D., Fichera, G., & Cordo, H. (2003). Targeting biological control across diverse landscapes: the release, establishment, and early success of two insects on mesquite (Prosopis spp.) insects in Australian rangelands. *Biological Control*, *26*(1), 8–20. https://doi.org/10.1016/S1049-9644(02)00107-X

van Leenders, C., & Bor, A. (2016). *Finance for One Planet* 2016 - CoP Financial Institutions and Natural Capital. Netherlands: Netherlands Enterprise Agency. Retrieved from www.rvo.nl/CoP_FINC

van Noordwijk, M., & Leimona, B. (2010). Principles for Fairness and Efficiency in Enhancing Environmental Services in Asia: Payments, Compensation, or Co-Investment? *Ecology and Society*, *15*(4), art17. https://doi.org/10.5751/ES-03664-150417

van Oosten, C. (2013). Restoring Landscapes - Governing Place: A Learning Approach to Forest Landscape Restoration. *Journal of Sustainable Forestry, 32*(7), 659–676. https://doi.org/10.1080/10549811.2013.818551

van Oosten, C., Gunarso, P., Koesoetjahjo, I., & Wiersum, F. (2014). Governing forest landscape restoration: Cases from Indonesia. *Forests, 5*(6), 1143-1162. https://doi.org/10.3390/f5061143

van Wilgen, B. W., Forsyth, G. G., Le Maitre, D. C., Wannenburgh, A., Kotzé, J. D. F., van den Berg, E., &

Henderson, L. (2012). An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. *Biological Conservation*, *148*(1), 28–38. https://doi.org/10.1016/j.biocon.2011.12.035

van Wilgen, B. W., Marais, C., Magadlela, D., Stevens, D., & Jezile, N. (2002). Win-Win-Win: South Africa's Working for Water Programme. In S. M. Pierce, R. M. Cowling, T. Sandwith, & K. MacKinnon (Eds.), *Mainstreaming Biodiversity in Development Case Studies from South Africa* (pp. 5–20). Washington, DC: World Bank. Retrieved from http://documents.worldbank.org/curated/en/233671468763768396/Mainstreaming-biodiversity-in-development-case-studies-from-South-Africa

van Wilgen, B. W., Richardson, D. M., Le Maitre, D. C., Marais, C., & Magadlela, D. (2001). The Economic Consequences of Alien Plant Invasions: Examples of Impacts and Approaches to Sustainable Management in South Africa. *Environment, Development and Sustainability*, *3*(2), 145–168. https://doi.org/10.1023/A:1011668417953

Vandenberg, L. N., Blumberg, B., Antoniou, M. N., Benbrook, C. M., Carroll, L., Colborn, T., Everett, L. G., Hansen, M., Landrigan, P. J., Lanphear, B. P., Mesnage, R., vom Saal, F. S., Welshons, W. V, & Myers, J. P. (2017). Is it time to reassess current safety standards for glyphosate-based herbicides? *Journal of Epidemiology and Community Health*, *71*(6), 613–618. https://doi.org/10.1136/jech-2016-208463

Vandergeest, P., & Peluso, N. L. (2006). Empires of forestry: Professional forestry and state power in Southeast Asia, Part 1. *Environment and History*, *12*(1), 31–64.

Vanheusden, B. (2007). Brownfield Redevelopment in the European Union. *Boston College Environmental Affairs Law Review, 34*(3). Retrieved from http://lawdigitalcommons.bc.edu/ealr/vol34/iss3/6

Veenman, S. A. (2014). *National Policies for cleaning up contaminated sites*. Cambridge: Cambridge University Press. Retrieved from http://repository.ubn.ru.nl/handle/2066/122144

Verchot, L. V., Van Noordwijk, M., Kandji, S., Tomich, T., Ong, C., Albrecht, A., Mackensen, J., Bantilan, C., Anupama, K. V., & Palm, C. (2007). Climate change: linking adaptation and mitigation through agroforestry. *Mitigation and Adaptation Strategies for Global Change*, *12*(5), 901–918. https://doi.org/V10.1007/s11027-007-9105-6

Versfeld, D. B., Le Maitre, D. C., Chapman, R. A., South Africa. Water Research Commission., & South African Council for Scientific and Industrial Research. (1998). *Alien invading plants and water resources in South Africa : a preliminary assessment*. WRC report (South Africa. Water Research Commission), no. TT 99/98. Pretoria : The Commission.

