Bojie Fu
Mark Stafford-Smith *Editors*

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Preface

Drylands cover 41% of the global land surface and support around two-fifths of the global population. By definition they are a challenging part of the Earth's terrestrial environment due to low water availability, long dry spells, and a propensity for degradation that is hard to reverse; they are under additional stress due to ongoing climate change exacerbating already extreme conditions for habitability. Yet their immense area means that drylands are a significant contributor to global function, through their role in climate regulation and carbon storage, their reservoir of biodiversity and their impact on such a large population. Extended periods of limited water availability result in great vulnerability to global climate changes and anthropogenic disturbances in drylands, and the relatively low human population density means that dryland social–ecological systems (SESs) are often distant from centres of governance, business and learning. Around half the human inhabitants of drylands are directly dependent on ecosystem productivity for their livelihoods, making these systems harbingers of the impacts of global environmental change.

SESs are complex adaptive systems that are constituted by interactions between diverse people and elements of diverse ecosystems. In the face of the dynamic interactions and feedbacks among the human and nonhuman elements of an SES, science still lacks the analytic tools to synthesize knowledge about SESs into explaining and exploring their social–ecological interactions and processes. In particular, research on the structure and function of dryland SESs has not received sufficient attention worldwide.

Given the speed and intensity of climate change and socioeconomic development, both of which are likely to aggravate issues such as land degradation, poverty, food and water insecurity in drylands, systematic research on the social and ecological processes and their interactions in dryland SESs is essential. This research must operate across sectors, scales, actors in society and countries to capture synergies among Sustainable Development Goals and manage conflicts that may arise due to tradeoffs between goals. The extent of the drylands, both physically and socially, means that the SDGs cannot be achieved globally unless they are also achieved in the drylands.

This collective book has been prepared by a joint working group committed to critical research on dryland SESs, as a timely synthesis of up-to-date knowledge in various thematic fields relevant to dryland SESs. It is meant to organize key salient concepts and establish a conceptual framework relevant to the interdisciplinary and cross-cultural understanding of dryland SESs, acknowledging their diverse geographical and social–ecological contexts and structures. Through synthesizing research across the world and reviewing scientific evidence for good practices, it aims to promote collaboration among researchers globally as well as communication with policymakers, managers and practitioners for sustainable dryland ecosystem management.

Beijing, China
Canberra, Australia
January 2023

Bojie Fu
Mark Stafford-Smith

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About the Editors

Bojie Fu is a Distinguished Professor at the Research Centre for Eco-Environmental Sciences of Chinese Academy of Sciences (CAS). He obtained his Ph.D. from a split Ph.D. programme in Peking University (Beijing, China) and University of Stirling (Stirling, UK) in 1989. He has been elected as an Academician of CAS, International Honorary Member of the American Academy of Arts and Sciences, a Fellow of the Academy of Sciences for Developing World (TWAS) and a Corresponding Fellow of the Royal Society Edinburgh UK. His research areas are land use and land cover change, landscape pattern and ecological processes, ecosystem services and management. He serves as the Vice President of the International Geographical Union (IGU) and the member of the multidisciplinary expert group on the Intergovernmental Science-Policy Platform for Biodiversity and Ecosystem Services (IPBES). He has published more than 500 scientific papers and 12 books, including *Nature*, *Science*, *Nature Geoscience*, *Nature Climate Change*, and *Nature Sustainability*. He has won the 2022 TWAS-Lenovo Science Award, the Tan Kah Kee Science Award-Earth Science, the Alexander von Humboldt Medal of EGU, the China National Natural Science Prize, the Outstanding Science and Technology Achievement Prize of CAS and the Ho Leung Ho Lee Science and Technology Prize-Geosciences.

Mark Stafford-Smith is based in Canberra, Australia, and contributes to research on adaptation and sustainable development. He has recently retired from CSIRO, Australia's national research organization, where he had been oversighting a highly interdisciplinary programme of research on many aspects of adapting to climate change, as well as regularly interacting with national and international policy issues around sustainable development. He has over 30 years' experience in drylands systems ecology, management and policy, including senior roles such as CEO of the Desert Knowledge Cooperative Research Centre in Alice Springs. His significant international roles include being past vice-chair of the International Geosphere-Biosphere Programme's Scientific Committee; co-chair of the 2012 Planet Under Pressure: New Knowledge Towards Solutions conference on global environmental change in the lead-up to Rio+20; and Chair (2013–2017) of the inaugural Science Committee for Future Earth, which helps to coordinate research towards global sustainability worldwide. He continues to publish, adding to over 200 peer-reviewed contributions to science, as well as many presentations and publications for less-specialized audiences.

Chapter 1
The Global-DEP: A Research Programme to Promote Sustainability of Dryland Social-Ecological Systems

Bojie Fu, Mark Stafford-Smith, Chao Fu, Yanxu Liu, Yanfen Wang, Bingfang Wu, Xiubo Yu, Nan Lu, and Dennis S. Ojima

Abstract In light of the escalating pace and heightened intensity of contemporary climate change and human interventions, a more systematic and comprehensive approach to research has become imperative for the realization of the Sustainable Development Goals (SDGs) within dryland regions. In 2017, a collaborative research consortium comprising experts from diverse nations proposed the Global Dryland Ecosystem Programme (Global-DEP). This initiative was designed to address the intricate challenges inherent in the diverse and fragile social-ecological systems (SESs) of drylands. Drawing from a synthesis of preceding studies on dryland SESs

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and insights garnered from extensive regional consultations, the consortium crafted the conceptual framework of Global-DEP, with SESs as its fundamental underpinning. Key elements of the frame-work encompass driving forces, impacts, feedback loops, and scale. The team identified four pivotal themes: (1) dryland SES dynamics and driving forces, (2) dryland SES structure and functions, (3) dryland ecosystem services and human well-being, and (4) ecosystem management and sustainable livelihoods in drylands. The intricate interconnections among these themes were meticulously examined to delineate 12 critical research priorities. Anchored upon this conceptual framework and the identified research imperatives, the Global-DEP science plan was formulated. This plan is poised to expedite actionable interdisciplinary research within dryland SESs, tailored to the regional and cultural nuances of these areas. The final aim is to bolster dryland research endeavors, catering to the requirements of land practitioners and policymakers, while effectively contributing to the attainment of SDGs in drylands.

Keywords Global-DEP · Drylands · Social-ecological systems · Drivers Structure and functions · Ecosystem services · Ecosystem management · Livelihoods

1.1 An Overview of Drylands and SDGs

Drylands encompass land areas characterized by a mean annual precipitation to mean annual potential evapotranspiration ratio (known as the aridity index) below 0.65. The aridity index defines four distinct dryland subtypes: hyper-arid (aridity index < 0.05), arid (0.05 ≤ aridity index < 0.20), semi-arid (0.20 ≤ aridity index < 0.50) and dry sub-humid (0.50 ≤ aridity index < 0.65). This definition classifies drylands as covering approximately 41% of the Earth's land surface, sustaining diverse ecosystems that deliver essential goods and services to over 2 billion inhabitants residing in these regions (Millennium Ecosystem Assessment (MEA) 2005).

Drylands are a critical part of the Earth's systems functioning due to their contribution to the global carbon cycle and their role in climate regulation both regionally and globally, as well as being a major reservoir of biodiversity (including the original genotypes of many key cereals) and host to immense human cultural diversity (Buisson et al. 2022; Castro et al 2018; Maestre et al. 2022; Safriel et al. 2005; Wang et al. 2022). Their ability to deliver these services compared to other terrestrial environments is challenged due to low water availability (Prăvălie 2016), long dry spells (Wang et al. 2012), and hard to recovery from degradation due to the reduced social-ecological resilience (Cowie et al. 2018; Stafford-Smith et al. 2009). The hydrological balance plays a central role in dryland regions (Verstraete et al. 2009). Extended periods of limited water availability result in sparse vegetation cover with great temporal and spatial fluctuation, and great vulnerability to global environment changes and anthropogenic disturbances (Safriel & Adeel 2008). An estimated 1 billion of dryland human inhabitants depend directly on ecosystem services for their livelihoods; despite being attuned to the challenges of dryland conditions when undisturbed, this population rapidly becomes vulnerable when these challenges are exacerbated—becoming a 'canary in the coalmine' for global change.

Dryland ecosystems offer a wealth of ecosystem goods and services for human well-being (Safriel et al. 2005; Stafford-Smith et al. 2009). Ecosystem services (ES) in drylands are water constrained, highly variable, and vulnerable to environmental changes; and there are clear trade-offs and synergies among ES such as water supply, food production and regulation services such as carbon fixation and soil conservation (D'Odorico et al. 2013). Water crises, land degradation and desertification are pervasive and have the potential to lead to a collapse of life support systems in the absence of appropriate conservation and utilization strategies. This presents profound implications for the livelihoods of marginalized communities on a local scale and can also trigger migration, unrest, and economic instability at regional and global levels—extending well beyond the boundaries of dryland areas.

Over recent decades, international scientific programmes and initiatives have addressed drylands as part of their mandates. The UNESCO's Man and the Biosphere Programme (MAB) in its early years took the arid and semi-arid zones as one of its focal ecosystem types and developed a plenty of projects in different regions, especially in Africa (Vannucci 1982). The Millennium Ecosystem Assessment specifically assessed the magnitude of desertification (i.e., land degradation in drylands) and its causal factors (Millennium Ecosystem Assessment (MEA) 2005), significantly warning that the conditions of global dryland ecosystems can deteriorate due to feedback loops between desertification, climate change, and biodiversity loss. The United Nations Food and Agriculture Organization (FAO) published reports on cereal production, forest and land use change in dryland (Koohafkan & Stewart 2008; Food and Agriculture Organization (FAO) 2016).

The year 2015 witnessed a pivotal milestone with the United Nations' adoption of "Transforming our World: The 2030 Agenda for Sustainable Development," delineating 17 Sustainable Development Goals (SDGs) (United Nations (UN) 2015). This framework provides a structured approach to balance essential human needs derived from ecosystem services (ES), such as food, water, and energy security, with human developmental aspirations encompassing poverty eradication, health, equity, education, and livelihoods. Addressing these complex trade-offs and potential synergies across values and governance domains, including infrastructure, urban development, and consumption patterns, has been a key consideration (Fu et al. 2019).

Drylands emerge as a pivotal resource both vital and interconnected in the attainment of the 2030 Agenda (Stafford-Smith & Metternicht 2021). SDG target 15.3 stands as a unique global objective, aiming to achieve land degradation neutrality (LDN) by 2030. The objective seeks to maintain or enhance the conservation of natural capital linked to land resources and the ecosystem services they provide. As a result, a systematic strategy becomes imperative for meeting human needs while sustaining the ecosystems and the benefits they yield within drylands across the globe. Moreover, the SDG target 15.3 cannot achieve only for itself considering the high links among SDG15 (Life on Land), SDG13 (Climate Action), SDG6 (Clean Water and Sanitation), SDG1 (No poverty), SDG2 (Zero Hunger), and other pertinent SDGs in the context of drylands (Yao et al. 2021). Thus, the success of SDG

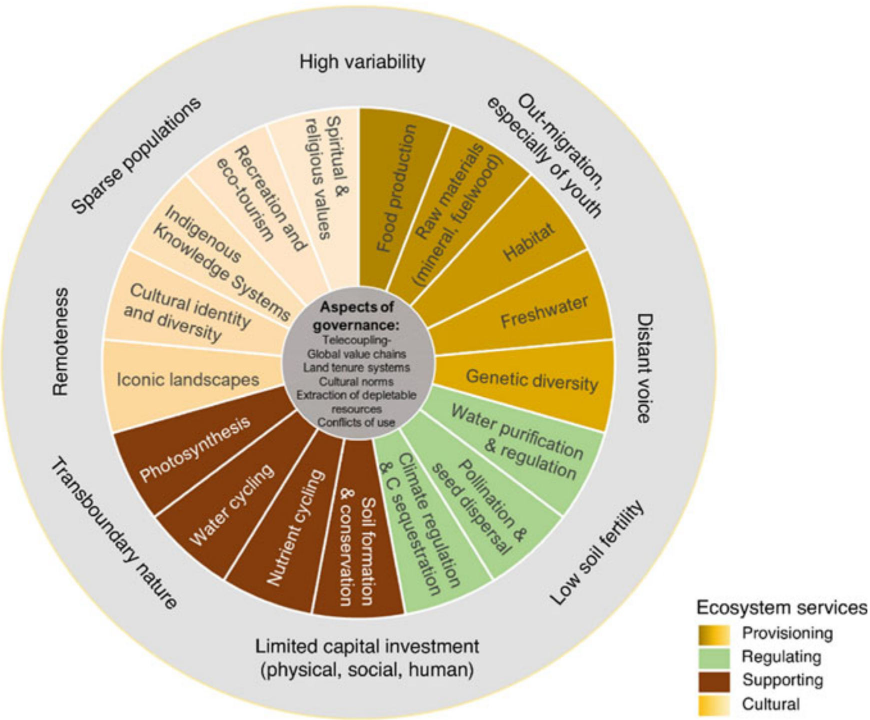

Fig. 1.1 More-or-less depletable services and resources delivered by drylands, loosely classified into the four categories of ecosystem services (wheel spokes), surrounded by key shared attributes of drylands (wheel tyre), and surrounding key aspects of governance needed for a GEC (wheel axle) (Stafford-Smith and Metternicht 2021)

15.3 hinges on addressing numerous other SDGs, including poverty, hunger, water access, energy, climate, and broader issues of equity, peace, and prosperity (Stafford-Smith & Metternicht 2021). The SDGs offer an optimal framework for navigating the intricate landscape of potential synergies and trade-offs encompassing the diverse array of resources and services offered by drylands (Fig. 1.1).

