12

RESTORING DESERT ECOSYSTEMS

Scott R. Abella

Introduction

Deserts include arid and semi-arid lands where evaporation plus transpiration (water lost from plants to the air) exceeds precipitation. The 2005 Millennium Ecosystem Assessment categorized dryland climates as hyper-arid, arid, semi-arid and sub-humid (Table 12.1). This classification was based on an aridity index, defined by how much greater evapotranspiration is than precipitation. Hyper-arid lands have at least 20 times more evapotranspiration than precipitation, indicating a moisture deficit. For comparison, humid climates often have less evapotranspiration than precipitation.

Hyper-arid, arid and semi-arid lands occupy one-third of Earth's 147,573,197 km^2 land area (Safriel *et al.* 2005). These drylands occupy most of Australia and Africa, much of Asia and western North America and dry portions of South America (Figure 12.1). Twenty per cent of Earth's human population lives in deserts, which increases to over one-third if drought-susceptible

Table 12.1 Classification of dryland climates and their characteristics based on the Millennium Ecosystem Assessment

Climate	*Aridity index*[a]	*Major biome*	*Earth's land (%)*	*Human population (%)*[c]
Hyper-arid	> 20:1	Desert	7	2
Arid	20:1 to 5:1	Desert	11	4
Semi-arid	< 5:1 to 2:1	Grassland	15	14
Sub-humid[b]	< 2:1 to 1.5:1	Woodland	9	15

Notes: [a] Ratio of annual evapotranspiration to precipitation. Evapotranspiration is the sum of evaporation and transpiration (water lost from plants to the air). All of these dry climates have ratios greater than 1, as evapotranspiration exceeds precipitation. This means there is a moisture deficit. Polar regions can also be dry but are not considered in this chapter as deserts.

[b] Sub-humid environments often support trees and savanna grasslands, but are sometimes grouped with deserts because they are susceptible to degradation during droughts.

[c] Percentage of the total human population living in each climate.

Source: Safriel *et al.* (2005)

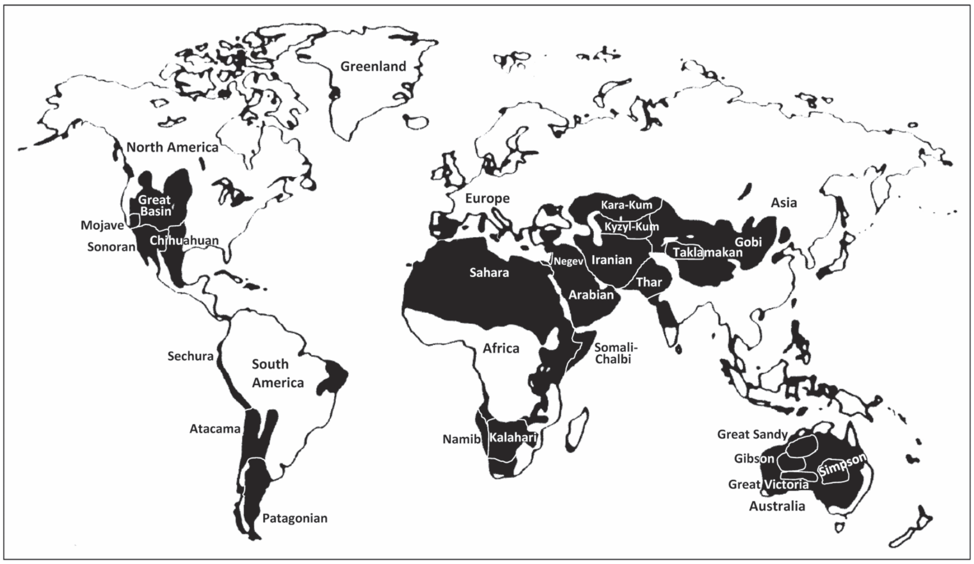

Figure 12.1 Location of hyper-arid, arid and semi-arid lands (all shaded in black), with examples of specific deserts highlighted

Source: map adapted from Ffolliott *et al.* (2001)

sub-humid lands are included. In addition to providing human habitat, 8 of 25 global biodiversity 'hotspot' regions are in drylands (*ibid.*). Despite their relatively low productivity, deserts (including sub-humid lands) store 27 per cent of Earth's soil organic carbon (*ibid.*). Remarkably, deserts further store 97 per cent of Earth's entire soil inorganic carbon. These examples illustrate that degradation of desert land, and its reversal through restoration, has both local and global implications.

While an idealized vision of a desert might be a hot, dry, sparsely vegetated land with sand dunes, this fits only a portion of the world's deserts. For example, the Great Basin Desert in western North America is cold most of the year, with freezing temperatures common, and receives snow (Figure 12.2). Climates within and among deserts are variable, with some characterized by greater precipitation or different seasonal patterns of rainfall than other deserts (Whitford 2002). A given desert can experience shifts in aridity through time with changes in climate or human land uses, such as increasing evaporation through alterations to the soil surface. A commonality is that ecological restoration is challenging in deserts, because deserts represent extremes of Earth's climates and precipitation is unreliable. It is not uncommon for some deserts to receive little or no rainfall for an entire year, or even multiple years.

Desert ecology principles paramount to restoration

Three principles of desert ecology germane to restoration include: (1) extreme spatial and temporal patterning of resource availability (e.g. water and nutrients), (2) unique nature and speed of vegetation change, and (3) prevalence of herbivory (eating of plant matter by animals) and granivory (eating of seeds by animals). An order of magnitude variation in the

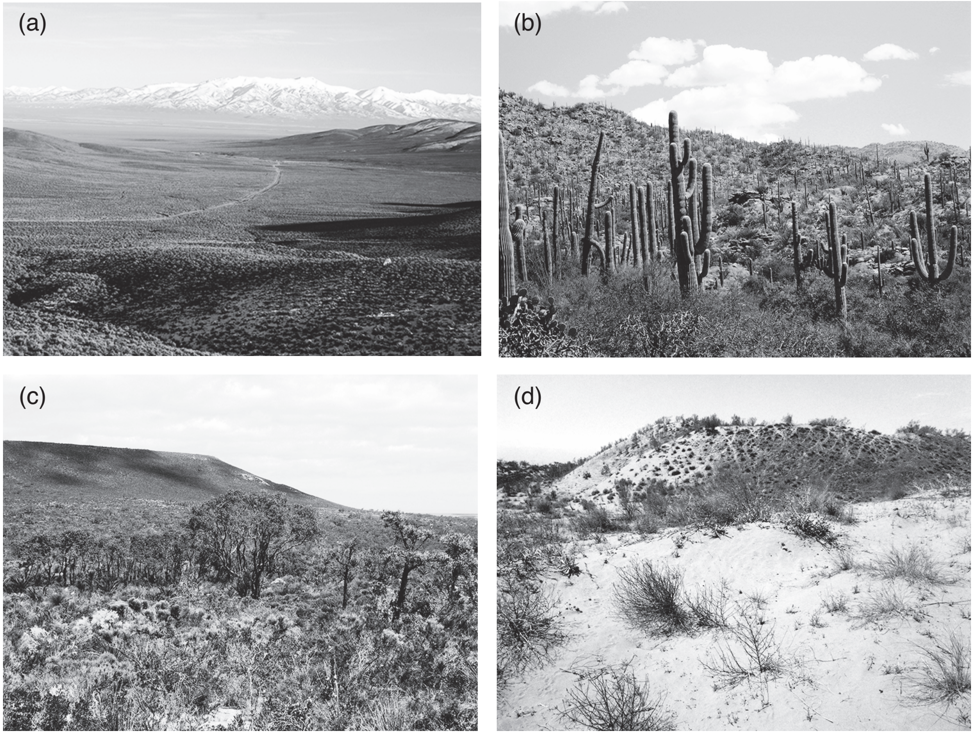

Figure 12.2 Examples of arid and semi-arid ecosystems: (a) Great Basin Desert, a cold desert in the western United States, with the photo taken in Nevada with snow on surrounding hills; (b) Sonoran Desert of the southwestern United States and Mexico, showing the columnar cactus giant saguaro, with the photo taken in Arizona; (c) transitional semi-arid woodland in Western Australia; and (d) Gurbantunggut Desert northeast of Urumqi, Xinjiang Uygur Autonomous Region, northwestern China, with the dark area on the right side of the slope covered with biological soil crust

Source: photos by S. R. Abella

concentration of soil nutrients within a few meters of horizontal space is common in deserts (Padilla and Pugnaire 2006). This spatial patterning of nutrient availability often results from the distribution of perennial plants, below which soil nutrients accumulate. These 'fertile islands' also trap dust and seeds, provide shade, ameliorate extreme weather, and often support high biological activity compared to the interspaces between perennial plants. Distribution of large shrubs can partly regulate recruitment of annual and other perennial plants, because some plant species depend on fertile islands for germination and seedling establishment (Abella and Smith 2013).