Veum, K. S., Kremer, R. J., Sudduth, K. A., Kitchen, N. R., Lerch, R. N., Baffaut, C., Stott, D. E., Karlen, D. L., & Sadler, E. J. (2015). Conservation effects on soil quality indicators in the Missouri Salt River Basin. *Journal of Soil and Water Conservation*, *70*(4), 232–246. https://doi.org/10.2489/jswc.70.4.232

Vieira, D. L. M., Holl, K. D., & Peneireiro, F. M. (2009). Agro-Successional Restoration as a Strategy to Facilitate Tropical Forest Recovery. *Restoration Ecology*, *17*(4), 451–459. https://doi.org/10.1111/j.1526-100X.2009.00570.x

Vigilante, T., Bowman, D. M. J. S., Fisher, R., Russell-Smith, J., & Yates, C. (2004). Contemporary landscape burning patterns in the far North Kimberley region of north-west Australia: human influences and environmental determinants. *Journal of Biogeography*, *31*(8), 1317–1333. https://doi.org/10.1111/j.1365-2699.2004.01104.x

Virto, I., Imaz, M. J., Ferández-Ugalde, O., Gartzia-Bengoetxea, N., Enrique, A., & Bescansa, P. (2015). Soil Degradation and Soil Quality in Western Europe: Current Situation and Future Perspectives. *Sustainability*, *7*(1), 313–365.

Vu, Q. M., Le, Q. B., & Vlek, P. L. G. (2014). Hotspots of human-induced biomass productivity decline and their social-ecological types toward supporting national policy and local studies on combating land degradation. *Global and Planetary Change*, *121*, 64–77. https://doi.org/http://dx.doi.org/10.1016/j.gloplacha.2014.07.007

Vymazal, J. (2011). Constructed wetlands for wastewater treatment: Five decades of experience. *Environmental Science and Technology*, *45*(1), 61–69. https://doi.org/10.1021/es101403q

Waggoner, P. E., & Ausubel, J. H. (2002). A framework for sustainability science: A renovated IPAT identity. *Proceedings of the National Academy of Sciences*, *99*(12), 7860–7865. https://doi.org/10.1073/pnas.122235999

Walker, I. R., & Macdonald, G. M. (1995). Distributions of Chironomidae (Insecta : Diptera) and Other Freshwater Midges with Respect to Treeline , Northwest Territories , Canada. *INSTAAR*, *27*(3), 258–263.

Walker, L. R., & Del Moral, R. (2009). Lessons from primary succession for restoration of severely damaged habitats. *Applied Vegetation Science*, *12*(1), 55–57. https://doi.org/10.1111/j.1654-109X.2009.01002.x

Walsh, F., Christophersen, P., & Mcgregor, S. (2013). Indigenous perspectives on biodiversity. In S. Morton, A. Sheppart, & W. . Lonsdale (Eds.), *Biodiversity* (pp. 81–100). Collingwood: CSIRO Publishing. Retrieved from http://www.publish.csiro.au/ebook/download/pdf/6967#page=88

Wang, C., Yang, J., & Zhang, Q. (2006). Soil respiration in six temperate forests in China. *Global Change Biology*, *12*(11), 2103–2114. https://doi.org/10.1111/j.1365-2486.2006.01234.x

Wang, K., Deng, L., Ren, Z., Shi, W., & Chen, Y. (2016). Dynamics of ecosystem carbon stocks during vegetation restoration on the Loess Plateau of China. *Journal of Arid Land*, *8*(2), 207–220. Retrieved from http://link.springer.com/article/10.1007/s40333-015-0091-3

Wang, K., Deng, L., Ren, Z., Shi, W., Chen, Y., & Shang-Gua, Z. (2016). Dynamics of ecosystem carbon stocks during vegetation restoration on the Loess Plateau of China. *Journal of Arid Land*, *8*(2), 207–220. Retrieved from http://link.springer.com/article/10.1007/s40333-015-0091-3

Wang, L., Shao, M., Wang, Q., & Gale, W. J. (2006). Historical changes in the environment of the Chinese Loess Plateau. *Environmental Science & Policy*, *9*(7–8), 675–684. https://doi.org/10.1016/J.ENVSCI.2006.08.003

Wang, Z., Song, K., & Hu, L. (2010). China's Largest Scale Ecological Migration in the Three-River Headwater Region. *Ambio*, *39*(5–6), 443–446. https://doi.org/10.1007/s13280-010-0054-z

Warren, A. (2002). Land Degradation Is Contextual. *Land Degradation & Development, 13*(6), 449–459.