1.2 Recent Developments in Dryland SES Research

Socio-ecological systems (SESs) are complex adaptive systems arising from dynamic interactions between ecosystems and human societies (Folke et al. 2016; Preiser et al. 2018). In dryland regions, human inhabitants draw upon local ecosystems to extract diverse resources, ranging from water to food, all in service of enhancing human well-being. The management of these ecosystems is profoundly influenced by an array of factors, including governmental policies, subsidies, payments for ecosystem services, and markets spanning local to global scales. These social processes hold pivotal significance, shaping the very fabric of SESs in drylands—encompassing their structure, attributes, and intricate interactions (Maestre et al. 2016). While the ramifications of climate change reverberate globally, adaptive strategies predominantly manifest at the local or regional level, necessitating the holistic consideration of ecological, social, and economic stimulants and responses inherent to specific SESs, particularly within dryland contexts (Scheffer et al. 2015). Evident shifts in the functioning of ecosystem goods and services serve as society's barometer of ecosystem change, potentially inciting societal reactions that, in turn, exert further impacts on ecosystems, thereby triggering a cycle of iterative feedback and response (McCollum et al. 2017).

Drylands are thus strongly coupled SESs, which are heavily influenced by people and by global change, with complex social-ecological interactions and feedbacks across scales (Reynolds et al. 2007). In light of this, the sustainability of dryland SESs necessitates a comprehensive approach rooted in an understanding of the dynamic interplays between nature and society. This entails an equal emphasis on the ways in which social transformations mold the environment, and conversely, how environmental shifts shape societal dynamics (Clark and Dickson 2003). This understanding extends to encompass social institutions, cycles, and order (Redman et al. 2004). Here we build on the recent development of dryland SES research through four lenses: SES dynamics and drivers, SES structure and function, ecosystem services in SES, and sustainability of SES.

Between 1991 and 2005, global drylands expanded by 4%, as highlighted by Feng and Fu (2013). Projections under the pessimistic climate change scenario (RCP8.5) suggest a further 23% increase in global dryland expansion by 2100, potentially accounting for 56% of the total global land area (Huang et al. 2016). The dynamics of drylands are intricate, characterized by multifaceted patterns encompassing both linear and nonlinear, gradual and abrupt shifts. These transformations are propelled by intricate interplays between biophysical and socio-economic factors, all underpinned by fundamental drivers that encompass abiotic elements (e.g., climate and soil properties), attributes of biological communities (e.g., diversity and spatial patterns), and human activities (e.g., grazing and agriculture) (Ruppert et al. 2015; Maestre et al. 2016). Many dryland landscapes have undergone marked degradation, often transitioning from productive vegetation-pattern states to barren, unproductive conditions (Zelnik et al. 2013). Widespread catastrophic shifts have been documented in dryland landscapes globally (Berdugo et al. 2017). Climate change exacerbates negative impacts on vegetation diversity and coverage, while disruptions in species interaction networks and suboptimal management practices—some of which manifest slowly—compromise the landscape resilience of dryland SESs in the face of extreme events (Hoover et al. 2014). Given the sparse nature of dryland vegetation, the efficacy of vegetation indices in reflecting actual changes becomes compromised, leading to ambiguous outcomes. A notable instance occurred between 1982 and 2013, when an increased global vegetation index masked the stark fact that actual vegetation had, in fact, declined on a global scale (Pan et al. 2018).

With the projected escalation in aridity and the anticipated rise in the frequency of drought occurrences across global drylands, the prevalence of abiotic factors governing land degradation, especially hydrological and aeolian soil erosion processes, could intensify (Ravi et al. 2010). The foreseen increase in aridity linked to climate change stands to adversely affect the multifaceted functions and services furnished by dryland ecosystems worldwide (Delgado-Baquerizo et al. 2016). Such amplified aridity levels have the potential to exacerbate soil erosion, land degradation, and desertification (Reynolds et al. 2007; Feng and Fu 2013). The employment of dynamic modeling techniques emerges as essential for gaining valuable insights into comprehending the trajectories of future dynamics within dryland SESs and the fundamental driving mechanisms steering these changes (Pelletier et al. 2015).

The intricate interplay between structure and function across various spatial scales unveils how SESs respond to the ongoing wave of global transformations, simultaneously playing a fundamental role in determining state shifts within drylands (Maestre et al. 2016; Mayor et al. 2013; Saco et al. 2018). The structures and functions of drylands, and how they interact may change significantly, even leading to shifts among alternative stable states (D'Odorico et al. 2013). When a critical threshold is crossed, SESs can undergo catastrophic change and reorganize into a different state (Angeler and Allen 2016; Turnbull and Wainwright 2019). However, the mechanisms that underlie the interactions between structure and function, and the resulting impacts on the state of SES are still controversial and poorly understood (Loreau and Mazancourt 2013). We must handle the complexity caused by multiple feedbacks among biotic and abiotic elements (Mayor et al. 2013; Turnbull et al. 2012), by interactions between structures and functions (Saco et al. 2018; Turnbull et al. 2012), and by the scale issues that challenge our ability to reveal how the structure and function of dryland SESs evolve (Berdugo et al. 2017). Climate changes usher in changes in nutrient input and loss rates, rates of plant photosynthesis, grazing patterns and intensities, soil fertility depletion, temporal and spatial water availability reductions, and the occurrence of dust storms; these extreme climatic events can even swiftly reshape landscape configurations (Lucatello et al. 2020). Consequently, abrupt or even catastrophic shifts in dryland SESs, accompanied by corresponding losses or gains in ecological and economic assets, might occur (Ursino 2019). However, the realm of predicting and confirming abrupt responses to a changing environment remains inadequately explored, leaving landscape response to stress highly variable and unpredictable (Zelnik et al. 2013). Regime shifts within dryland SESs can emerge from gradual or rapid reactions to alterations in external drivers and feedback loops, culminating in gradual, abrupt, or catastrophic outcomes (Saco et al. 2020). Although regime shifts within single dimensions are often addressed on an ecosystem scale due to the relatively straightforward relationships between variables, the nonlinearity, intricacies of feedback systems, and the presence of behavioral thresholds in dryland SESs render comprehensive and realistic predictions challenging (Burthe et al. 2016).

Beyond the provisioning of essential services like food, freshwater, and fuel, the critical regulating services such as soil conservation, hydrological regulation, and cli-mate regulation, alongside the cultural services offered by the distinctive biodiversity, ecosystems, and landscapes of drylands, stand as paramount indicators of human well-being within dryland SESs (Fu et al. 2013). Global shifts in the environment have profoundly reshaped the provision of ecosystem services in drylands, along with their intricate supply-and-demand dynamics and the inherent trade-offs that manifest across diverse scales (Lu et al. 2018). To quantitatively assess alterations in dryland ecosystem services across spatial and temporal scales, an array of mapping and scenario analysis tools have been devised for regional simulations of ecosystem services (Hu et al. 2015; Smith et al. 2011). The interactions within dryland SESs encompass multifaceted dimensions, encompassing service types (e.g., food, water, energy, and services related to ecological security), beneficiaries (e.g., farmers, retailers, and environmentalists), locations (e.g., upper or lower reaches of watersheds), and temporal periods or generations (Seppelt et al. 2011). These interactions are further shaped by public infrastructure elements like roads, dams, drinking water pipelines, and cultural amenities, which facilitate residents in remote regions to access the supply or transportation of local ecosystem services to areas with demand beyond the realm of drylands (Castro et al. 2014; Miyasaka et al. 2017).

Human well-being emerges as a state intricately intertwined with specific environmental conditions. It encompasses material circumstances, freedom of choice, health, social relations, security, inner tranquility, and spiritual experiences—all essential for maintaining a high quality of life (Summers et al. 2012). Through intensive land use practices encompassing cultivation, grazing, deforestation, resource extraction, and excessive utilization of freshwater resources, human activities within dry-lands can potentially induce various forms of land degradation and water resource deterioration. These impacts are often exacerbated by climate change, leading to consequential effects on the delivery of ecosystem services (D'Odorico and Bhattachan 2012). Elevated levels of human well-being can indirectly yield benefits to ecosystem services, as the adverse consequences on ecosystem services are frequently mediated by institutional, cultural, and governance factors, along with conflicts. These mediating factors might operate more effectively at higher levels of human well-being (Lucatello et al. 2020).

Under the influence of degrading factors, dryland ecosystem services come under pressure, curbing human access to necessities like food, water, energy, and ecological security, thereby compromising sustainable livelihoods to varying extents across distinct dryland SESs (Keesstra et al. 2018). In recent times, nature-based solutions (NBS) have gained prominence as approaches aimed at safeguarding, sustainably managing, and restoring natural or altered ecosystems to effectively and adaptively address societal challenges. NBS stands as a prospective framework to reverse the trajectory of degradation evident in dryland ecosystems, which threatens both biodiversity and human well-being. NBS aligns conservation and development objectives, offering a pathway to counteract the detrimental effects of degradation (Cohen-Shacham et al. 2019; Keesstra et al. 2018).

The primary biophysical constraints challenging the sustainability of dryland SESs encompass natural resource limitations and ecosystem degradation, with high emphasis on water scarcity and encroaching desertification (Huber-Sannwald et al. 2012). Social and economic constraints, such as limited access to markets and resources, weak governance structures, and inadequate information about alternative production technologies, further curtail the available options for inhabitants of drylands (van Ginkel et al. 2013). The disparity between the supply and demand of ecosystem services in drylands stands as a significant hurdle for landowners, producers, land managers, land use planners, and policymakers. This challenge is amplified as land quality sits at the juncture of ecosystem functioning and human security, encompassing vital elements like clean water, air, food, and energy—the bedrock of livelihood development in dryland SESs (Reed et al. 2015). Consequently, there exists a pressing need to guide and facilitate transdisciplinary and participatory research efforts aimed at combating land degradation and harmonizing dryland ecosystem services. This calls for collaboration from all stakeholders, including academia, governmental and nongovernmental organizations, civil societies, local stakeholders, and policymakers, with a goal to foster collective knowledge generation, continuous system monitoring, reevaluation, and capacity enhancement in dryland stewardship across all tiers (Challenger et al. 2018).To cultivate resilient livelihoods within dryland SESs, innovative approaches are essential from all participants—ranging from primary producers to policymakers—to identify, quantify, and address the driving forces and interactions that shape and constrain the development and progression of dryland livelihoods (King et al. 2018).

Good governance in drylands involves institutions for decision making by a range of stakeholders, including individuals, both in formal positions of power and as 'ordinary' citizens, households, communities and organizations (Lopez-Porras et al. 2018). Building capacity in education, health, gender equality, technology, and comprehensive analysis is also closely related to promotion of dryland SES governance (Reed and Stringer 2016; Cherlet et al. 2018; Middleton 2018). This in turn helps regions and countries ensure future water, food, energy, and ecological security, to mitigate climate change, and to advance the capacity for good governance (Griggs et al. 2013). The SDGs can be regarded as a major governance instrument to combat desertification, drought, and land degradation that combine and scale up established socioeconomic principles (Rica et al. 2018); and the logic of analysing the interconnections between SDGs permits the potential to mainstream sustainability (Bautista et al. 2017).