Similar to their spatial patterning of biological activity, deserts are described as 'pulse systems' because brief periods of resource availability (e.g. following rainstorms) can influence an entire year's plant and animal activity (Whitford 2002). The extreme temporal variability of deserts can drive trajectories of desert restoration projects. A well-timed rain, for instance, could trigger plant establishment early in a restoration project to initiate a persistent trajectory that hinged upon presence of perennial plants. On the other hand, dry conditions could result in loss of restoration materials (e.g. seeds) and minimal restoration success.

Deserts may seem recalcitrant to change, but this masks appreciable short- and long-term change that can occur in desert vegetation. Vegetation change in deserts after disturbance does not necessarily fit an idealized 'succession' described for moister regions (Abella 2010). For example, a post-disturbance succession after removal of temperate forest may include initial colonization by annual plants, followed by perennial grasses or shrubs, and then trees to form a forest largely lacking annual plants. In deserts, annual plants are components of both early colonizing and mature ecosystems, though annuals may only be present in high-rainfall years. Species in some desert shrublands and grasslands re-sprout if their tops are killed. Thus, species of the mature community re-establish directly, without an intervening early colonizing community, not fitting traditional succession as a transition from one community to another. In other cases, short-lived perennial species, uncommon in mature vegetation, can initially colonize, eventually giving way to re-establishment of species of the mature community. While brief droughts can rapidly alter vegetation through death of certain perennial species, the pace of plant colonization after disturbance is generally slower in deserts than in moister environments.

Herbivory and granivory in certain deserts is extreme. While herbivory can have a major influence in temperate ecosystems, it can be so extreme in deserts as to remove plant cover completely, because there is little forage to begin with and plants grow slowly. One example is the Arabian Desert in the Middle East, where domestic camels almost remove plant cover entirely (Figure 12.3). Restoration will fail unless herbivory is accounted for or restoration sites are protected. Similarly, insects, mammals, and birds can consume large quantities of seeds, including those intended for restoration. The importance of granivory varies among deserts (Brown *et al*. 1979), and where it is prevalent, is a major consideration for restoration.

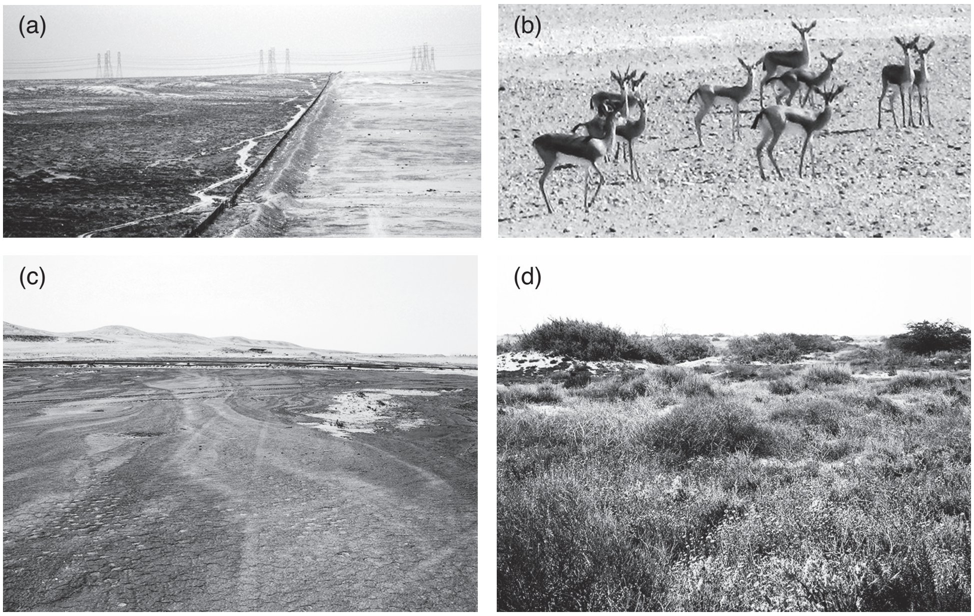

Figure 12.3 Context of restoration in the Arabian Desert of Kuwait: (a) fencing is essential to protect restoration sites from domestic camel grazing; (b) gazelle, one of several species of large animals that have been reduced or eliminated in the Arabian Desert; (c) 'oil lake' resulting from the Gulf War; (d) shrubland developing free from camel grazing inside a protected area, one of several possible reference communities for restoration

Source: photos by S. R. Abella

Why is restoration needed in deserts?

Deserts have long been used by humans for extractive purposes that reduce long-term productivity or generate negative off-site impacts, stimulating restoration as a land management tool. Many deserts are endowed with valuable rocks and minerals, making mining common in deserts. Restoration is mandated after mining in Australia, for example, to reduce negative off-site impacts (e.g. generation of dust), re-establish biodiversity, and maintain land productivity for other possible uses (Commander *et al*. 2013). Dryland agriculture has occurred in many deserts, with farms frequently abandoned or with declining productivity over time. Restoration may help stabilize eroding soil, or be performed in certain locations (such as near water sources) to help sustain productivity within a working agricultural landscape. Rangeland for grazing livestock is the most extensive land use of deserts and has been practiced for thousands of years in some deserts such as the Arabian. Overgrazing is a leading cause of desertification, which is land degradation in arid lands that reduces land productivity and typically makes deserts even drier (e.g. by reducing capacity for rainwater to soak into soil). Humans also commonly withdraw groundwater or alter surface water flow, both of which can affect plant productivity and spur restoration. Invasion by non-native plants in certain deserts, such as the Mojave Desert in the United States, has changed desert fuels and corresponded with increased extent of wildfires. These fires destroy mature desert vegetation. Numerous other disturbances – such as roads no longer needed – are environments where restoration is conducted in deserts. Desired functional outcomes, like reducing dust or enhancing habitat for conservation-priority wildlife species, can also spur restoration. Degraded arid lands are missing key ecological functions and are liabilities locally and globally through their influence on the atmosphere.

Restoration goals and reference conditions

Considerations for whether to use restoration and guidelines for conducting restoration projects in deserts are similar to those for other ecosystems (Society for Ecological Restoration 2004). A first task is identifying if or how an area is degraded, such as based on whether key ecosystem components (e.g. surface soils, perennial plants) are missing. Another task is determining what type (or combination) of management is most appropriate to reverse the degradation. Restoration is just one of numerous possible management interventions along a continuum of management options. For example, constructing a fence to reduce livestock grazing and promote plant growth would usually be considered a management action but not active ecological restoration. Fencing plus actively performing treatments to assist soil recovery or planting native plants inside the exclosure, however, would often be considered restoration. Land management activities like habitat creation or land reclamation (e.g. using non-native plants for habitat or soil stabilization) can be useful for particular management objectives but should not be termed desert restoration.

As in temperate ecosystems, reference conditions underpin desert restoration. Reference conditions represent our understanding of the ecological conditions (e.g. species present, natural types of disturbance such as flooding, depth of the soil) characterizing ecosystems relatively free from degradation. These conditions can be based on knowledge of an ecosystem before it was degraded, nearby less-degraded sites, or derived through modelling ecological processes like losses or gains of soil nutrients.