Watson, V. (2016). Locating planning in the New Urban Agenda of the urban sustainable development goal. *Planning Theory*, *15*(4), 435–448. https://doi.org/10.1177/1473095216660786

Wauters, E., Bielders, C., Poesen, J., Govers, G., & Mathijs, E. (2010). Adoption of soil conservation practices in Belgium: An examination of the theory of planned behaviour in the agri-environmental domain. *Land Use Policy*, *27*(1), 86–94. https://doi.org/10.1016/j.landusepol.2009.02.009

WAVES. (2017). *Annual Report 2017*. Wealth Accounting and the Valuation of Ecosystem Services. Retrieved from https://www.wavespartnership.org/sites/waves/files/kc/WAVES-Annual-Report-final.pdf

WBCSD. (2016). *Sustainability beyond fence-lines: Why landscape approaches make business sense*. World Business Council for Sustainable Development (WBCSD), and the Sustainable Trade Initiative (IDH), The Forests Dialogue (TFD), and the Sustainable Food Lab (SFL). Retrieved from http://www.wbcsd.org/Clusters/Natural-Capital-and-Ecosystems/Resources/Sustainability-beyond-fence-lines-brief-paper

Weatherdon L. V., Fletcher R., Jones M. C., Kaschner K., Sullivan E., Tittensor D. P., Mcowen C., Geffert J.

L., van Bochove J. W., Thomas H., Blyth S., Ravillious C., Tolley M., Stanwell-Smith D., Fletcher S., Martin C. S. (2015). *Manual of marine and coastal datasets of biodiversity importance*. Cambridge (UK): UNEP World Conservation Monitoring Centre. Retrieved from https://www.unep-wcmc.org/system/dataset_file_fields/files/000/000/340/original/Weatherdon15-UNEP-WCMC-Dec2015.pdf?1449230861

Weber, K. T., & Horst, S. (2011). Desertification and livestock grazing: The roles of sedentarization, mobility and rest. *Pastoralism: Research, Policy and Practice*, *1*(1), 19. https://doi.org/10.1186/2041-7136-1-19

West, S. (2012). REDD + and adaptation in Nepal. *REDD Net*, (September 2011), 1–13. Retrieved from http://redd-net.org/resource-library/REDD++and+adaptation+in+Nepal

West, T. O., & Marland, G. (2003). Net carbon flux from agriculture: Carbon emissions, carbon sequestration, crop yield, and land-use change. *Biogeochemistry*, *63*(1), 73–83. https://doi.org/10.1023/A:1023394024790

Whalen, J., (Chunbao) Xu, C., Shen, F., Kumar, A., Eklund, M., & Yan, J. (2017). Sustainable biofuel production from forestry, agricultural and waste biomass feedstocks. *Applied Energy*, *198*, 281–283. https://doi.org/10.1016/J.APENERGY.2017.05.079

Whiting, S. N., Reeves, R. D., Richards, D., Johnson, M. S., Cooke, J. A., Malaisse, F., Johns, R., Mcintyre, T., Purvis, O. W., Salt, D. E., Schat, H., Zhao, F. J., & Baker, A. J. M. (2010). Research Priorities for Conservation of Metallophyte Biodiversity and their Potential for Restoration and Site Remediation, *Restoration Ecology 12*(1), 106–116.

Wilhite, D. A., Sivakumar, M. V. K., & Pulwarty, R. (2014). Managing drought risk in a changing climate: The role of national drought policy. *Weather and Climate Extremes*, *3*, 4–13. https://doi.org/10.1016/j.wace.2014.01.002

Williams, C., & van Triest, S. (2009). The impact of corporate and national cultures on decentralization in multinational corporations. *International Business Review*, *18*(2), 156–167. https://doi.org/10.1016/j.ibusrev.2009.01.003

Williams, G. M., Kroes, R., & Munro, I. C. (2000). Safety Evaluation and Risk Assessment of the Herbicide Roundup and Its Active Ingredient, Glyphosate, for Humans. *Regulatory Toxicology and Pharmacology*, *31*(2), 117–165. https://doi.org/10.1006/rtph.1999.1371

Windfuhr, M. (2016). FAO: Voluntary Guidelines on Responsible Governance of Tenure of Land, Forests and Fisheries - Relevance, Reception and First Experiences in Implementation,. In G. H., H. I., Q. T., R. O., & P. Wegerdt (Eds.), *International Yearbook of Soil Law and Policy*. Cham: Springer.