1.3 Global-DEP and Its Conceptual Framework

In recent decades, a plenty of frameworks for understanding Social-Ecological Systems (SESs) have been put forth, with Ostrom's framework standing out as one of the most widely employed (Ostrom 2009; McGinnis and Ostrom 2014). The SES framework offers valuable insights into evaluating the intertwined social and ecological facets contributing to sustainable resource utilization and management. This framework can be applied in a spatially explicit, quantitative manner to identify opportunities and trade-offs when striving for the sustainability of interconnected SESs (Leslie et al. 2015). Several other frameworks have also emerged, each engaging with varying aspects of the SES perspective. One such framework is the Composition-Structure-Process-Service framework, designed to dissect the underlying mechanisms driving Ecosystem Services (ES) production. Functioning as an application-oriented linking framework, it bridges landscape patterns, ecosystem processes, and ES, while also embracing landscape design for sustainable ecosystem management across different scales (Fu et al. 2013). An enhanced iteration of this framework, named the Pattern-Process-Service-Sustainability framework, has been refined to incorporate the dynamics of interconnected natural and human systems (Fu & Wei 2018). Another integrated framework synthesizes the core tenets of the ES cascade concept and the Driver-Pressure-State-Impact-Response (DPSIR) framework. This amalgamation aims to position ES within a broader SES context, encompassing the cycle of ES provision, societal feedback, and analytical depiction of social-ecological interactions. It aims to serve as a valuable instrument for policy development that promotes the sustainability of dryland ecosystems and thereby safeguards the livelihoods of their associated users (Nassl and Löffler 2015).

The Dryland Development Paradigm (DDP), introduced by Reynolds et al. (2007), has gained considerable influence as a guiding framework for dryland development. Drawing on empirical analyses within dryland systems science, Stringer et al. (2017) derived an updated version of the DDP (DDP#2). This iteration comprises three integrative principles and advocates a shift away from a research-for-development approach. The DDP emphasizes the need to always consider both human and environment aspects of dryland systems, but also to avoid careless generalization, highlighting for research to be concerned with the diversity of global drylands and their social-ecological characteristics. For example, Safriel et al. (2005) highlighted the interrelationships between major ES, between ES and biodiversity, and between ES and the livelihoods that ecosystems support across the aridity gradient. As another example, Stafford-Smith et al. (2011) formalized a conceptual systems model of key migration processes in drylands globally, which recognizes a series of factors at local and broader scales that contextually affect how critical ES are to local livelihoods and how these then interact with what adaptive capacity households may have to stay or move. Furthermore, Huber-Sannwald et al. (2012) amalgamated the DDP and other conceptual frameworks, coupling them with an exhaustive analysis of biophysical, socio-economic, and historical data. Their study assessed challenges and opportunities for livelihood development within the Amapola dryland ecosystem, a semi-arid region in Mexico. Their findings called for an effective, flexible, and viable policy framework that could enhance the biotic and cultural diversity of drylands locally, ultimately transforming drylands across the globe into a resilient biome, in the face of global environmental and social shifts.

The Global Dryland Ecosystem Programme (Global-DEP) was approved as a key international cooperation project under the International Partnership Program (IPP) of Chinese Academy of Sciences (CAS) in August 2017. It is an international cooperation initiative jointly proposed by Prof. Bojie Fu from the CAS and Dr Mark Stafford-Smith from the Commonwealth Scientific and Industrial Research Organization (CSIRO) in Australia, with an aim of developing an actionable research

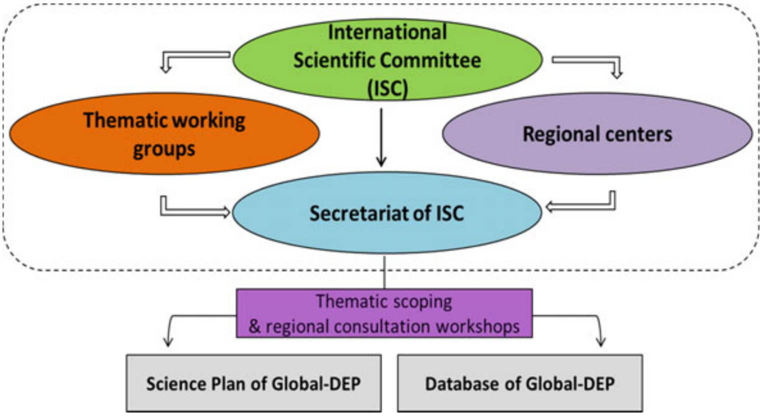

Fig. 1.2 Global-DEP organizational structure

plan to address the challenges facing diverse and fragile dryland SESs. A Scientific Committee was established to orchestrate the development of the program's Science Plan, and a dedicated Secretariat was put in place to provide essential technical support. In addition, the program created four thematic work groups and five regional work groups, featuring principal investigators from CAS as well as counterparts from nations such as the United States, Spain, Senegal, and Australia, among others (Fig. 1.2).

The conceptual framework of the Global-DEP was meticulously crafted by amalgamating insights from diverse disciplines and examining previous frameworks. This framework, grounded in the perspective of SES, underscores the imperative of comprehending several pivotal components. These encompass the drivers shaping SES, the intricate interplay of SES structure and functions, the critical realm of ecosystem services and its impact on human well-being, and the management responses required to actualize the SDGs. The framework draws attention to the interlinked and multi-scale nature characterizing dryland SESs, an insight resonant with the DDP. This recognition culminates in the proposition of a cohesive quartet of research themes, propelled by the forces of global environmental transformations and globalization. These research themes are strategically oriented towards achieving SDG objectives through a dynamic interplay of responses and feedback loops weaving together ecological and social facets (Fig. 1.3). This fundamental framework, though presented in a simplified manner, was further expanded upon by Fu et al. (2021). Its significance lies in its ability to engage researchers spanning ecology and social sciences, both converging on the realm of dryland SESs. Additionally, this framework provides the bedrock for the formulation of both the scientific and actionable agendas of Global-DEP.

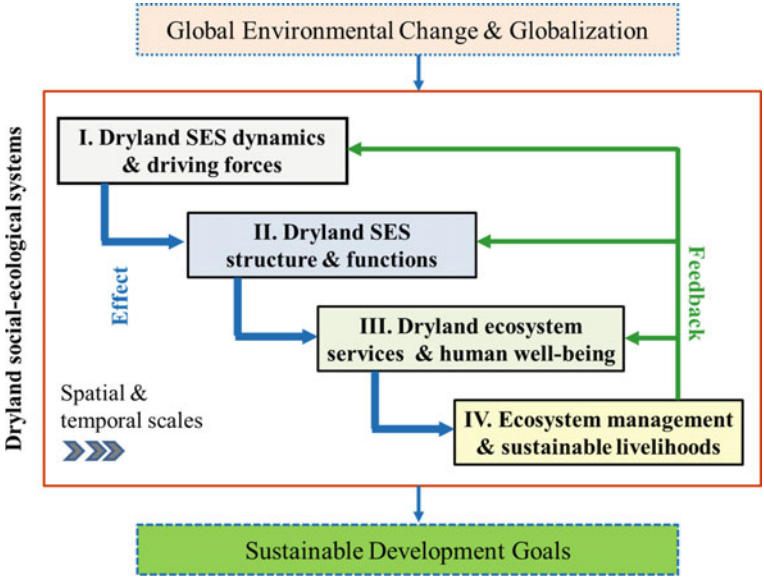

Fig. 1.3 Simplified diagram of Global-DEP conceptual framework

1.4 Research Themes and Priorities

Based on the overarching framework of helping dryland SESs meet the SDGs, each of the four themes raises specific research priorities as described below.

Theme I: Dryland social-ecological system dynamics and driving forces

The dynamics inherent in dryland SESs are a product of the intricate amalgamation of diverse linear and non-linear patterns, coupled with both gradual and sudden shifts. These dynamics are propelled by an interplay of biophysical and socio-economic factors. This thematic exploration seeks to unveil the critical variables essential for comprehending these large-scale dynamics, thereby fostering an overarching understanding of the distinctions among distinct dryland SESs. Such insights serve as a fundamental platform for discerning transferrable findings across different locales and projecting the trajectories of pivotal drivers shaping SES dynamics in other thematic domains.

Research priority 1.1: what are the essential dryland variables (EDVs) of the macroscopic dynamics of dryland SES?

Essential variables are the minimum set of variables required to characterize change in a system (Reyers et al. 2017). Essential variables for climate, biodiversity, water, socio-ecological systems and SDGs have been proposed successively in recent years (Reyers et al. 2017). Social-ecological activities in drylands are dominated by water availability; and the responses of dryland SES to climate change and anthropogenic disturbances can be reflected by changes in land cover (Maestre et al.

2016). Dryland landcover is particularly characterized by sparse and patterned vegetation and soil biocrusts. Research to identify these sensitive essential variables and to enhance the monitoring of their dynamics is essential to underpin understanding of the driving forces behind them (Li et al. 2021), and to improve management of dryland SES.

Research priority 1.2: what are the driving forces of the macroscopic dryland SES dynamics?

Climate change and human activities notably loom as pivotal drivers of dryland SES dynamics, amplifying the risks of land degradation and desertification (MEA 2005). Moreover, dryland SESs are usually water-limited by definition. Remote sensing technology provides many key water-related products that can assist the macroscopic study of dryland SES dynamics, including patterns over space and time of soil moisture, precipitation, evapotranspiration, water stress of vegetation, and evapotranspiration partition (Wang et al. 2012). As an entry point to understand the contextualized contributions of climate change and human activities, the research frontier is to identify how these factors together determine the development and degradation of drylands across spatiotemporal gradients of water availability.

Research priority 1.3: what are the future trajectories of macroscopic changes in dryland SES?

Extreme climate events will become more frequent, widespread and intensified under projected trends of global warming, resulting in significant changes in dryland (Huang et al. 2017). With population growth, human activities, such as grazing, also impose greater pressures on dryland SES. Although a variety of models have been proposed and applied to simulate land use transformations in drylands, there is still a high uncertainty across models and scenarios. Tackling the intricate questions underlying future dryland SES trajectories necessitates predictive work encompassing varied climate scenarios, human interventions, and desertification trends based on observed trends in the foundational EDVs.

Theme II: Dryland social-ecological system structure and functions

Intrinsic to the stability and resilience of SESs in drylands are the intricate inter-plays of their structures, functions, and interactions. A comprehensive grasp of state shifts in local dryland SESs goes beyond predictions based solely on isolated indicators due to the substantial spatiotemporal variations, sensitivity, and vulnerability to natural and human-induced disturbances. This thematic exploration strives to uncover the intricate biotic and abiotic mechanisms governing regime shifts in dryland SESs. By adopting both comprehensive and context-specific viewpoints, this theme aims to elucidate how these SESs evolve under diverse circumstances, addressing queries about tipping points and alterations in regimes that could have profound ramifications for the provisioning of ES across varied dryland SESs.

Research priority 2.1: how do ses structure, functions and their interactions change in drylands?

Understanding interactions between the structure and functioning of dryland SES at multiple spatial scales can substantially improve our understanding of how drylands respond to ongoing global environment changes. The ecosystem structure of drylands interacts with function through multiple feedbacks, particularly hydrological feedbacks (D'Odorico and Bhattachan 2012). Therefore, connectivity, scale, and threshold behavior in hydrological systems are of common concern in dryland landscapes. The research frontier is revealing how ecohydrological and socioeconomic processes drive the evolution of SES structures, functions, and their interplay in diverse and scale-dependent dryland contexts.

Research priority 2.2: how do dryland SES structures and functions respond to climate change?

The intricate interplay between structure and function across various spatial scales affords insights into the SES responses to global transformations and how these dynamics underpin shifts in SES states (Fu et al. 2021; Maestre et al. 2016). Given the geographical heterogeneity inherent in different dryland SESs, predicting the trajectories of local dryland changes necessitates an in-depth comprehension of the mechanisms and resilience maintenance strategies in the face of climate change. This entails exploring how these structures and functions recalibrate under shifting climatic conditions.

Research priority 2.3: what is the SES mechanism for regime shifts in drylands?

Regime shifts, irreversible or sustained alterations, often bring detrimental impacts to drylands (Scheffer et al. 2015). These shifts can stem from gradual changes or swift responses to external drivers and feedbacks. Addressing these shifts necessitates a deep dive into the context-specific social-ecological feedback loops embedded in drylands, where threshold behaviors come to the forefront. Fostering a comprehensive understanding of these shifts involves developing holistic indicators, models, and multi-variable approaches capable of prognosticating the likelihood of regime shifts in dryland SESs moving forward.

Theme III: Dryland ecosystem services and human well-being in a changing environment and society

The intricate relationships between ES and human well-being in diverse dryland settings present a complex challenge. Discerning the dimensions of human well-being most pertinent to dryland ecosystems, as well as how changes in ES impact well-being within specific SESs, stands as the core objective of this theme. This exploration seeks to identify pathways that harness the value of ES for livelihood enhancement, catering to a wider array of beneficiaries both within and beyond SESs. This theme is set to propel the necessity for comprehensive monitoring to prevent the occurrence of collapse thresholds and amalgamate context-specific insights into the connections between ES and human well-being, thus influencing local management and policy choices in drylands.

Research priority 3.1: how do dryland ecosystem services change across space and time?