The extreme temporal variation of deserts complicates evaluations of reference conditions and care must be used to account for this. Long-term vegetation monitoring in Joshua Tree National Park in the Mojave Desert provides an example. Miriti *et al*. (2007) inventoried perennial plants in a 1 ha plot every five years from 1984 to 2004. The density of adult plants of the large shrub creosote bush (*Larrea tridentata*) changed little during the 20-year period, but populations of other shrubs and perennial forbs fluctuated (Figure 12.4). The shrub white bursage (*Ambrosia dumosa*) had a relatively constant number of individuals (1,555 to 1,714/ha) during the 1984 to 1999 inventories, but then plummeted to only 523 individuals in 2004. The forb desert globemallow (*Sphaeralcea ambigua*) had densities of 50 to 81 plants/ha between 1984 and 1999 before completely disappearing in 2004. Droughts occurred at the study site between 1988 and 1991 and 1999 and 2003. While the every-5-year plant census could not detect annual fluctuations, the 2004 inventory reflected mortality of perennial plants associated with the severe 1999–2003 drought (Miriti *et al*. 2007). If reference conditions at this site were assessed only in 2004, we would underestimate densities of most species relative to the previous 20 years and completely 'miss' desert globemallow. This example shows how recent weather could influence our perception of reference conditions and that stability of plant populations (including through droughts) varies among desert perennial species.

Techniques for restoring components of desert ecosystems

Particular components of desert ecosystems might be degraded or missing. Only one component might require restoration if the rest of the ecosystem is healthy, or several components could require restoration within a comprehensive ecological framework. The next sections discuss techniques for components most commonly requiring restoration.

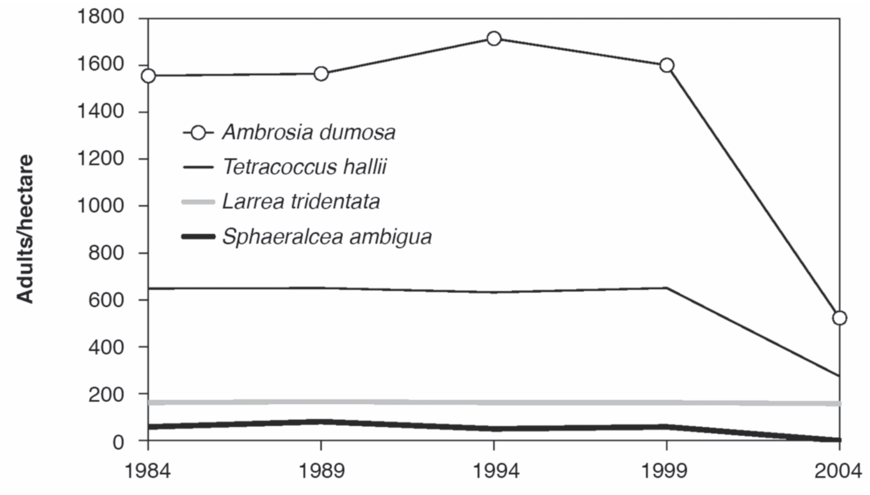

Figure 12.4 Fluctuating density and species composition of the perennial plant community of a site in Joshua Tree National Park, Mojave Desert, United States. Mortality of certain perennial species corresponded with extended dry periods, making consideration of variable weather patterns important in assessing reference conditions

Source: data from Miriti *et al.* (2007)

Soil

Soil formation is slower in deserts than in temperate ecosystems, and salvaging topsoil from a disturbed area for later reapplication can be among the most effective restoration strategies. The upper 5 cm of soil can contain almost all the viable seed bank of desert soils (Guo *et al.* 1998). Similarly, the upper 10 cm stores much of a desert soil's plant-available nutrients, organic matter, and microorganisms. Because of these concentrated resources in the topsoil, applying salvaged topsoil accelerated plant colonization better than seeding at decommissioned quarries in the Namib Desert (Burke 2008).

Several techniques can help increase efficiency and success of salvaging desert soils. Properties of the soils should be considered, including avoiding salvaging toxic soils and soils within or near infestations of non-native plants. Timing salvage to correspond with the end of maximum dispersal of native seeds within a year might help maximize retention of native seed banks. Salvaging only the upper 5 to 10 cm of soil can be critical in many deserts to avoid 'diluting' the organic-rich topsoil with subsoil (Scoles-Sciulla and DeFalco 2009). Ideally, topsoil would not be stored at all and immediately applied to a restoration site. This is rarely feasible, necessitating topsoil be stored. In arid land of India, stored topsoil lost 27 per cent of its nitrogen during a year of storage and another 10 per cent during the second year (Ghose 2001). Keeping stockpiles short, perhaps less than 60 cm tall to avoid creation of anaerobic conditions, might help increase longevity of biotic components. Furthermore, treatments such as covering stockpiles in water-proof tarps or even planting vegetation on piles can extend longevity of soil biota if soil must be stored for a long time (Golos and Dixon 2014). Strategically re-applying topsoil where it is needed most and matching soil types with the donor and recipient sites likely makes the best use of limited amounts of salvaged soil.

Compacted and eroded soils are common legacies of arid land degradation. These soils are problematic because retention of seed and fine soil particles, holding of water, and accumulation of soil organic matter are compromised. Roughening compacted soil using machinery or hand tools can break up impermeable surfaces and create indentations to trap water and organic matter (Bainbridge 2007). In areas of severe wind erosion, constructing a series of parallel barriers (e.g. short fences) can slow wind speed and result in deposition and retention of wind-blown soil. Covering soil with mulches and organic matter also can help stabilize eroding soils and alleviate compaction. Materials used for cover can include mats, nets, or networks of cylindrical structures (e.g. sticks) pushed into the soil, all of which can be of organic origin and biodegrade. Materials for mulching can include straw, wood chips, charcoal, or gravel.

Trade-offs of different cover and mulch materials need to be considered. For example, wood chips can make soil nutrients unavailable to plants by providing food for soil microbes, which then uptake nutrients. This could be desirable for reducing nutrient-loving, non-native plants, but undesirable if growth of native plants is harmed. On the other hand, reduced nutrient availability might be compensated for by increased moisture retention under protective mulch. Whichever techniques are used, a key for restoration on compacted or eroding soil is promoting retention of natural soil structure and organic matter, facilitating establishment of plants and soil biota for long-term soil health.

Biological soil crusts

Located on or near the soil surface, biological soil crusts include bacteria, cyanobacteria, algae, mosses, liverworts, fungi and lichens (West 1990). Crusts in particular deserts or sites within deserts may only contain one or a few of these components. Moreover, the composition of crust can change during ecosystem development. Bacteria are often the initial colonizers after disturbance, and lichens the last. Not all desert soils have crusts – crusts are typically absent on naturally unstable soils (e.g. ephemeral stream channels, shifting sand dunes) and on soil perpetually disturbed by human activities, including through off-road vehicles or livestock grazing. Where crusts are found, they are major features of desert soils and contribute to soil stability, nutrient cycling, interactions with vascular plants (variously enhancing or reducing vascular plant establishment), soil temperature regimes and biodiversity. Loss of soil crusts can de-stabilize soil, accelerating soil erosion, and result in loss of carbon storage and nutrients.

Most research on biological soil crusts has focused on how disturbance affects them, but recent attention has turned toward restoring crusts (Bowker 2007). One example is in the Gobi Desert of Inner Mongolia, China. Wang *et al.* (2009) collected algae-dominated soil crust from relatively undisturbed sites, propagated the algae in a greenhouse, and created a slurry to increase the volume of inoculum. The slurry was then applied to eroding soil at field sites. Within one year, cyanobacteria covered 42 per cent of the soil surface (compared to 0% on controls). Between the first and third years, different species of cyanobacteria colonized the soil surface, and mosses colonized by the second year. The resulting formation of biological soil crust changed soil functions. Soil organic carbon was < 1 g/kg in controls without inoculum, but was 9 g/kg by the third year on plots receiving inoculum. The cover of vascular plants also increased as the soil crusts formed. Effective restoration treatments are likely to vary with the type of crust, stability and properties of the underlying soil, and resources available.