Wingeyer, A. B. Amado, T. J. C. Bidegain, M. P. Studdert, G. A., Varela, C. H. P., Garcia, F. O., & Karlen, D. (2015). Soil quality impacts of current South American agricultural practices. *Sustainability*, *7*, 2213–2242.

Winslow, M. D., Vogt, J. V., Thomas, R. J., Sommer, S., Martius, C., & Akhtar-Schuster, M. (2011). Science for improving the monitoring and assessment of dryland degradation. *Land Degradation & Development*, *22*(2), 145–149. https://doi.org/10.1002/ldr.1044

WOCAT. (2009). *Benefits of sustainable land Management*. University of Berne: World Overview of Conservation Approaches and Technologies and Centre for Development and Environment.

Wu, H., Zhang, J., Ngo, H. H., Guo, W., Hu, Z., Liang, S., Fan, J., & Liu, H. (2015). A review on the sustainability of constructed wetlands for wastewater treatment: Design and operation. *Bioresource Technology*, *175*, 594–601. https://doi.org/10.1016/j.biortech.2014.10.068

Wubs, E. R. J., van der Putten, W. H., Bosch, M., & Bezemer, T. M. (2016). Soil inoculation steers restoration of terrestrial ecosystems. *Nature Plants*, *2*(107).

Wunder, S., Engel, S., & Pagiola, S. (2008). Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics*, *65*(4), 834–852. https://doi.org/10.1016/j.ecolecon.2008.03.010

Yang, L., Lu, F., Wang, X., Duan, X., Song, W., Sun, B., Chen, S., Zhang, Q., Hou, P., Zheng, F., Zhang, Y., Zhou, X., Zhou, Y., & Ouyang, Z. (2012). Surface methane emissions from different land use types during various water levels in three major drawdown areas of the Three Gorges Reservoir. *Journal of Geophysical Research: Atmospheres*, *117*(D10), n/a-n/a. https://doi.org/10.1029/2011JD017362

Yapi, T. (2013). *An assessment of the impacts of invasive Australian wattle species on grazing provision and livestock production in South Africa*. Faculty of AgriSciences. Stellenbosch: Stellenbosch University. MSc. Thesis.

Yucer, A. A., Kan, M., Demirtas, M., & Kalanlar, S. (2016). The importance of creating new inheritance policies and laws that reduce agricultural land fragmentation and its negative impacts in Turkey. *Land Use Policy*, *56*, 1–7. https://doi.org/10.1016/j.landusepol.2016.04.029

Zabel, A., & Roe, B. (2009). Optimal design of pro-conservation incentives. *Ecological Economics*, *69*(1), 126–134. https://doi.org/10.1016/j.ecolecon.2009.08.001

Zedler, J. B., Doherty, J. M., & Miller, N. A. (2012). Shifting Restoration Policy to Address Landscape Change, Novel Ecosystems, and Monitoring. *Ecology and Society*, *17*(4). https://doi.org/10.5751/es-05197-170436

Zeller, H., Zah, R., Gioacchini, M., & Faist, M. (2016). *2nd Edition of the Environmental Impact Valuation as base for a Sustainable Fashion Strategy*. Retrieved from https://group.hugoboss.com/files/user_upload/Nachhaltigkeit/Produkt/31_05_2017_2nd_White_Paper_EIV.pdf

Zhao, F., Hitzhusen, F., & Chern, S. (1991). Impact and implications of price policy and land degradation on agriculture growth in developing countries. *Agriculture Economics*, *5*, 311–324.

Zhao, Q., Bai, J., Huang, L., Gu, B., Lu, Q., & Gao, Z. (2016). A review of methodologies and success indicators for coastal wetland restoration. *Ecological Indicators*, *60*, 442–452. https://doi.org/10.1016/j.ecolind.2015.07.003

Zimmermann, H, Hofmann, J, Witt, A. (2006). A South African Perspective on Prosopis. *Biocontrol News and Information*, (27), 6–9.

Zomer, R. J., Trabucco, A., Coe, R., Place, F., Van Noordwijk, M., & Xu, J. (2014). *Trees on farms: an update and reanalysis of agroforestry's global extent and socio-ecological characteristics*. Working Paper 179. Bogor, Indonesia: World Agroforestry Centre (ICRAF) Southeast Asia Regional Program. Retrieved from http://www.worldagroforestry.org/downloads/Publications/PDFS/WP14064.pdf