Dryland ES have high spatial and temporal variability due to the high variability in natural and social conditions, such as ecosystem type, climate, extreme events or disturbances, and economic development level. Enhancing our ability to model and predict the changes in these services across different scales in space and time is pivotal. This involves refining model structures, incorporating modules or parameters that account for the unique characteristics of dryland ecosystems, and generating more reliable estimates of ES at the local level. The research frontier includes biophysical modeling of ES at multiple scales, ES valuation not limited to monetary value, identification on the key drivers of ES change, and then simulating ES change in future scenarios.

Research priority 3.2: what are the interactions between multiple ecosystem services and supply–demand relationships?

Understanding the trade-offs and synergies resulting from interactions among various ES is essential for devising adaptable land use strategies within dryland SESs. Due to the spatial heterogeneity of ecosystems and population distribution in drylands, both the supply and demand of ES have high spatial variability (Castro et al 2014). With spatially heterogeneous and temporally dynamic human needs, the trade-offs between ES and people can be exacerbated, causing complex interactions among multiple beneficiaries, locations, and human generations. Therefore, the research frontier is to explore all the potential tradeoffs among the multiple dimensions of human demand for ES, particularly considering the future needs for ES under dryland environmental change; as well as to understand the supply–demand mismatches of dryland ES at different scales, and then track the potential dryland ES flows that depend on socioeconomic and environmental teleconnections.

Research priority 3.3: how are dryland ecosystem services linked to human well-being?

Clarifying how changes in ES alter their contribution to human well-being is key to the entangled dryland challenges, and to promoting the resilience of these SESs and finding solutions that balance ecological protection and socioeconomic development. This entails deciphering the ideal blend of natural and social capital for fostering well-being and understanding how other forms of capital, like technology and infrastructure, play a role in bolstering ES within dryland SESs. The research frontier is to understand the pathways and mediating factors that enable ES to deliver human well-being, to quantify the relationship between ES and human well-being, to optimize landscapes to produce ES, and to understand how best to provide payment for ES.

Theme IV: Ecosystem management and sustainable livelihoods in drylands

The immense diversity of global drylands – encompassing varying environments, degradation levels, social and cultural dimensions, and human reliance – underscores the necessity for nuanced and contextually-tailored management objectives and strategies. This theme is designed to forge connections between community development and ecosystem management, ensuring the attainment of SDGs within dryland SESs. Drawing upon insights from other themes, it aspires to proffer management and policy alternatives, while simultaneously pinpointing the EDVs, a contextual grasp of tipping point dynamics in ES provisioning, and the pathways by which these services translate into human well-being across distinct geographical contexts.

Research priority 4.1: how can sustainable ecosystem management schemes be developed in drylands?

While instances of site-specific practices for sustainable ecosystem management exist, the development of universally effective strategies for diverse drylands remains a challenge. Nature-based solutions (NBS) offer a promising avenue, encompassing actions that shield, sustainably manage, and restore natural or modified ecosystems. These approaches, adaptable to shifting external circumstances and contextual nuances, can guide ecosystem management principles in drylands. To advance this, key steps include quantifying EDVs pertinent to dryland NBS, devising novel management techniques that accommodate uncertainty and extended timeframes, evaluating the limitations of NBS in the variable dryland climate, and comprehensively factoring in trade-offs, complexities, and impending climate shifts when applying NBS in these regions.

Research priority 4.2: how can livelihood be maintained in drylands?

Livelihoods are diverse across dryland ecosystems, but their differentiation and variation are based on adaptive responses to local environmental and social conditions. Site-specific environmental knowledge and the aspirations of resident populations remain largely unconsidered within expert assessments and management strategies in dryland SES. Understanding the prime drivers of livelihood changes—determined by EDVs—is crucial. Equally important is grasping how development strategies and socio-economic changes can fortify livelihood resilience and robustness, especially in times of mounting uncertainty and risk. The research frontier includes identifying the ecological capacity for livelihoods in different drylands, quantifying the responses of livelihood-related indicators and livelihood resilience to climate change in drylands, and developing strategies to enhance livelihood capital.

Research priority 4.3: how can sustainable governance be promoted in specific dryland SES contexts?

The SDGs serve as a significant global governance tool to combat land degradation, desertification, and drought. The relations between SDGs and their interconnections with drylands governance (Stafford Smith and Metternicht 2021) should be fully explored, since measures to promote access to food (SDG 2), water (SDG 6), and energy (SDG 7), if applied under an unsustainable governance regime, could be counterproductive in enabling sustainable consumption and production (SDG 12), could aggravate climate change (SDG 13), and could undermine conservation outcomes relevant to SDG 15 (Safriel 2017). Therefore, the research frontier includes evaluating and setting priorities for achieving SDGs in specific dryland SES contexts, and construction of a cross-scale and multilevel dryland SES case study database to help explore sustainable governance pathways.

1.5 Summary and Perspectives

The development of the conceptual framework and research priorities forming the cornerstone of the Global-DEP Science Plan for dryland SESs has been a collaborative effort, marked by substantial consultations during Scientific Committee meetings and regional workshops conducted in China, Australia, and Africa. The out-comes of these endeavors have been disseminated through special issues in international journals, which has circulated the program's concepts, data, and case studies (Fu et al. 2021). A pivotal stride towards the integration of the Global-DEP into the broader landscape of land system science has been the establishment of the Global Dryland SES working group under the aegis of the Global Land Programme (https://glp.earth/). This strategic move solidifies the linkages with the broader community of land system scientists, further facilitating cross-disciplinary and international collaborations.

In light of the escalating challenges confronting rapidly transforming dryland SESs, the paramount objective of Global-DEP remains to encapsulate pivotal concepts relevant to interdisciplinary comprehension and cross-cultural insight into dryland SESs. Its overarching structure is designed to resonate with the diverse contexts of drylands, enabling it to act as a responsive tool for fostering research collaboration, policy dialogue, management practices, and sustainable livelihoods in these ecosystems.

Though the above-presented conceptual framework constitutes a simplified depiction of dryland SESs, Global-DEP diligently follows a standardized approach aimed at informing transformative policies and practices across these systems, while engaging researchers, policymakers, practitioners, and local stakeholders on a global, regional, and local scale. The programme operates with the intention of incorporating feedback and engagement from diverse locales, capitalizing on local knowledge, and considering the perspectives, opportunities, and challenges experienced by stakeholders in drylands.

The fluidity of the conceptual framework reflects its adaptability to the evolving research landscape and the dynamic demands of sustainable development in global drylands. To this end, the Global-DEP framework is set to undergo regular updates and revisions to align with research progress and evolving requirements. This iterative approach ensures that the framework remains a living synthesis of research priorities, continually guiding efforts toward enhancing the well-being of dryland ecosystems, landscapes, and livelihoods in the face of an ever-changing environment and the imperative of sustainable development.

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Part I
Thematic Issues of Dryland SESs

Chapter 2
Dryland Dynamics and Driving Forces

Bingfang Wu, William Kolby Smith, and Hongwei Zeng

Abstract Drylands are the largest biomes on Earth, yet also one of the most vulnerable to climate change and human activities. Dryland ecosystems in the world are characterized by unique and distinctive features and are known to be particularly sensitive to natural and anthropogenic disturbances. Understanding the dynamics of dryland ecosystems and their direct and indirect drivers in socio-economic and natural terms will not only provide facts and knowledge about the dynamics and drivers of future trajectories, but also provide scientific guidance for the development of appropriate measures to improve the resilience of dryland ecosystems in response to human-driven climate change. We first provide an overview of the peculiar nature of dryland land cover, which features spare and patterned vegetation, soil biocrust, and man-made solar energy surface. We specifically highlight new opportunities for remote sensing observations and discuss their potential to provide new insights into dryland ecosystem functions and services. We next discuss the importance of and trends in water availability with emphasis on the different plant water utilization strategies found across global drylands, non-rainfall water absorption, water availability estimation, and hydrological impact of land cover changes. Together these factors determine the development and degradation of drylands across global gradients of water availability. We then outline the role of climate change, population increase, and human activities in driving dryland changes. We end with a forward-looking perspective on future dryland research.

Keywords Dryland · Vegetation pattern · Biocrust · Water utilization strategy · Water availability · Driving forces · Dynamics

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2.1 Introduction

Dryland are the largest biome on Earth (Schimel 2010), yet one of the most vulnerable to climate change and human activities (Smith et al. 2019; Reynolds et al. 2007). The basic dryland landscape has long been sculpted by the interaction between low but highly variable annual rainfall, high frequency of droughts and heat waves (Huang et al. 2017), and human activities (e.g., soil cultivation, livestock grazing, and fire use). Thus, drylands are characterized by unique and distinctive features (Wu et al. 2021b), including nutrient-poor soils (Ci and Yang 2010), spare vegetation cover (Tarnita et al. 2017), biocrust (Antoninka et al. 2020), and distinct water utilization strategy (Wang et al. 2017c). These traits further influence ecosystem functions and services and reduce the resilience of ecosystem to changes in specific drivers, by providing low annual productivity (Smith et al. 2019), and regulating atmospheric carbon dioxide concentrations (Biederman et al. 2017; Ahlström et al. 2015). As a result, dryland ecosystem is regarded as a complex coupled human-environmental system (Reynolds et al. 2007). Drylands are a major component of the land surface and play an important role in global environmental change and ecological sustainability (Maestre et al. 2016; Lian et al. 2021; Li et al. 2021a), and a better understanding about drylands will help develop appropriate measures that can address anthropogenic climate changes.

Abiotic factors (e.g., precipitation and temperature), biome attributes (e.g., diversity, spatial patterns, and species invasion), and human activities (e.g., grazing, farming, and urbanization) are widely considered as the main drivers of dryland ecosystem dynamics, which represent a sophisticated social-ecological system (Maestre et al. 2012, 2016; Lian et al. 2021; Li et al. 2021a). Specifically, reduction of dryland resilience can partly be attributed to the negative impacts of extreme climate events, as it could lead to declines in vegetation diversity and cover (Delgadobaquerizo et al. 2013; Dannenberg et al. 2019), thereby disrupting species interaction networks (Burkle et al. 2013). CO_2 is an important dryland change driver as well, which produces important feedbacks to the local and regional hydrological cycles by promoting plant growth and ameliorating plant water stress (Lian et al. 2021; Gonsamo et al. 2021; Donohue et al. 2013). As the most important sources of livelihoods in drylands and the direct impact of human activities on dryland ecosystems, grazing and soil cultivation are major contributors to land degradation and desertification in drylands (Evans and Geerken 2004; Reid et al. 2005). Rapid urbanization and ecological conservation and restoration are also human activities that have impacted drylands, with the former ordinarily causing a loss of species diversity, carbon stocks, and ecosystem services (Tian and Qiao 2014; Liu et al. 2019b), but also having a positive effect on poverty alleviation, and the latter enhancing greening and ecosystem services (especially in the drylands of northern and northwestern China), but also generating considerable local water stress (Li et al. 2021a).

The intensifying variability of precipitation in drylands and the risk of global warming increase the threat to ecosystem recovery in drylands compared to other humid areas (Huang et al. 2016; Berdugo et al. 2020), as modelled by the future climate scenarios, which predicts the dryland area will increase by 11–23% by the end of this century (Huang et al. 2016; Prăvălie 2016). High variability of precipitation would reduce the soil moisture and suppress the growth of vegetation in dry season, and the increasing air temperature accompanied by abundant solar radiation result in high potential evapotranspiration (Reynolds et al. 2007) and further intensify local water stress, increasing the risk of land degradation. As the nature of low fertility of dryland soils, both tillage and grazing could cause quick and major impacts to dryland ecosystem. Thus, human resource extraction usually exacerbates land degradation in drylands as well (Li et al. 2021a; Evans and Geerken 2004). Considering the increase of global temperature and population, the risk of land degradation and desertification in dryland regions is rising, as drylands are particularly sensitive to rapid rates of physical and social change. Effective government policies are particularly important due to the growing economic and social demand for rangelands and irrigated farmlands, which will influence the attributes of drylands and the functional interactions in dryland landscapes (García-Palacios et al. 2018; McCollum et al. 2017). Regional decision-makers need to consider rapid changes in precipitation, water scarcity status, and temperature changes when proposing adaptation strategies for local ecosystems and socioeconomic development (Zhang et al. 2021). The dynamic process of dryland ecosystems and its direct and indirect driving forces in both socioeconomic and natural aspects should be studied, because these factors play a critical role in revealing the changing trends of dryland ecosystems at macro scales and thus provide the facts and knowledge about the future trajectories of dryland ecosystems dynamics.