Perennial plants

Perennial plants are a primary component targeted for restoration in deserts because of their importance to soil formation and other biota. Perennial plants can be restored through planting nursery-grown seedlings (outplanting), salvaging plants before disturbance for later replanting, seeding, or facilitating natural recruitment (Abella and Newton 2009). Successful outplanting projects start with quality seed and seedlings grown under conditions appropriate for the species. At least 6 to 12 months of growth in a nursery are commonly required for seedlings to develop root systems providing the best chance of survival when planted at restoration sites. This makes advance planning, sometimes two years or more, necessary to collect seed and grow plants. Seedlings can be grown in a variety of containers, such as round plastic pots, deeper rectangular pots and biodegradable materials that can be planted without having to remove the soil and roots (Bainbridge 2007).

Understanding the ecology of planting sites and the plant species is vital to identify treatments required to enhance plant establishment. For example, herbivory is often intense, requiring that planting sites be fenced to exclude large herbivores or that individual plants be enclosed in shelters. Shelters, such as plastic cylinders, can provide protection from herbivory and ameliorate hot, dry weather. Other treatments, including irrigation, may be evaluated on a cost/benefit basis relative to simply planting more untreated plants. Irrigation is typically difficult to implement at remote restoration sites and can have unintended effects, such as promoting non-native plants and wetting biological soil crusts during time periods detrimental to their growth. With due consideration of these types of trade-offs, strategically delivering water or using slow-release irrigation gels (e.g. DriWater) has increased plant establishment at remote sites such as Saudi Arabia semi-arid woodlands (Aref *et al.* 2006). Not all species require treatments such as irrigation or shelter, so identifying species-specific needs helps increase efficiency of restoration.

Salvaging plants and facilitating natural recruitment both make use of existing plant material on site and can be cost-effective revegetation options. Unless salvaging at the donor and planting at the recipient site can be performed in one operation, the salvaged plants require care in nurseries similar to outplanting. Protecting naturally recruited seedlings to enhance their survival and growth is little studied but may be particularly suited for species difficult to propagate or where transporting plants is difficult.

Seeding perennial species is difficult, owing to infrequent conditions suitable for plant establishment, similar to why natural recruitment events are rare (Abella *et al.* 2012). Germination ecology is poorly understood in many deserts. Enhancing knowledge of germination ecology underpins identifying which species are amenable to seeding and what treatments are required for success. A general approach for enhancing seeding success includes: collecting quality seeds and storing them appropriately unless used immediately, conducting any treatments required to enhance germination, implementing any species-specific treatments for protecting seeds (e.g. coating them) or improving soil substrates, and timing seeding optimally within a year to coincide with favourable conditions. While still no guarantee of success, these procedures can provide the best chance for success if weather conditions are favourable for plant establishment. Additionally, pairing outplanting with seeding might maximize chances that some plants become established.

Annual plants

Annual plants can be restored through seeding and indirectly through establishing perennial plants or improving site conditions. Perennial plants in natural desert ecosystems typically facilitate recruitment of annual plants, and diversifying the perennial plant community can increase diversity of annual plants (Abella and Smith 2013). Improving knowledge of annual plant germination ecology and restoration is particularly important in changing desert climates, as the frequency and timing of weather conditions suitable for germination may shift.

Hydrology and springs

Humans have manipulated water flow in deserts for thousands of years to increase retention of water or concentrate it in certain areas for agriculture or domestic purposes. One example is the Negev Desert in Israel, where ancient rainwater harvesting systems of terraced hillsides, small canals, conduits (low rock structures to direct water) and cisterns remain visible on the landscape (Evenari *et al.* 1982). Some of these millennia-old water harvesting techniques are employed in contemporary restoration to retain water on site and enhance infiltration into soil (Bainbridge 2007).

Three of the main surface water hydrological patterns in deserts include: (1) water flow across the soil surface as sheet flow, (2) ephemeral stream channels flowing after rains, and (3) permanently flowing rivers and springs. Human disturbances that disrupt sheet flow include soil compaction that limits infiltration of some of the flowing water into the soil; tracks from off-road vehicles that form artificial flow paths; disturbances like fire or oil spills that create hydrophobic layers; and roads that divert flow or concentrate it into a few pathways. Ephemeral stream channels are disrupted by land-clearing disturbance and roads that cut across channels and sever connectivity (and without culverts to allow water to pass under the road). In addition to restoring plants and soils, several tactics can help re-establish natural drainage patterns. Decommissioning unwanted roads and removing roadside berms can stop roads from diverting surface flow and severing stream channels. Creating small water catchments on disturbed soil surfaces can retain water and provide locations for plant recruitment.

Desert springs and rivers are special cases in desert restoration because of their important functions and because availability of water is not necessarily as limiting at these sites as it is elsewhere. Major ways humans have altered springs is by piping water away from springs, excavating soil around springs in an attempt to store water, or pumping groundwater to lower the water table. In these circumstances, a main goal of restoration is reconfiguring how water is distributed on the site. For springs that have been piped or excavated, removing the pipes or filling in storage basins may be an initial restoration step. A lowered water table makes restoration of natural surface flow particularly difficult. Partial restoration may still be possible by re-contouring to lower the elevation of the spring to the new level of the water table, allowing outflow to the surface. At many springs, restricting access of livestock to project sites may be critical to allow re-establishment of plants, and this likely requires balancing access of native animals to the water source.

Restoring surface water in deserts can produce tremendous ecological benefits. For example, Patten *et al*. (2008) developed models for transitions from dry upland plant species composition to wetland plant composition based on changing the depth to the water table even slightly, at one-metre increments. Similarly, high biodiversity in aquatic invertebrates, fish and birds might depend on permanency of flowing springs and associated wetlands in central Australia arid lands (Box *et al*. 2008). Oases occupy only small portions of desert landscapes, but have a disproportionate restoration potential for enhancing biodiversity and ecosystem functions.

Animals

Several species of large animals are native to Middle Eastern deserts and provide examples of the challenges and opportunities for restoring large desert animals. Two examples are the gazelle (*Gazella* spp.) and the Arabian oryx (*Oryx leucoryx*) in the Arabian Desert, spanning several countries such as United Arab Emirates, Kuwait and Saudi Arabia. Gazelles are small antelopes that are herbivores and fast runners – some can run 100 km/hour in short bursts. The Arabian oryx is a medium-sized antelope, weighing about 70 kg, which roamed in herds searching for plants following rains and can go several weeks without water. Gazelles and oryx were hunted by humans for food for thousands of years, but catastrophic declines largely began in the early 1900s with introduction of rifles, vehicles and intensified livestock grazing (Thouless *et al*. 1991). A major land use currently affecting gazelle and oryx, and their potential for restoration, is grazing by domestic camel. For example, the United Arab Emirates contained 250,000 mostly free-roaming camels in the mid-2000s, or 3 camels/km^2 (El-Keblawy *et al*. 2009). Camels today are kept primarily for racing, as camel racing competitions are a major cultural activity in the Arabian Desert.

Even where hunting and harassment by humans of gazelle and oryx have been curtailed, the animals face a situation of sparse to non-existent forage on camel rangelands. As a result, the main conservation approach has been to create fenced reserves and reintroduce gazelle and oryx inside. One example is the 225-km^2 fenced Dubai Desert Conservation Reserve, established by 2003 and occupying 5 per cent of Dubai (El-Keblawy *et al*. 2009). Camels still grazed in much of the reserve, but a 27-km^2 portion had only gazelles and oryx. As has been observed in some other fenced reserves containing only native animals, vegetation composition quickly differentiated between the areas open to camel grazing and those only open to native animals. The small shrubs ramram (*Heliotropium kotschyi*) and rattlebox (*Crotalaria aegyptiaca*) were only found with native animals, whereas the unpalatable large shrub rimth (*Haloxylon salicornicum*) and the sedge thenda (*Cyperus conglomeratus*) dominated camel rangelands.