In this chapter, peculiar dryland land cover and water availability and their changes and drivers are reviewed, synthesized, and discussed, particularly in relation to remote sensing application, for an understanding of research progress and future directions to cope with anthropogenic climate change. Diversity is absolutely the significant feature of drylands in the world, which cover about 41% of Earth's land surface and hold to more than 38% of total global population. Although large space has been devoted to discussing vegetation pattern, biocrust, photovoltaic black surfaces and plant water strategies in this chapter, we do not try to find their common features, but rather to integrate diversity into the whole description.

2.2 Peculiar Dryland Land Cover and Changes

One of the most distinctive features that distinguishes dryland ecosystems from other ecosystems is their unique and diverse land cover, which is the dynamic mixture of herbaceous, shrubs, trees, biological soil crusts (biocrusts), and bare ground. Land use/cover and its changes (LUCC) have been explored extensively (Liu et al. 2020; Li et al. 2017; Chen et al. 2015; Wu et al. 2017), but with little attention given to unique land cover types and their characteristics in dryland regions. In this section, the peculiar nature of dryland land cover is reviewed with a focus to highlight new opportunities for remote sensing observations and their potentials to provide new insights into the functions and services of dryland ecosystems.

To adapt to the harsh arid environment and water scarcity, the vegetation in dryland areas has evolved self-organizing patterns, the special spatial pattern ranging from patches to stripes to labyrinths (Mander et al. 2017; Tarnita et al. 2017), as the result of mutual compromise between dryland vegetation and the environment. Researchers have studied the self-organizing patterns for their potential value in indicating the transition of dryland ecosystems toward desertification (Konings et al. 2011; Ludwig et al. 2007, 2002), however, recent study points out that the self-organizing pattern should be regarded as a signal of resilience instead of evade tipping point (Rietkerk et al. 2021).

Biocrusts, a kind of photochemical soil commonly existing on the surface of drylands worldwide, are another uniquely prominent feature in drylands (Belnap 2003; Ferrenberg et al. 2017; Smith et al. 2019). Biocrusts are mainly formed by the interaction of bacteria, fungi, and algae with soil particles to develop a thin, dense, shell-like community of organisms on the soil surface (Ngosong et al. 2020; Ferrenberg et al. 2017). Biocrusts reflect the unique form of non-rainfall water use strategy by dryland organisms (Wu et al. 2021b), which largely changes the redistribution of surface water in deserts and sandy lands, and plays an important role in carbon and nitrogen cycling and soil organic matter formation (Reed et al. 2012; Darrouzet-Nardi et al. 2015; Rodriguez-Caballero et al. 2018).

In addition, as one of the most important man-made surfaces in dryland ecosystems, photovoltaic panels (PVs) are highly valued and rapidly expanding in drylands due to their ability to provide large amounts of green energy to humans. PVs alter the albedo in deserts and change the radiation balance at the surface, indirectly affecting local hydrological cycle and climate change (Arnds et al. 2017). However, the current feedback mechanisms of PVs on land–atmosphere interactions, and the impact of PVs on local environment and ecology are yet understood (Barron-Gafford et al. 2016). As a power means of earth observation, remote sensing has been able to accurately identify the distribution of PVs nationwide and worldwide (Yu et al. 2018; Kruitwagen et al. 2021), providing strong support for the exploration of the impact mechanism.

2.2.1 Vegetation Pattern and Changes

Vegetation in many drylands is patterned regularly. These spatial patterns are prominent in dryland ecosystems, where they are often manifested as bare soil embedded in patches of vegetation (Okin et al. 2015). These patterns include regular vegetation strips alternating with bare ground, vegetation spots and labyrinths, and regular bare ground gaps within contiguous vegetated areas (Couteron and Lejeune 2001; Klausmeier 1999), such as tiger bushes in Sahel (Rietkerk et al. 2004; Klausmeier 1999), fair circles in the Namib Desert (Juergens 2013) of Africa, cyclic vegetation patterns in southern Australian deserts (Fatchen and Barker 1979), desert shrub in

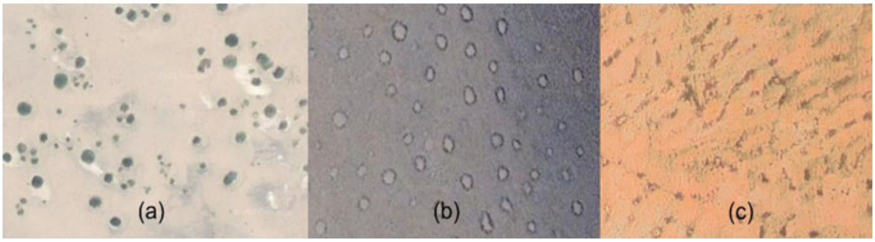

Fig. 2.1 **a** Desert shrub in Tengger Desert, China. **b** Fair circle in Namibia desert. **c** Tiger bush in northern Sahel

north-western China (Fig. 2.1), and sparse desert scrub vegetation in the tropical deserts of Mexico. Although some common understandings have been reached that large-scale regular spatial pattern may result from local biological interactions in homogeneous landscapes (Hassell et al. 1991), the specific interaction mechanism in different dryland ecosystem is still controversial. Thus, different hypotheses and models have been proposed and simulated to explain the existence of these patterns (Tarnita et al. 2017; Zhao et al. 2019; Juergens 2013; von Hardenberg et al. 2001).

The spatial pattern of vegetation self-organizes under harsh environmental conditions, related to the amount of rainfall provided to the surface according to some studies (Mander et al. 2017; Tarnita et al. 2017). If environmental conditions deteriorate, the ecosystem may tip to a barren degraded state. Therefore, the spatial self-organization of vegetation in drylands can be used as a warning signal for tipping toward an alternative stable state (Kefi et al. 2007). Early warning signals based on spatial patterns are thus highly important (Scheffer et al. 2009) as indicators for imminent tipping (Rietkerk et al. 2004) where degradation may become irreversible (or difficult to reverse). However, recent researches are prone to consider these spatial patterns as signal of ecosystem resilience instead of warning signal of critical transitions, because they are observed to stay stable for a wide range of conditions, allowing complex systems to persist beyond a tipping point (Rietkerk et al. 2021). In any case, the vegetation pattern is important for the evolution of complex ecosystems.

These patterns may positively affect essential ecosystem functions, such as ecosystem productivity in Savana (Pringle et al. 2010). The vegetation pattern is self-organized through scale dependent feedback, associated with the modification of a range of plant functional traits (Zhao et al. 2019). During the formation of spatial self-organization, vegetation can optimize nutrient utilization and enhance individual competitiveness by regulating the root-to-shoot ratio and other traits. Furthermore, vegetation could create better microhabitats for benthos through the formation of self-organization, increasing their total abundance and species richness, thus improving ecosystem productivity and stability. In addition, these patterns could enhance the landscape function in arid and semi-arid rangeland regions from the aspect of landscape as well (Bastin et al. 2002). Through the trapping and retaining of rain water, soil particles, and organic matter from vegetation patch, these patterns provide more favorable habitat for vegetation and fauna. Thus, the landscape with such vegetation patterns is considered more functional and healthier and these vegetation patches' spatial pattern can be regarded as indicators to measure the health of arid and semi-arid landscapes (Bastin et al. 2002; Ludwig et al. 2002, 2007).

The composition, structure, and function of dryland ecosystems often vary greatly over short lateral distances, reflecting the high spatial heterogeneity of moisture, which typically varies with elevation, soil type, and distance from water sources (Biederman et al. 2017). Vegetation on the soil surface both intercepts and redistributes surface water and promotes water infiltration, and it also enhances the direct and potential evaporation of stored water from soil. Thus, its spatial distribution has a strong influence on the spatial variation of moisture, which has been used to explain the formation of spatial self-organization of vegetation in drylands. On the one hand, vegetation enhances the infiltration of water into the soil and promotes the growth of vegetation to a certain spatial extent. On the other hand, competition for water among vegetation inhibits its further expansion (Rietkerk et al. 2002). In addition, the structure and composition of vegetation are influenced by seasonal to annual variations in water availability (Dakos et al. 2011; Gremer et al. 2015). Frequent or intense droughts can fundamentally alter the structure of vegetation ecosystems because long-term limited water availability inhibits further vegetation growth (van der Molen et al. 2011), while high-intensity fluctuations in water in space and time make vegetation ecosystems highly vulnerable to global environmental changes and anthropogenic disturbances (Safriel and Adeel 2008).

However, changes between the formation or disappearance of these patterns and water availability are not in real time, with hysteresis phenomenon, in fact, pervasive (van de Koppel et al. 2002). Such changes imply that both spotted vegetation patterns and bare ground at very low rainfall levels in the drylands represent the steady state of the respective ecosystems (von Hardenberg et al. 2001). The disappearance of spotted vegetation indicates a complete loss of effective root networks and enhanced water infiltration mechanisms, while bare ground may be re-covered with vegetation only when rainfall levels greatly exceed the formation level of spotted vegetation patterns (Scheffer 2020).

Therefore, the spatial patterns of the surface and patch-size distributions are interesting elements to be observed by high-resolution satellite data (Xu et al. 2015) or multi-angular data since they determine the partition of water and allow for a diagnosis of the state of ecological functioning. In addition, the spatial distribution of vegetation in the form of spaced clumps (clumped vegetation) produces anisotropic radioactive reflectance that significantly alters the surface albedo. The specific bidirectional reflectance distribution functions they present are thus beneficial to differentiate surfaces with clumped vegetation from others (He et al. 2012).

However, multi-angle satellite data such as Multi-angle Imaging Spectro Radiometer (MISR) (Chopping et al. 2008) and Compact High Resolution Imaging Spectrometer onboard the Project for On-board Autonomy (CHRIS-PROBA) (Verrelst et al. 2008) have dropped out of use, and only a few satellites such as ZY-3 are currently available to provide this type of measurement (Wang et al. 2021). ZY-3 was launched in January 2021 and carries three high-resolution panchromatic cameras and an infrared multispectral scanner (IRMSS). Positioned in the forward, longitudinal, and rearward views, respectively, these cameras allow three-dimensional mapping and can be used to map spare vegetation distributions. Recently, there has been a sharp increase in high-resolution satellite data with short payback periods taken from different viewing angles, allowing for multiple perspectives of the same target. Thus, they can be used for the observation of spare vegetation patterns. In addition, by obtaining old aerial photographs, such as declassified photographs from military satellites like the Corona series, it has become possible to track changes in vegetation patterns in drylands since the 1960s (Andersen 2006), which might provide surprising information (Li et al. 2021a).

2.2.2 *Biocrust and Changes*

Biological soil crust (BSC) is a photoautotrophic community composed of algae, bacteria, lichens, mosses, and other microorganisms that widely grow and develop in vegetation interspaces. They are the most characteristic pioneer organisms in drylands, accounting for 40–70% of the living cover (Belnap et al. 2016), and developed a thin layer of shell organisms on the surface of soil or rock by the interaction between fungi, green algae, cyanobacteria, lichens, and mosses and soil particles (Fig. 2.2) (Lan et al. 2017). BSC is often used as one of the major indicators of ecosystem stability and degraded ecosystem restoration evaluation and plays an important role in improving soil physical properties and promoting ecological restoration (Root et al. 2017; Couradeau et al. 2019; Zhou et al. 2019). Therefore, it is necessary to avoid its degradation or disappearance (Zhao et al. 2021; Giraldo-Silva et al. 2020; Antoninka et al. 2020). Some researchers found that the interaction mode between BSCs and soil will change the physical and biological environment, determine the soil texture, nutrient composition, and soil surface morphology, and affect the hydrological cycle and the capture of soil, organic matter, seeds, and nutrient-rich dust (Zhou et al. 2019; Lehnert et al. 2018; Pointing and Belnap 2012). These changes to the environment will affect the whole ecosystem.