Restoration of these animals must also fit the landscape context, as an example from central Saudi Arabia illustrated. The 2,200-km^2 Mahazat as-Sayd Protected Area was fenced in 1988 and oryx and gazelle were reintroduced in the 1980s and 1990s (Islam *et al*. 2010). Mortality of the animals was heavy in dry years, with 560 oryx and 2,815 gazelle dying between 1999 and 2008. Historically, populations likely moved hundreds of kilometres to locate forage after localized rains. Despite the relatively large size of the fenced reserve, the fence prevents natural roaming of the animals over a much larger area. To partly offset this, revised management plans included providing supplemental forage and water within strategic locations of the reserve (*ibid*.).

Case studies

Western Australia arid zone

Mining is a major land use in arid lands of Australia, and restoration is commonly performed after mining. One study recently compared outplanting and seeding for restoration on disturbed borrow pits at a mine on the Edel Peninsula, within the Shark Bay World Heritage Area (Commander *et al*. 2013). The area receives 22 cm/year of rainfall, and natural vegetation consists of desert shrubland. Based on a reference condition of nearby, undisturbed vegetation, seed was collected of three shrub species (*Acacia tetragonophylla*, *Atriplex bunburyana* and *Solanum orbiculatum*) in sufficient quantity for direct seeding and for propagating some plants in a nursery. To roughen the soil surface, the borrow pits were ripped with a grader. Some areas then received seed broadcast on the soil surface, while others received outplanting. Rainfall was only 68 per cent of average during the two-year study.

Researchers concluded that on these relatively small sites (~ 1 ha), outplanting more rapidly revegetated the soil than did seeding (Commander *et al*. 2013). The percentage of seeds producing a seedling varied with timing of seeding and was highest where the soil was ripped. Survival of outplants varied among species (being highest in *Atriplex bunburyana* at 42%) and was enhanced by ripping. Fertilizer, water-holding gel, and pruning minimally influenced survival. In dry periods, outplanting likely outperforms seeding, but if timing of seeding can be flexible, brief windows of moist conditions might enable seeding to be effective.

Mojave Desert, United States

Parks managed by the US National Park Service, such as Lake Mead National Recreation Area in the Mojave Desert, undergo periodic maintenance of highways traversing the parks. Restoration was conducted where an existing road was to be removed and re-routed nearby through intact desert (Abella *et al*. 2015a). The restoration approach included salvaging topsoil (upper 5 to 20 cm) and 2,105 individuals of 23 native perennial species including cacti, shrubs, forbs and grasses from segments to be destroyed by the new road route. After construction, topsoil was re-applied in 2010, salvaged plants were moved from temporary nurseries to planting sites, and survival of the salvaged plants (some of which received different irrigation treatments) was monitored over a 27-month period. The study period received close to the average of 16 cm/year of precipitation.

Half of the salvaged plants survived the process of salvage and one year of residence in a temporary nursery. About 27 per cent of individuals then survived at least 27 months at restoration sites. Species able to survive being salvaged also generally had among the best survival after planting back at field restoration sites. Cacti had nearly 100 per cent survival and did not require any supplemental irrigation. Several shrub species performed well – such as shadscale saltbush (*Atriplex confertifolia*) with 47 per cent survival and white bursage with 45 per cent survival. The forb desert globemallow also was successful, with 38 per cent survival. The type of irrigation (watering by hand, or DriWater as a slow-release gel) interacted with species differently. Both irrigation types similarly enhanced survival of white bursage by 1.4 times, whereas only hand watering increased survival of desert globemallow. The benefits of planting on salvaged topsoil were substantial: transplants exhibited 56 per cent survival on topsoil, compared to 25 per cent without topsoil. Topsoil alone (without irrigating plants) resulted in plant survival nearly equivalent to that produced by irrigating plants. The project met goals of visual restoration and rapidly revegetating severely disturbed soil, but understanding long-term survival of the planted species and their influence on natural recruitment is desirable (Figure 12.5).

Arabian Desert, Kuwait

On the Arabian Peninsula receiving an average of 12 cm of rainfall annually, Kuwait exemplifies progressive land degradation and the type of conditions desert restoration must ameliorate. Similar to elsewhere in the Arabian Desert, Kuwait experienced an increase in camel grazing,

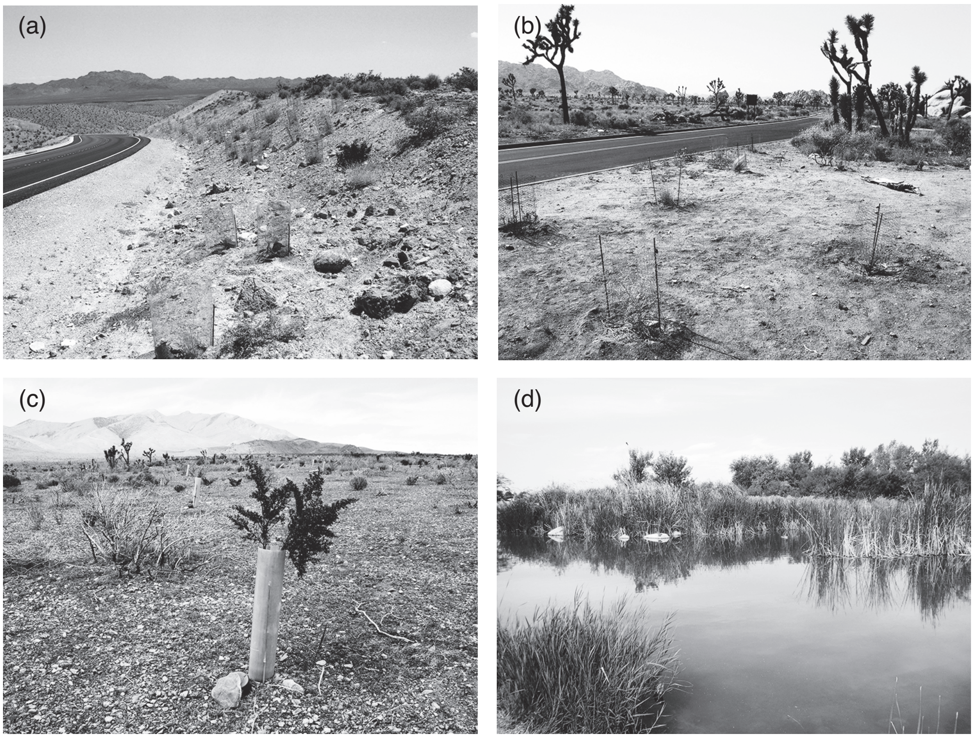

Figure 12.5 Examples of restoration in the Mojave Desert, United States: (a) protecting outplanted perennial plants to revegeate disturbed slopes in Lake Mead National Recreation Area; (b) roadside revegetation in Joshua Tree National Park; (c) protecting outplanted creosote bush for post-wildfire restoration near Las Vegas, Nevada; and (d) establishing desert wetlands to provide the functions of watershed management and wildlife habitat

Source: photos by S. R. Abella

off-road vehicle use, and resource extraction (mainly oil and mining) during the 1900s. This corresponded with lowered plant cover, shifts in plant composition toward unpalatable species, and soil degradation including increased wind erosion producing dust storms harmful to human health (Al-Hurban 2014). Invasion of Kuwait by the Iraqi military in 1990–1991 and the resulting Gulf War devastated already degraded desert land. In addition to damage from construction of military fortifications, movements of war vehicles across the desert, and explosions, 700 Kuwaiti oil wells and pipelines were destroyed by the retreating army. 'Oil lakes' formed across the desert and hardened (Figure 12.3).

Restoration on the current landscape must confront several challenges. First, additional surveys for land mines and unexploded ordinance left by the war are required before restoration can begin. Second, camel grazing is so intense that it must be ameliorated (e.g. by constructing and maintaining fences) before attempting restoration. Third, the legacy of human land use spans millennia, making evaluations of reference conditions difficult. For example, Brown and Al-Mazrooei (2003) found that within four years after fencing, the shrub arfaj (*Rhanterium epapposum*) re-established from underground stumps that had probably remained in the soil for decades. However, this shrubland may itself have been a product of decades to centuries of livestock grazing, and replaced an earlier *Acacia* woodland containing palatable grasses (Brown and Al-Mazrooei 2003). A current approach in Kuwait is to focus restoration within protected areas, survey potential restoration sites for unexploded ordinance, and evaluate a diverse mixture of species and community types for their restoration potential.