The formation of BSCs is generally divided into five stages: sand, physical crust, algal crust, lichen, and moss. With the different stages of BSCs, their biomass and other physical properties also change. Due to their unique physiological structure and function, BSCs have a strong ability to adapt to the environment (Zhou et al. 2020). They present important ecological functions, such as enrichment of soil nutrients and nitrogen fixation through photosynthesis (Wang et al. 2017b; Ngosong et al. 2020), and affect key desert ecosystem processes in arid and semiarid areas to a great extent, and are related to Earth system functions through potential impacts on global biology and climate (Williams et al. 2016). In the 1980s and early 1990s, researchers conducted considerable research on BSCs, especially in Australia, Israel, and the western United States (Jeffries et al. 1992; Bolton et al. 1993; Tueller 1987; Bonell and Williams 1986). The study of BSCs involves many directions, such as the impact of BSCs on soils, runoff and hydrology, and the ecological role of BSCs in

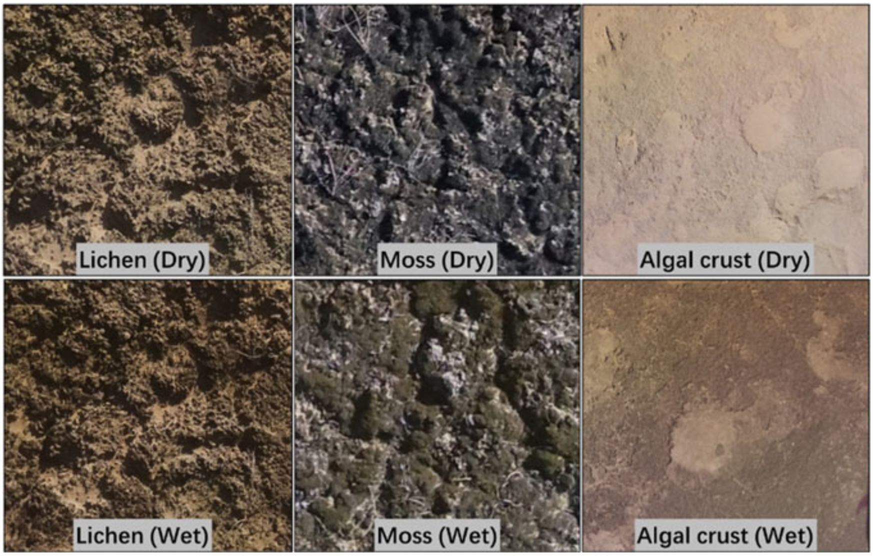

Fig. 2.2 Different kinds of biological soil crusts (BSCs)

surface restoration. BSCs in China are mainly distributed in the Taklimakan Desert, Gurbantünggüt Desert, Tengger Desert, Mu Us Sandy Land, Kubuqi Desert, Ulan Buh Desert, etc. (Fig. 2.3) and have been studied extensively (Yang et al. 2019b; Weber et al. 2008).

BSCs covering drylands worldwide are an important functional vegetation unit and play an important role in the carbon and nitrogen cycling of desert ecosystems (Chamizo et al. 2012; Weber et al. 2015; Swenson et al. 2018; Klopatek 1992; Ferrenberg et al. 2017). On a global scale, cryptogamic covers absorb approximately 3.9 Pg/yr C and 49 Tg/yr N, which account for approximately 7% of the net primary production of terrestrial vegetation and nearly half of the terrestrial biological nitrogen fixation, respectively (Elbert et al. 2012). However, BSC occurrence is mainly driven by a combination of precipitation, temperature, and land management, and land use and

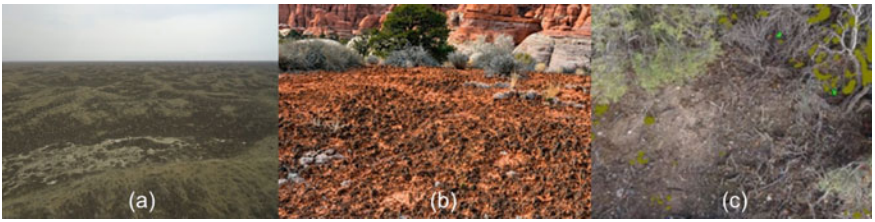

Fig. 2.3 BSCs are widely distributed in drylands, such as **a** the Gurbantünggüt Desert, China, **b** Colorado Plateau desert, United States (Smith et al. 2019), and **c** Mediterranean (Benvenutto-Vargas and Ochoa-Hueso 2020)

climate change might cause a loss of BSC coverage by approximately 25–40% within the next 65 years, which will substantially affect ecosystem functioning, including reducing the microbial contribution to nitrogen cycling and enhancing the emissions of soil dust (Rodriguez-Caballero et al. 2018). Therefore, as one of the important species in dryland ecosystems and desertification areas, BSC is of great importance for regional management. However, we lack accurate data on the spatial distribution and change in BSCs, which also leads to great limitations in our understanding of BSC coverage and functions (Smith et al. 2019). Therefore, how BSCs respond to climate change and how BSCs change the nutrient cycle (Darrouzet-Nardi et al. 2018) and spectrum of arid land (Rutherford et al. 2017) need further exploration.

BSCs can change the color and spectral characteristics of bare soil. Spectrum bands between 600 and 700 nm have absorption characteristics associated with the specific pigments in the components of BSCs, which may be used for remote sensing detector identification (Chen et al. 2005; Ustin et al. 2009; Karnieli et al. 1999). Accordingly, remote sensing offers an opportunity to detect and monitor the distribution of BSCs. However, several factors hamper the detection of BSCs, as summarized by Smith et al. (2019). The spectral reflectance of BSCs dominated by algal crust, determines the spectral difference between vascular plants and sandy soil, and proposed the crust index (CI) (Karnieli 1997). The phycocyanin in cyanobacteria has higher reflectivity in the blue spectral band than in similar substrates without BSCs. According to the normalized difference of the red and blue spectral bands, the CI image is more sensitive to ground objects than the original image. The application of the proposed CI index can be performed by any image obtained by a sensor containing a blue band. Other indices, such as Biological Soil Crust Index (BSCI), Continuum Removal Crust Identification Algorithm (CRCIA), and Crust Development Index (CDI) (Chen et al. 2005; Weber et al. 2008; Rodríguez-Caballero et al. 2017) were proposed for BSC detection with multispectral satellite images. Evaluation of the CI and BSCI with the field based on spectral and hyperspectral images shows that these two methods are not suitable for accurately distinguishing BSCs from bare soil and plants in heterogeneous areas (Weber et al. 2008). The CI is based on the normalized difference between blue and red bands, which makes it more suitable for detecting cyanobacteria dominated BSCs. The BSCI can better detect lichen dominated BSCs in the Gurbantünggüt Desert (China) based on the red, green, and near-infrared bands of the Landsat ETM + sensor. BSCs in arid areas are mainly sensitive to moisture, greatly different in dry and wet conditions (Smith et al. 2019).

Additionally, the study of BSCs coverage is very important because the difference in spectral response between BSCs, bare soil, and vegetation observed on a detailed scale leads to the difference in spectral response in heterogeneous mixing areas, which depends on the relative coverage of bare soil, vegetation, and BSCs observed in most dryland ecosystems. Random forest model was used to optimize BSCs extraction by using band combinations similar to that of the CI and BSCI, and it tested the coverage of BSCs in Mu Us sandy Land in North China using multispectral datasets (Landsat-8 and Sentinel-2 datasets). The findings showed that applying the random forest algorithm to Sentinel-2 dataset can accurately calculate the distribution of BSCs (Chen et al. 2019b). However, there are still many deficiencies in monitoring the temporal and spatial distribution of BSCs based on remote sensing, and the existing biocrust indices have limitations. When BSCs are soaked in dew and rain, their physical characteristics such as color will change rapidly, thus changing the spectral response. In addition, in the rainy season, the time span of BSCs is long when chlorophyll is quickly formed, and has an impact on the spectral response. The above problems increase the uncertainties to estimate BSCs cover and surface feature extraction, which are usually ignored in vascular plant research (Ferrenberg et al. 2017). Smith et al. (2019) suggested that a key first step to remote sensing monitoring of BSCs is their explicit incorporation into existing remote sensing algorithms. Most existing land cover algorithms do not include a BSC classification, although BSCs account for 12% of the terrestrial Earth surface (Rodriguez-Caballero et al. 2018) and 40–70% of the living cover in drylands (Belnap et al. 2016). New China cover products have taken this important step forward and added a BSC classification at a resolution of 10 m for years 2015 and 2020 in areas with less than 20% vegetation cover.

2.2.3 Photovoltaic Black Surfaces

As one of the important factors leading to global warming and climate change, the burning of fossil fuels, such as petroleum and coal, is also an important factor that many countries rely on for development (Mohsin et al. 2019). Energy demand continues to rise, which is likely to further increase carbon dioxide emissions worldwide. Studies have shown that photovoltaic electricity generation accounts for 10% of the grid, which will cause a 12.3% reduction in the global CO_2 emissions volume (Zhai et al. 2012). Therefore, governments around the world are committed to achieving the goal of energy savings and emission reduction through the use of renewable resources (Zhu et al. 2019). Renewable energy comes from abundant natural resources, such as solar energy, wind energy, and biomass. Therefore, it is considered to be an eco-friendly energy source with zero to minimum carbon dioxide emissions (Malahayati 2020).

Drylands are the main places to host green energy, such as solar and wind energy. Studies have shown that if 4% of the Earth's desert areas were fitted with PVs, the energy provided could meet the consumption needs of the entire world (Prăvălie 2016). According to the climatic definition of the sum of ultragrid and arid regions, there are currently approximately 30 million square kilometers (approximately 20% of terrestrial land) of desert area (Ezcurra 2006). Photovoltaic (PV) solar power generation has grown 41% annually since 2009, and the trend is still expanding. It is estimated that by 2040, PV solar power generation capacity will increase nearly ten times. Geospatial data describing the energy system are needed to manage the intermittent power generation, mitigate climate change risks, and determine biodiversity, conservation, and land conservation priorities due to land use and land cover changes required for trade-PVs deployment. Globally, there are approximately 68,661 facility footprints for spatial positioning (Fig. 2.4), with 423 GW (−75/+77 GW) generated

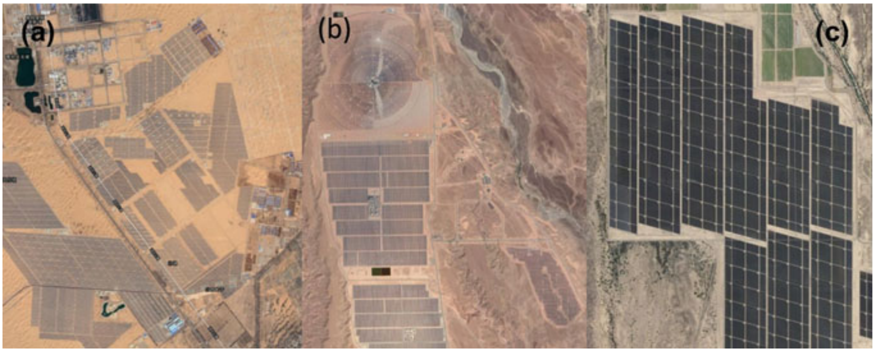

Fig. 2.4 Photovoltaic black surface in **a** Zhongwei, China, **b** Centre solar Ouarzazate, Morocco, and **c** Arizona, United States

from the end of 2017 to September 2018 (Kruitwagen et al. 2021). In the International Energy Agency's (IEA) Sustainable Development Scenario, it is estimated that 4,240 GW of PV solar power generation capacity will be deployed by 2040, which is a 10,000-fold increase from 385 MW in 2000 (Zhu et al. 2019) and a tenfold increase from 2018.

The largest proportion of PV solar panels is located in farmland, followed by dryland and grassland (Kruitwagen et al. 2021). PVs system installed in farmland can achieve significant power generation without potentially reducing the crop yield (Miskin et al. 2019). Some countries, such as China and the United States (Yu et al. 2018; Kruitwagen et al. 2021), have deployed a large amount of photovoltaic solar energy in arid areas. Although the black surface of the PV solar panel absorbs most of the sunlight, which changes the distribution of solar energy, only a small part (approximately 15%) of the input energy is converted into electrical energy. The rest is converted into thermal energy in the form of heat. At the same time, the color of solar panels is usually darker than the color of the ground they cover, resulting in considerable extra energy absorbing and releasing in the form of heat into the surrounding environment, and thus leading to climate change. Meanwhile, the deployment of large-scale PV power plants by changing the amount of albedo affects the absorption and storage of energy on the ground (Arnds et al. 2017). The surface runoff and percolation potential are significantly increased at the local scale (Pisinaras et al. 2014). If this effect only occurs in local areas, then it may not affect desert areas and less-populated areas. However, in the context of carbon neutrality and carbon peaks worldwide, countries are laying PV solar panels over a large area of thousands of square kilometers. From such a large area, the heat reradiated by PV solar facilities will pass through the atmosphere. The circulation effect redistributes energy, which will have an impact on regional and even global climates.

With the large-area use of PV solar panels, environmental problems and climate change have attracted increasing attention. In fact, whether ground observations or satellite remote sensing observations are performed, the ecosystem and microclimate of the area where the PV solar panel is installed have undergone complex changes (Barron-Gafford et al. 2016).