Functional outcomes and benefits of desert restoration

Expanding research on the functional outcomes of desert restoration is desirable to improve matching restoration treatments with specific goals and to explore the full potential of restoration. Some examples illustrate functional benefits that can be anticipated locally and globally if restoration in deserts expands.

In the Western Rajasthan region, India, planting the native shrubs rimth and phog (*Calligonum polygonoides*) on desertified land increased storage of organic carbon in the upper 20 cm of soil by 59 per cent (Rathore *et al.* 2015). By curtailing soil erosion and stimulating photosynthesis and accumulation of organic matter, desert restoration has high potential for sequestrating carbon and limiting release of carbon into the atmosphere.

In the western Mojave Desert of California, United States, abandoned eroding farmland created dust storms resulting in air quality violations and interfering with airplane and vehicle travel (Grantz *et al.* 1998). Revegetating these lands, through seeding and outplanting, reduced airborne dust by up to 99 per cent in the 1990s, significantly improving air quality.

Jilantai Salt Lake, in the Ulan Buh Desert, is one of the most economically important salt resources in China (Gao *et al.* 2002). However, salt production was compromised by encroachment of wind-blown sand accelerated through increased woodcutting and livestock grazing. Protection (fencing), combined with planting four native shrub and tree species, increased air humidity by 10 per cent (reducing effects of desertification) and reduced sand encroachment to the salt lake by 85 per cent (*ibid.*).

Restoration has high potential for enhancing habitat quality for desert animals, including threatened species. One example is the desert tortoise (*Gopherus agassizii*), a long-lived reptile (≥ 50 years) of the western Sonoran and Mojave Desert in the United States. To improve forage quality and quantity, several fencing and seeding treatments were tested. Fencing and seeding pelletized seed (coated in a protective substance) increased by sixfold the native annual desert plantain (*Plantago ovata*), a food plant favoured by desert tortoises (Abella *et al.* 2015b).

With over one-fifth of the human population living in deserts and sharing habitat with an unknown number of invertebrate, animal and plant species, expanding restoration in degraded deserts is likely to produce numerous benefits. Future work to improve desert restoration could focus on refining understanding of reference conditions and restoration goals, further developing cost-effective treatments for meeting different goals across spatial and temporal scales, and monitoring functional benefits produced by restoration.

Acknowledgements

I thank Sharon Altman for adapting Figure 12.1, Lindsay Chiquoine for reviewing the section on biological soil crust, and Stuart Allison for helpful comments to improve the chapter.

References

Abella, S. R. 2010. Disturbance and plant succession in the Mojave and Sonoran Deserts of the American Southwest. *International Journal of Environmental Research and Public Health* 7:1248–1284.

Abella, S. R. and A. C. Newton. 2009. A systematic review of species performance and treatment effectiveness for revegetation in the Mojave Desert, USA. In A. Fernandez-Bernal and M. A. De La Rosa (eds), *Arid environments and wind erosion*, pp. 45–74. Nova Science Publishers, Hauppauge, NY.

Abella, S. R. and S. D. Smith. 2013. Annual-perennial plant relationships and species selection for desert restoration. *Journal of Arid Land* 5:298–309.

Abella, S. R., D. C. Craig and A. A. Suazo. 2012. Outplanting but not seeding establishes native desert perennials. *Native Plants Journal* 13:81–89.

Abella, S. R., L. P. Chiquoine, A. C. Newton and C. H. Vanier. 2015a. Restoring a desert ecosystem using soil salvage, revegetation, and irrigation. *Journal of Arid Environments* 115:44–52.

Abella, S. R., L. P. Chiquoine, E. C. Engel, K. E. Kleinick and F. S. Edwards. 2015b. Enhancing quality of desert tortoise habitat: augmenting native forage and cover plants. *Journal of Fish and Wildlife Management* 6(2):150521135029003.

Al-Hurban, A. E. 2014. Effects of recent anthropogenic activities on the surface deposits of Kuwait. *Arab Journal of Geoscience* 7:665–691.

Aref, I. M., L. I. El-Juhany and M. N. Shalby. 2006. Establishment of acacia plantation in the central part of Saudi Arabia with the aid of DRiWATER. In *2nd International Conference on Water Resources and Arid Environment*, 26–29 November 2006, pp. 110–118. King Saud University, Saudi Arabia.

Bainbridge, D. A. 2007. *A guide for desert and dryland restoration*. Island Press, Washington, DC.

Bowker, M. A. 2007. Biological soil crust rehabilitation in theory and practice: an underexploited opportunity. *Restoration Ecology* 15:13–23.

Box, J. B., A. Duguid, R. E. Read, R. G. Kimber, A. Knapton, J. Davis and A. E. Bowland. 2008. Central Australian waterbodies: the importance of permanence in a desert landscape. *Journal of Arid Environments* 72:1395–1413.

Brown, G. and S. Al-Mazrooei. 2003. Rapid vegetation regeneration in a seriously degraded *Rhanterium epapposum* community in northern Kuwait after 4 years of protection. *Journal of Environmental Management* 68:387–395.

Brown, J. H., O. J. Reichman and D. W. Davidson. 1979. Granivory in desert ecosystems. *Annual Review of Ecology and Systematics* 10:201–227.

Burke, A. 2008. The effect of topsoil treatment on the recovery of rocky plain and outcrop plant communities in Namibia. *Journal of Arid Environments* 72:1531–1536.

Commander, L. E., D. P. Rokich, M. Renton, K. W. Dixon and D. J. Merritt. 2013. Optimising seed broadcasting and greenstock planting for restoration in the Australian arid zone. *Journal of Arid Environments* 88:226–235.

El-Keblawy, A., T. Ksiksi and H. El Alqamy. 2009. Camel grazing affects species diversity and community structure in the deserts of the UAE. *Journal of Arid Environments* 73:347–354.

Evenari, M., L. Shanan and N. Tadmor. 1982. *The Negev: the challenge of a desert*. Harvard University Press, Cambridge, MA.

Ffolliott, P. F., J. O. Dawson, J. T. Fisher, I. Moshe, D. W. DeBoers, T. E. Fulbright, J. Tracy, A. Al Musa, C.

Johnson and J. P. M. Chamie. 2001. *Arid and semiarid land stewardship: a 10-year review of accomplishments and contributions of the International Arid Lands Consortium*. General Technical Report RMRS-GTR-89. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Gao, Y., G. Yu Qiu, H. Shimizu, K. Tobe, B. Sun and J. Wang. 2002. A 10-year study on techniques for vegetation restoration in a desertified salt lake area. *Journal of Arid Environments* 52:483–497.

Ghose, M. K. 2001. Management of topsoil for geo-environmental reclamation of coal mining areas. *Environmental Geology* 40:1405–1410.

Golos, P. J. and K. W. Dixon. 2014. Waterproofing topsoil stockpiles minimizes viability decline in the soil seed bank in an arid environment. *Restoration Ecology* 22:495–501.

Grantz, D. A., D. L. Vaughn, R. Farber, B. Kim, M. Zeldin, T. Van Curen and R. Campbell. 1998. Seeding native plants to restore desert farmland and mitigate fugitive dust and PM_{10}. *Journal of Environmental Quality* 27:1209–1218.

Guo, Q., P. W. Rundel and D. W. Goodall. 1998. Horizontal and vertical distribution of desert seed banks: patterns, causes, and implications. *Journal of Arid Environments* 38:465–478.

Islam, M. Z., K. Ismail and A. Boug. 2010. Catastrophic die-off of globally threatened Arabian oryx and sand gazelle in the fenced protected area of the arid central Saudi Arabia. *Journal of Threatened Taxa* 2:677–684.

Miriti, M. N., S. Rodríguez-Buriticá, S. J. Wright and H. F. Howe. 2007. Episodic death across species of desert shrubs. *Ecology* 88:32–36.