The change mechanism in local areas is mainly the feedback of land–atmosphere interaction, at the same time, it's also the main mechanism of drought in the Sahara due to overgrazing. Studies have found that similar land cover changes can trigger ecological and local climate responses, especially in arid/semiarid regions. Interestingly, for the first time, research has linked this land–atmosphere interaction feedback mechanism with solar PVs in the Sahara Desert. The results show that the installation of large-scale PV solar panels has improved the vegetation conditions in the area and increased precipitation (Li et al. 2018b). When a global atmospheric model with a dynamic surface is used for simulation, the PV black surface mask will cause higher land surface temperature and convergent currents compared to the desert surface, which will lead to more rainfall and promote vegetation growth. The expansion of vegetation coverage further reduces the surface albedo, and this positive feedback mechanism further expands the initial temperature and humidity conditions in the area. The construction of a PV station changes the original surface roughness, which affects the ground reception and reflection longwave radiation, wind field type, turbulence intensity, atmospheric boundary layer height, etc. which in turn change the ventilation and heat dissipation conditions of the PV station. These changes will change the local temperature and change the radiation balance (Millstein and Menon 2011). Based on the RCP2.6 scenario, with the installation of PV modules in the Northern Hemisphere, the temperature decreases by as much as 1 °C in the eastern region; however, in the Southern Hemisphere, due to the limited installation area of PV modules, the cooling effect is significantly reduced, and the change in temperature has led to changes in the global precipitation pattern (Li and Gao 2021).

However, these local-scale changes are expected to have larger-scale effects via ocean dynamics and atmospheric remote correlation. At the same time, these effects may significantly alter the assessment of the mitigation potential of solar farms, but existing models are unable to fully capture these effects due to the assumption of a constant ocean temperature and heat transport.

Large-scale PV solar farms constructed in the Sahara Desert are expected to meet the world's energy demand while also increasing the rainfall and vegetation cover in the region. However, in other locations far from the region, such impacts may offset this regional benefit. It has been shown that the redistribution of precipitation has led to drought and forest degradation in the Amazon, global surface temperature increases and sea ice disappearance, especially in the Arctic, due to increased polar heat transport and northward expansion of deciduous forests in the Northern Hemisphere (Lu et al. 2021). These remote impacts through atmospheric teleconnections and ocean dynamics by other large-scale PV farms in the rest of the world have not been addressed.

2.3 Dryland Water Availability and Changes

Another unique feature of dryland ecosystems is their low rainfall but high variability and high frequency of extreme weather events (Huang et al. 2017; Zhang et al. 2021). Water is the main influencing factor for dynamic changes in dryland ecosystems (Hoover et al. 2020), both for ecosystem sustainability and livelihood (D'Odorico and Bhattachan 2012). The greatest feature of dryland is the lack of water, which affects natural and managed ecosystems, restricts the production of livestock and crops, wood, fodder, and other plants, and affects the provision of environmental services. In arid and semiarid regions, strong coupling occurs between ecological and hydrological balance. This coupling demonstrates the central role of hydrological balance in dryland areas (Verstraete et al. 2009). In this section, we synthesize the importance of and trends in water availability with the emphasis on different plant water utilization strategies found across global drylands, non-rainfall water absorption, water availability estimation, and hydrological impact of land cover changes. Together these factors determine the development and degradation of drylands across global gradients of water availability.

Drylands are highly sensitive to strong daily, seasonal, and interdecadal perturbations of water availability (Sloat et al. 2018; D'Odorico et al. 2006). On a short time scale, such as from daily to interannual, dryland changes are dominated by climate variability. However, on a longer time scale of ten to one hundred years, the physiological effects of carbon dioxide-induced vegetation fertilization have important feedbacks on the local and regional hydrological cycles (Lian et al. 2021). Plants can evolve physiological and developmental processes to cope with unfavorable growth conditions. In the context of a rapidly growing population with a continuous increase in demand for water and food, the role of plant physiological mechanisms in coping with water stress and promoting their own growth will become even more important (Lian et al. 2021).

Renewable water from drylands is estimated at only 8% of the world's total, which is insufficient to support ecosystems at optimal functioning (Ma 2005). Moreover, water competition for humans and environmental demand has caused water scarcity and constrained the economic development in many regions around the world including the Colorado River Basin of Argentina (Wild et al. 2021). The Murray Darling River, one of the major food production basins in Australia, is in a similar state and recently a plan was established to help protect and restore regional water resources (Leblanc et al. 2012). Similarly, China has established a "Stringent Water Resources Management System", or the "Three Red Lines" as a long-term framework for addressing key water challenges (Wu et al. 2021a). Rather than an environmental challenge, achieving water security is a governance issue that requires political will, resources, and leadership (Stringer et al. 2021) to develop a synergy approach that considers the needs of humans, the environment, and ecosystems in drylands.

2.3.1 Plant Water Utilization Strategy

Natural vegetation has ecological significance in inhibiting drought and maintaining the stability of riparian ecosystems in arid and semiarid regions (Ye et al. 2010). The ability of plants to tolerate and recover from periodic water stress affects their competitiveness, survival, and distribution, thus leading to shifts in plant communities as environmental conditions change (Kilgore et al. 2021). Drought stress is a serious adverse factor that limits plant growth and productivity (Reddy et al. 2004). Drought stress induces a range of physiological and biochemical responses in plants. In arid and semiarid regions, plants have formed many adaptive mechanisms and strategies in response to water deficit through long-term natural selection and coevolution (Bacelar et al. 2006; Dichio et al. 2006). When encountering drought, the production of the phytohormone abscisic acid is triggered in plants through the accumulation of stress tolerance-related permeates and proteins, which in turn leads to the closure of stomata and induces the expression of related stress genes (Shinozaki and Yamaguchi-Shinozaki 2007). Plant root arbuscular mycorrhizal symbionts are the oldest and most common strategy to improve plant nutrient access and environmental stress response (Klironomos 2003). For example, inoculation of arbuscular mycorrhizal fungi under water deficit conditions is an effective measure to ensure or increase corn yield (Celebi et al. 2010).

Research on the relationship between climate and plants dynamics at large or regional scales has mainly focused on the response of the different vegetation types under a climate gradient to different climate types (Thuiller et al. 2004). Climate factors generally include temperature (annual average temperature, accumulated growth temperature, highest monthly average temperature, etc.), water (precipitation amount and timing, potential ET, vapor pressure deficit, air humidity), light (solar radiation), and other factors, which often determine the distribution of plants (Reich and Oleksyn 2004). Precipitation is one of the factors that determine the plant species distribution and community composition. Especially in areas with rapid declines in water availability, precipitation is the limiting factor which related to the plant diversity. Therefore, along the precipitation gradient, the plant water strategy becomes a matter of choosing whether to absorb water quickly or store water effectively. Plants can implement water-saving strategies to avoid drought through the effective use of limited water resources (Liu and Ma 2015). In dryland ecosystems, plants have developed many distinct strategies to allow the use of fog water through the canopy (Wang et al. 2017c). More than 80% of plant distribution patterns in Western Africa are significantly related to annual rainfall (Maharjan et al. 2011). Wood density and deciduousness are the determinants of plant drought tolerance. The plant species in this area are mainly tall trees that present slender and straight trunks, branching near the top platy roots, smooth and thin bark, and large dark green leaves and leathery texture. These characteristics allow plants to quickly absorb water when the rainy season comes.

In Asia, *Populus euphratica* mainly grows in extremely arid desert regions. It is mainly distributed around the Taklimakan Desert in China. *Populus euphratica* is a typical abiotic stress-resistant woody species. Biologists discovered in the *Populus euphratica* genetic spectrum that small molecules and noncoding microRNAs play an important role in growth, development, and drought stress response (Li et al. 2011). To adapt to the arid environment, the leaves on the shoots of young trees are narrow and long, while the leaves on large trees are round. After a long period of evolution, *Populus euphratica* is tolerant to light and resists heat, drought, salt, and alkali conditions. *Populus euphratica* will follow wherever the desert river flows. They can survive well by relying on the protection of well-developed root systems and a groundwater level which is not lower than 4 m (Fig. 2.5a).

In North America, *Carnegiea gigantea* (Saguaro cactus) (Fig. 2.5b) is a long-lived columnar cactus endemic to the Sonoran Desert, found in Mexico and southern Arizona in the United States. Among their many adaptations, Saguaro utilizes a distinctive photosynthetic pathway evolved to minimize water loss in hot and arid environments known as Crassulacean acid metabolism (CAM) photosynthesis

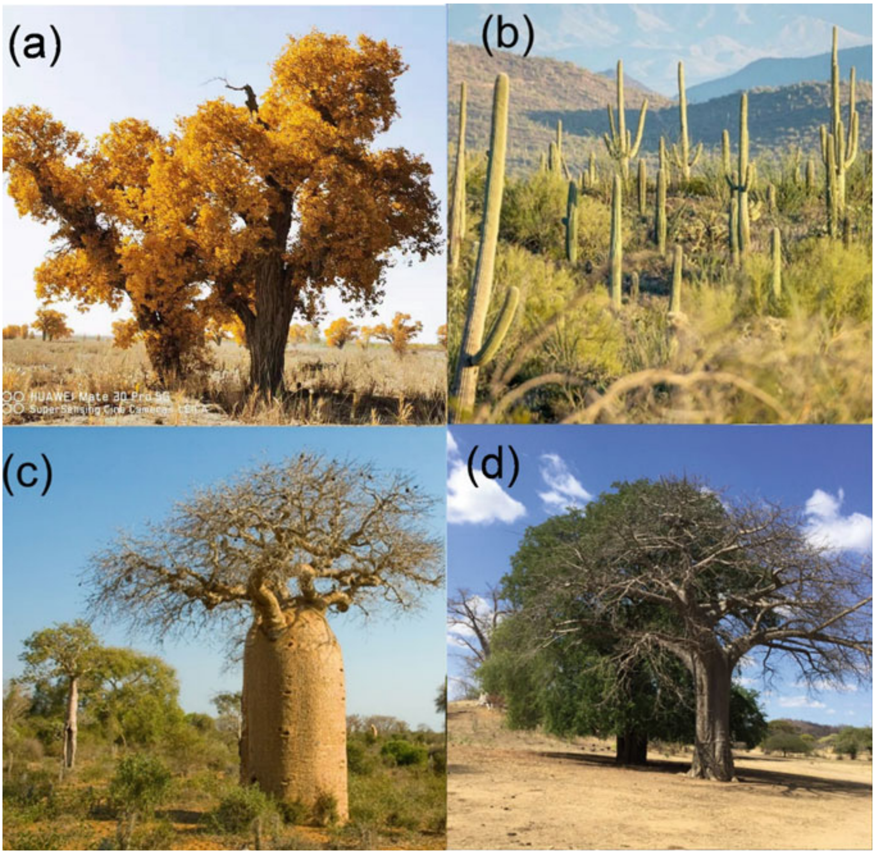

Fig. 2.5 *Populus euphratica* forest in **a** Xinjiang Uyghur, China, **b** *Saguaro cactus* in Western USA, **c** *Cavanillesia arborea* in South America, and **d** *Baobabs* in Mozambique

(Bräutigam et al. 2017). Saguaro, such as all CAM plants, absorbs CO_2 through stomatal pores at night when air temperatures are relatively low and then keeps the stomata closed throughout the day when radiation and air temperatures are relatively high, which greatly reduces water loss (Knauft and Arditti 1969). Saguaro also has shallow root systems to capture rainwater from the slightest of rainfall events, spines as modified leaves that protect the plant and prevent water loss associated with dry winds, and pleats that allow the plant to expand to hold more water during wet periods and contract during extended dry periods (Lajtha et al. 1997).

In South America, *Cavanillesia arborea* (bottle tree) (Fig. 2.5c) is a plant with distinctive water storage strategy. Bottle trees are native to the Brazilian Plateau of South America. The water is stored in the bottle tree, which has a very sturdy trunk, but few branches and leaves. This shape, different from the shape of other trees, is affected by the drought environment. In the area where bottle trees are found, the dry season and the rainy season alternate throughout the year, but the rainy season is short and the dry season is long. To survive in such an environment, they only grow sparse leaves during the rainy season, and not in the dry season. This growth form of the bottle tree reduces the transpiration and loss of water. Some bromeliads in Mexico develop specialized trichomes (Andrade 2003), and several *Crassula* species located in the Namib Desert take water up through hydathodes into their leaves (Martin and von Willert 2000).

In the African mainland, Madagascar, and Australia, *Baobabs* (Adansonia) (Fig. 2.5d) are native to the dry and hot savannah region, characterized by their massive size and multiple uses (Sanchez et al. 2010). Although all *baobab* trunks are thick, the woodwork is very sparse, a characteristic evolved to survive the dry season. During the rainy season, the thick body and loose wood are used to absorb and store large volume of water to withstand the long dry spells, which is why elephants, eland, and other animals chew the bark during the dry seasons. Whenever the dry season comes, it will quickly shed all its leaves to reduce water loss. Not only the trunk, but also the leaf characteristics of *baobab* trees reflect the species' wisdom in coping with long droughts. Studies have shown that the stomatal density of the leaves has a high correlation with climatic characteristics, positively correlated with local temperature and negatively correlated with precipitation (Sanchez et al. 2010; Abrams et al. 1990).