Padilla, F. M. and F. I. Pugnaire. 2006. The role of nurse plants in the restoration of degraded environments. *Frontiers in Ecology and the Environment* 4:196–202.

Patten, D. T., L. Rouse and J. C. Stromberg. 2008. Isolated spring wetlands in the Great Basin and Mojave Deserts, USA: potential response of vegetation to groundwater withdrawal. *Environmental Management* 41:398–413.

Rathore, V. S., J. P. Singh, S. Bhardwaj, N. S. Nathawat, M. Kumar and M. M. Roy. 2015. Potential of native shrubs *Haloxylon salicornicum* and *Calligonum polygonoides* for restoration of degraded lands in arid western Rajasthan, India. *Environmental Management* 55:205–216.

Safriel, U., Z. Adeel, D. Niemeijer, J. Puigdefabregas, R. White, R. Lal, M. Winslow, J. Ziedler, S. Prince, E. Archer, C. King, B. Shapiro, K. Wessels, T. Nielsen, B. Portnov, I. Reshef, J. Thonell, E. Lachman and D. McNab. 2005. Dryland systems. In R. Hassan, R. Scholes and N. Ash (eds), *Millennium Ecosystem Assessment: ecosystems and human well-being: current state and trends*, volume 1, pp. 623–662. Island Press, Washington, DC.

Scoles-Sciulla, S. J. and L. A. DeFalco. 2009. Seed reserves during surface soil reclamation in eastern Mojave Desert. *Arid Land Research and Management* 23:1–13.

Society for Ecological Restoration. 2004. *The SER international primer on ecological restoration*. Society for Ecological Restoration International, Tucson, AZ.

Thouless, C. R., J. G. Grainger, M. Shobrak and K. Habibi. 1991. Conservation status of gazelles in Saudi Arabia. *Biological Conservation* 58:85–98.

Wang, W., Y. Liu, D. Li, C. Hu and B. Rao. 2009. Feasibility of cyanobacterial inoculation for biological soil crusts formation in desert area. *Soil Biology and Biochemistry* 41:926–929.

West, N. E. 1990. Structure and function of microphytic soil crusts in wildland ecosystems of arid to semi-arid regions. *Advances in Ecological Research* 20:179–223.

Whitford, W. G. 2002. *Ecology of desert systems*. Academic Press, New York.

ROUTLEDGE HANDBOOK OF ECOLOGICAL AND ENVIRONMENTAL RESTORATION

Ecological restoration is a rapidly evolving discipline that is engaged with developing both methodologies and strategies for repairing damaged and polluted ecosystems and environments. During the last decade the rapid pace of climate change coupled with continuing habitat destruction and the spread of non-native species to new habitats has forced restorationists to re-evaluate their goals and the methods they use. This comprehensive handbook brings together an internationally respected group of established and rising experts in the field.

The book begins with a description of current practices and the state of knowledge in particular areas of restoration, and then identifies new directions that will help the field achieve increasing levels of future success. Part I provides basic background about ecological and environmental restoration. Part II systematically reviews restoration in key ecosystem types located throughout the world. In Part III, management and policy issues are examined in detail, offering the first comprehensive treatment of policy relevance in the field, while Part IV looks to the future. Ultimately, good ecological restoration depends upon a combination of good science, policy, planning and outreach – all issues that are addressed in this unrivalled volume.

Stuart K. Allison is the Watson Bartlett Professor of Biology and Conservation, and Director of the Green Oaks Field Study Center at Knox College, Galesburg, Illinois, USA. He is the author of *Ecological Restoration and Environmental Change* (Routledge, 2012).

Stephen D. Murphy is Professor and Director of the School of Environment, Resources and Sustainability at the University of Waterloo, Ontario, Canada. He is the editor-in-chief of *Restoration Ecology*.

ROUTLEDGE HANDBOOK OF ECOLOGICAL AND ENVIRONMENTAL RESTORATION

Edited by
Stuart K. Allison and Stephen D. Murphy

Routledge
Taylor & Francis Group
LONDON AND NEW YORK

First published 2017
by Routledge
2 Park Square, Milton Park, Abingdon, Oxon OX14 4RN

and by Routledge
711 Third Avenue, New York, NY 10017

Routledge is an imprint of the Taylor & Francis Group, an informa business

British Library Cataloguing-in-Publication Data
A catalogue record for this book is available from the British Library

Library of Congress Cataloging in Publication Data
[CIP data]

ISBN: 978-1-138-92212-9 (hbk)
ISBN: 978-1-315-68597-7 (ebk)

Typeset in Bembo
by FiSH Books Ltd, Enfield

CONTENTS

CONTRIBUTORS

Scott R. Abella, Professor, University of Nevada Las Vegas, School of Life Sciences, Las Vegas, Nevada USA

Susan C. Adamowicz, Land Management Research and Demonstration Biologist, United States Fish and Wildlife Service, Rachel Carson National Wildlife Refuge, Wells, Maine, USA

Stuart K. Allison, Professor, Department of Biology, Knox College, Galesburg, Illinois, USA

Juan J. Armesto, Professor, Department of Ecology, Pontifical Catholic University of Chile, Santiago, Chile

Susan Baker, Professor, Cardiff School of the Social Sciences, Cardiff University, Cardiff Wales, UK

Jonathan D. Bakker, Associate Professor, School of Forestry and Environmental Sciences, College of the Environment, University of Washington, Seattle, Washington, USA

Alex Baumber, Postdoctoral Research Fellow, Institute of Environmental Studies, The University of New South Wales, Sydney, Australia

Sean M. Bellairs, Senior Research Fellow, Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia

Brock Blevins, GIS Analyst, NASA Goddard Space Flight Center, Washington DC, USA

James Blignaut, Professor, Department of Economics, University of Pretoria, Pretoria, South Africa

Keith Bowers, Landscape Architect, Restoration Ecologist, President and Founder, Biohabitats, Inc., Baltimore, Maryland, USA

Ben Brown, Founder, Blue Forests, Graduate Student, Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia

David M. Burdick, Associate Research Professor, Department of Natural Resources and the Environment, University of New Hampshire, Durham, New Hampshire, USA

Marcela A. Bustamenta-Sánchez, Professor, Department of Forestry Science, University of Concepcion, Concepcion, Chile

Alton C. Byers, Director – Science and Exploration, The Mountain Institute, Washington, DC, USA

Robert Cabin, Associate Professor, Department of Environmental Studies, Brevard College, Brevard, North Carolina, USA

Michael A. Chadwick, Lecturer, Department of Geography, King's College London, London, United Kingdom

Robin L. Chazdon, Professor, Department of Ecology and Evolutionary Biology, University of Connecticut, Storrs, Connecticut, USA

Young D. Choi, Professor, Department of Biological Sciences, Purdue University Northwest, Calumet, Indiana, USA

An Cliquet, Professor, Department of European, Public and International Law, Ghent University, Ghent, Belgium

Loren D. Coen, Research Professor, Department of Biological Sciences and Harbor Branch Oceanographic Institute, Florida Atlantic University, Fort Pierce, Florida, USA

Eric Conklin, Director of Marine Science, The Nature Conservancy, Honolulu, Hawaii, USA

Heather Cray, Graduate Student, School of Environment, Resources and Sustainability, University of Waterloo, Waterloo, Canada

Mark Dobrowolski, Adjunct Lecturer, School of Plant Biology, University of Western Australia, Perth, Australia

Joan C. Dudney, Graduate Student, Department of Environmental Science, Policy and Management, University of California, Berkeley, California, USA

Beatriz Duguy Pedra, Professor, Department of Plant Biology, University of Barcelona, Barcelona, Spain

Peter W. Dunwiddie, Affiliate Professor, School of Environmental and Forest Sciences, University of Washington, Seattle, Washington, USA

Stephen R. Edwards, Vice Chair, Resilience and Social Learning, IUCN Commission on Ecosystem Management, Baker City, Oregon, USA

Mirijam Gaertner, Research Coordinator, Center for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland, South Africa

Lauren M. Hallett, Postdoctoral Research Scholar, Department of Ecology and Evolutionary Biology, University of Colorado, Boulder, Colorado, USA

Brice B. Hanberry, Research Associate, GIS and Spatial Analysis Laboratory, Department of Forestry, University of Missouri, Columbia, Missouri, USA

Boze Hancock, Coordinator, TNC-NOAA CRP Partnership at The Nature Conservancy, Providence, Rhode Island, USA

Tibor Hartel, Associate Professor, Environmental Science Department, Sapientia Hungarian University of Transylvania, Cluj-Napoca, Romania

Liam Heneghan, Chair and Professor of Environmental Science and Studies, Institute for Nature and Culture, DePaul University, Chicago, Illinois, USA

Oisín Heneghan, Graduate Student, Department of Environmental Science and Studies, DePaul University, Chicago, Illinois, USA

Eric Higgs, Professor, School of Environmental Studies, University of Victoria, Victoria, British Columbia, Canada

Richard Hobbs, Professor, IAS Distinguished Fellow, School of Plant Biology, University of Western Australia, Crawley, Western Australia, Australia

Patricia Holmes, Ecologist, Environmental Resource Management Department, City of Cape Town, Cape Town, South Africa

Darwin Horning, Senior Community and Regional Planner, Interdisciplinary Program in Sustainability, University of British Columbia, Okanagan, British Columbia, Canada

Austin T. Humphries, Assistant Professor, Department of Fisheries, Animal and Veterinary Sciences, University of Rhode Island, Kingston, Rhode Island, USA

Lindsey B. Hutley, Professor of Environmental Science, Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia

Stephen T. Jackson, Director, Department of the Interior Southwest Climate Science Center, U.S. Geological Survey, Tucson, Arizona, USA

Theresa B. Jain, Research Forester, US Forest Service, Rocky Mountain Research Center, Moscow, Idaho, USA

Erik Jeppesen, Professor, Department of Bioscience, Aarhus University, Silkeborg, Denmark

John M. Kabrick, Research Forester, US Forest Service, Northern Research Station, University of Missouri, Columbia, Missouri, USA

Paul A. Keddy, Independent Scholar, Lanark County, Ontario, Canada

Todd Keeler-Wolf, Senior Vegetation Ecologist, California Natural Diversity Database, California Department of Fish and Game, Sacramento, California, USA

Benjamin O. Knapp, Assistant Professor, Department of Forestry, University of Missouri, Columbia, Missouri, USA

Timo Kuuluvainen, Principal Investigator, Department of Forest Sciences, University of Helsinki, Helsinki, Finland

David Lamb, Honorary Professor, School of Agriculture and Food Science, Center for Mined Land Rehabilitation, University of Queensland, Brisbane, Australia

Kemit-Amon Lewis, Coral Conservation Manager, The Nature Conservancy, US Virgin Islands, USA

Zhengwen Liu, Professor, Nanjing Institute for Geography and Limnology, Chinese Academy of Sciences, Nanjing, China

Stephanie Mansourian, Environmental Consultant, Mansourian.org, Gingins, Switzerland

Michael T. McTavish, Graduate Student, School of Environment, Resources and Sustainability, University of Waterloo, Waterloo, Canada

Alexander L. Metcalf, Research Assistant Professor, College of Forestry and Conservation, University of Montana, Missoula, Montana, USA

Elizabeth Covelli Metcalf, Assistant Professor, Department of Society and Conservation, University of Montana, Missoula, Montana, USA

Jakki J. Mohr, Regents Professor of Marketing and Gallagher Distinguished Faculty Fellow, School of Business Administration, Department of Management and Marketing, University of Montana, Missoula, Montana, USA

Ladislav Mucina, Professor Iluka Chair in Vegetation Science and Biogeography, School of Plant Biology, University of Western Australia, Perth, Australia

Sandra Cristina Muller, Adjunct Professor, Department of Ecology, Federal University of Rio Grande do Sol, Porto Alegre, Brazil

Stephen D. Murphy, Professor and Director of the School of Environment, Resources and Sustainability, University of Waterloo, Waterloo, Canada

Jessica Hardesty Norris, Technical Writer, Biohabitats Inc., Baltimore, Maryland, USA

Gerhard Ernst Overbeck, Professor, Laboratory of Grassland Vegetation, Department of Botany, Federal University of Rio Grande do Sol, Porto Alegre, Brazil

Stephen Packard, Ecological Restoration Pioneer and Visionary, Northbrook, Illinois, USA

Michael P. Perring, Research Associate, Ecosystem Restoration and Intervention Group, School of Plant Biology, University of Western Australia, Perth, Western Australia, Australia

Karel Prach, Professor, Department of Botany, Faculty of Science, University of South Bohemia and Institute of Botany, Czech Academy of Sciences, Ceske Budejovice, Czech Republic

Charles Price, Adjunct Lecturer, School of Plant Biology, University of Western Australia, Crawley, Western Australia, Australia

José M. Rey Benayas, Professor, Departamento de Ciencias de la Vida, Universidad de Alcalá, Alcalá de Henares, Spain

Jilliane Segura, Graduate Student, Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia

Benjamin Smith, Graduate Student, Earth and Environmental Dynamics Research Group, Department of Geography, King's College London, London, United Kingdom

Cecilia Smith-Ramírez, Professor, Department of Forestry Science and Natural Resources, University of Austral Chile, Valdivia, Chile

Martin Søndergaard, Senior Researcher, Department of Bioscience, Aarhus University, Silkeborg, Denmark

Andrew Spaeth, Forest Program Director, Sustainable Northwest, Portland, Oregon, USA

Erica N. Spotswood, Postdoctoral Research Scholar, Department of Environmental Science, Policy and Management, University of California, Berkeley, California, USA

Rachel Standish, Senior Lecturer in Ecology, School of Veterinary and Life Sciences, Murdoch University, Murdoch, Western Australia, Australia

John A. Stanturf, Senior Scientist, Center for Forest Disturbance Science, US Forest Service Southern Research Station, Athens, Georgia, USA

Katharine Suding, Professor, Department of Ecology and Evolutionary Biology, University of Colorado, Boulder, Colorado, USA

Peter Torok, Professor, Department of Ecology, University of Debrecen, Debrecen, Hungary

Libby Trevenen, Technical Assistant, School of Plant Biology, University of Western Australia, Crawley, Western Australia, Australia

Alberto Vilagrosa, Fundación CEAM, Joint Research Unit University of Alicante, Alicante, Spain

Carol L. Williams, Research Scientist, Center for Agroforestry, University of Missouri, Columbia, Missouri, USA

ACKNOWLEDGEMENTS

An edited volume like this one is very much a group effort. We are tempted to say a team effort, but the word team implies a group that is close-knit and has worked together for a long time towards a common goal. While the authors of the many chapters in this book share the common goal of understanding and advancing the practice of ecological and environmental restoration, we are certainly not a close-knit group. Many of the authors are frequent colleagues and friends of the editors, and via this handbook we have gotten to know many others who previously we knew only through publications and reputation.

First and foremost we must thank all of the authors of the chapters in this volume for their willingness to contribute a chapter despite no promise of any reward beyond the satisfaction of producing a good piece of work. We especially appreciate the kindness of strangers who worked with us despite not knowing us well or in person. All of the authors have all been extremely patient throughout the process of putting the book together and have quickly answered the many queries we had for them as we reviewed chapters and put everything together.

We extend a huge thank you to our editors at Routledge – Tim Hardwick and Ashley Wright. They have been encouraging, supportive, and have provided many excellent suggestions that helped improve the book. They have also been patient as we worked to get everything ready for publication. This book would never have been completed without their comfort and confidence in our ability to succeed with the project. We also thank Hamish Ironside for copy-editing the entire book. Special thanks to Karl Harrington and everyone at Fish Books, who did the typesetting of the handbook.

Finally, many, many thanks to our colleagues and families who have supported and encouraged us at every step of the way. We put this book together in the hope that it will inspire a new generation of restorationists so that our children and students will live in a world of beautiful, functional landscapes and ecosystems that benefit the entire planet, we humans and all of our fellow beings on this wonderful Spaceship Earth.