Organism responses in environments where rainfall is intermittent, and where the amplitude and longevity of soil moisture pulses are more important than the mean soil water levels, have long been characterized as a pulse—reserve systems (Smith and McAllister 2008). In Oceania, mainly in Australia, the stem succulent strategy can cope with very dry conditions but depends on reliable re-charge every year, a condition that is safe in the north and central American deserts but not met in central Australia (Smith and Morton 1990). Some organisms create niches for others, incidentally or mutualistically, and it seems that this occurs to protection from resource limitations more frequently in arid biomes than other ecosystems. For example, it's more common in arid environments that the presence of one plant facilitates the establishment or growth of another, of the same or, usually, different species (Flores and Jurado 2003).

2.3.2 Nonrainfall Water

Nonrainfall water includes dew, fog, and water vapor adsorption, which is an important surface water input to dryland ecosystems in arid and semi-arid zones (Gao et al. 2020). It is believed that nonrainfall water inputs (NRWIs) are extremely important for water-limited dryland ecosystems and play an important role in the dynamics of dryland ecosystems (Kaseke et al. 2017). NRWIs represent a particular water source especially for biocrusts, insects, and plants in desert regions (Kaseke et al. 2012; Zhang et al. 2015). More significant for dryland areas where precipitation is low relative to water demand (e.g., potential ET) than other non-dryland areas, NRWIs also occur in non-dryland areas, such as the Caribbean Islands and New England, USA (Wang et al. 2017c).

NRWIs are mainly provided to the soil surface through three levels: the adsorption of fog, dew, and water vapor. Each level has its own formation mechanism, and formation occurs under certain meteorological conditions and/or surface conditions (Wang et al. 2017c; Zhang et al. 2015; Meng and Wen 2016). In some inland deserts, precipitation in the form of rain or snow is the main water resource while dew and fog are considered ancillary moisture resources (Jia et al. 2014).

Dew in the land surface process model affects the microwave bright temperature and backscattering coefficient of vegetation (Kabela et al. 2009). Dew evaporation contributes 5% of the total water vapor flux measured above the canopy with a microwave radiometer (Schneebeli et al. 2011). It has been shown that the formation of dew is mainly influenced by the intensity of radiative cooling, water vapor pressure, and wind speed (Yokoyama et al. 2021).

Dew water may play an important role in keeping plants hydrated by absorbing water through the leaf surface. Dew in the semiarid desert valley area of northeastern Nevada contributes approximately 14 mm to the total annual water content which represents approximately 10% of the annual rainfall (Malek et al. 1999), and in the Taklimakan Desert of China, the average daily amount is 0.13 mm for over 77% of the growing season days, with a cumulative amount of dew for those days 16.1 mm (Zhuang and Zhao 2017). Dew even increases the CO_2 assimilation rate and leads to the complete recycling of plant water status and leaf pigment content, which is significant in the hydration and activation metabolism of water stressed in summer (Munné-Bosch and Alegre 1999).

Fog can be detected and mapped with both geostationary and polar satellite data (Amani et al. 2020; Wu and Li 2014). In some coastal locations, vegetation appears to use fog-drop water year-round. Seasonal advective and orographic fog supply the only significant annual moisture along 3000 km long in the hyperarid coastal belt of Peru and Chile (Moat et al. 2021). In addition to providing water sources, fog water also helps to change the energy balance of vegetation, reduce transpiration, increase stomatal conductance, and increase the CO_2 absorption rate (Martin and von Willert 2000).

These NRWI components, typically very small but important for determining the magnitude of water and energy flux (e.g., latent heat) during the dry season (Gao et al.

2020; Wang et al. 2017c; Uclés et al. 2013), may become a major water source that helps reduce various water pressures for living things in a particular environment. Studies have shown that canopy dew at the Mizhi Experimental Station in Shaanxi Province significantly reduces night sap flow, and the contribution of NRWIs to ET can reach 18.4% (Gao et al. 2020). In some coastal desert areas, NRWIs may exceed precipitation (Henschel and Seely 2008).

In recent years, NRWI research has again become a research focus, arising largely from the state of human-water tensions in the world. There is growing evidence that these small but important NRWIs have a significant impact not only on vegetation growth but also on the survival of other microorganisms and the maintenance of the dynamic balance of geo-biochemistry under arid and low rainfall conditions (McHugh et al. 2015). However, NRWI research currently mostly uses traditional observations to focus on local areas and small-scale units to explore its magnitude and ecological effects on local areas. The method to use remote sensing for performing large-scale quantitative estimations of NRWIs and their ecological effects on drylands is still lacking.

2.3.3 Water Availability

The major challenge for managing water resources in modern, developed river basins is to determine the safe, sustainable limit for water utilization. In many river basins around the world, water is overallocated and over extracted for use in agriculture, cities, or industry. In fact, irrigated agriculture accounts for roughly 80% of global freshwater consumption and 40% of global crop production, used to stabilize food and feed production across dryland regions (Döll and Siebert 2002; Siebert and Döll 2010; McCabe and Wolock 2007). As a result, insufficient water is available for environmental and ecosystem flows, and in many places, rivers no longer reach the sea, groundwater tables have dropped, and lakes and natural habitats have become dry and degraded (Grafton et al. 2013).

Global, regional, and basin-wide estimates of water availability rely on models (Masood et al. 2015; Trenberth et al. 2007; Hanasaki et al. 2008), subject to large uncertainties due to soil hydraulic parameters, weather conditions, and land cover changes, and how these factors correlate with water availability (Hanasaki et al. 2008). Human activities and climate change have greatly influenced the natural hydrological cycle and changed the availability of water resources (Grafton et al. 2013). In recent decades, the natural landscape and associated hydrological characteristics have changed considerably worldwide as well as in China. In China's Loess Plateau, evidence has shown that revegetation intensity has been approaching regional sustainable water resource limits (Zastrow 2019; Feng et al. 2016). There is a huge decrease in the ratio of annual runoff to precipitation in many catchments because of the expansion of forests which consume more precipitation (Zastrow 2019). The overexpansion of cropland and ecological shelterbelt is reported as the major causes for the shrinkage of the Ebinur Lake basin (Zeng et al. 2019). Likewise, the Colorado River Basin (CRB) in the United States has experienced an increasing demand for water due largely to agricultural intensification, driving more frequent periods of water shortages as precipitation didn't increase while temperature increase, and leading to failures in meeting water allocation demand (Woodhouse and Pederson 2018).

All of these interventions change the spatiotemporal component development of the hydrological cycle; however, they are not adequately reflected in the hydrologic models on which estimates of sustainable water use are based. These conditions continue to simulate water availability under natural conditions (Hanasaki et al. 2013). It is complex to precisely assess the available water amount for human use in a highly developed basin using available hydrological methods. It is also difficult to set up full coverage monitoring system through metering facilities due to the high cost of both building and maintenance (Berbel and Esteban 2019). Thus, objective estimates of the available consumable water (ACW) for human use, i.e., the consumption cap at the basin or subbasin scale require a water consumption balance approach (Wu et al. 2018). Such an approach would provide a solid basis for analyzing the influences of climate change, cropland expansion, and large-scale revegetation programs.

ACW is the total amount of water in the basin sustainably available for human activities after accounting for natural inflows and outflows and the requirements of environments and ecosystems. To guarantee sustainable water resource management, ACW is calculated with the following principles: (1) groundwater overexploitation is forbidden; (2) enough water is saved for sustainable natural ecological systems; (3) basic environmental flow is considered in river systems; and (4) the water cycle between ground and surface water systems is conserved (Wu et al. 2014).

Essentially, the ACW provides water managers with a maximum allowable human activities cap or the upper limit on water consumption in basins or watersheds, which can also be called the water boundary at basins (Zipper et al. 2020). The ACW water balance equation can be rewritten as follows:

$$ACW = (SystemInflows - SystemOutflows) + Precipitation - ET_{natural}$$

In this equation, *System Inflows* include upstream river inflows and inter-basin transfers, both of which are monitored and measured with stream gauges. *System Outflows* include the outflow of water dedicated for environmental needs (for habitat and aquatic ecosystem services, etc.), as well as any outflows unusable by humans, such as flood runoff exceeding designed dam or water bank holding capacity, groundwater recharge to saline aquifers, and rivers and lakes sewage discharge. *Precipitation* is the average annual precipitation in the basin, a key parameter for the ACW. *Precipitation* can be accurately measured at individual rainfall station sites, but at the basin scale, station-based monitoring may produce large errors due to the insufficient number of stations, especially in upstream mountainous areas where rainfall can be relatively high. New remote sensing technology can now be combined with rainfall station monitoring to produce high-precision precipitation data sets across all land-use types in the basin. *Natural ET* ($ET_{natural}$) is the uncontrollable ET from natural forests, grasslands, wetlands, etc. Traditional water resource accounting methods cannot accurately measure $ET_{natural}$ which is critical to calculate the realistic, sustainable cap on human water consumption in the basin—i.e., the ACW. In the water balance equation, the basin ACW can be calculated by subtracting $ET_{natural}$ from the other measurable inflows and outflows. Remote sensing can be used to determine $ET_{natural}$ (see below). Satellite remote sensing is a revolutionary technology that allows, for the first time, comprehensive and accurate measurement of the three key dimensions in the ACW sustainable water balance equation: *land-use*, *precipitation*, and $ET_{natural}$.

Remote sensing to classify land-use—Satellite imagery is used to optically classify land-use into categories relevant to consumptive water use amounts and patterns. Land cover throughout the basin is divided into two categories: (1) *natural land cover* is the land without human intervention, including natural forests, grasslands, shrubs, bare lands, etc.; and (2) *artificial land cover* is the land with human development, such as agricultural fields (irrigated or rainfed) and urban settlements (including irrigated and impervious surfaces). For instance, across agricultural regions of the western Unites States, recent land-use classification algorithms capable of mapping irrigated croplands have revealed widespread land fallow during drought events, especially in regions of secondary water rights (Deines et al. 2017; Norton et al. 2021). There is an intrinsic link between each different land-use type and its associated ET rate, which can be determined by statistical processing of the RS data, as discussed below. Land-use changes over time in any basin, and remote sensing can regularly update land-use images to re-calculate ET estimates.

Remote sensing to measure precipitation—Coupling remote sensing derived precipitation (Pradhan et al. 2022) with ground rainfall observations can generate accurate precipitation estimates with fine spatial resolution throughout an entire river basin or watershed. The approach adopts machine learning based statistical technology to quantify the relationship between precipitation and such influential factors as vegetation, topography, cloud, and other physical variables, and build the regression formulation: P = F(Veg., Terrain., Cloud, Others), and then use the high resolution variables as input to generate high resolution precipitation (Elnashar et al. 2020). This downscaling approach works well for annual precipitation estimates. For monthly or daily precipitation estimates, more sensitive variables with precipitation than vegetation should be integrated into downscaling models, such as the physical suitable variables of cloud that can overcome inaccuracies due to the possible lag in response of vegetation to precipitation (Duan and Bastiaanssen 2013).

Remote sensing to predict $\mathbf{ET_{natural}}$—Natural ET can be calculated from specialized remote sensing-based algorithms such as ET measurements from ETWatch (Wu et al. 2020), land cover and land use data, and known environmental factors. ET algorithms can be mechanistic or based on machine learning statistical models (Javadian et al. 2020). Total ET for any plot of land (or pixel in the satellite image) is a function of environmental factors such as precipitation, vapor pressure deficiency, wind speed, radiation status, surface temperature, soil properties, terrain, etc. Remote sensing, environmental and meteorological data can be combined in advanced, cognitive computational models using data mining in a machine learning to generate the statistical link between the natural land cover types, environmental factors, and the estimated ET. Statistical models can be used to predict ET rates for any land-use type throughout a given basin. Figure 2.6 shows a schematic of the pixel-by-pixel land-use correlations with ET rates used in machine learning analytical models for natural ET. First, the land cover is reclassified into natural and artificial land covers, and then the environmental factors and ET are divided into natural and artificial components; Second, the machine learning method is used to explore the link between ET and environmental factors of natural land cover to build the natural ET prediction model; Third, the environmental factors of the artificial land cover are used as input to predict the natural ET of artificial land cover types; Finally, the natural ET of the artificial land cover are subtracted from the total ET to obtain the human management ET.

The RS-determined values for land-use, precipitation, and ETnatural can then are used in the water balance equation, along with traditional measurements for external-basin inflows and outflows. The outcome determines the ACW—i.e., the sustainable limit on total human water consumption in the basin.

This new RS-ACW water accounting methodology provides significant benefits over the traditional water resources assessment (WRA) method. Satellite RS data allow regular, inexpensive re-estimation of water balances from contemporary land-uses in the basin on an annual basis (or monthly, with some modifications). The RS-ACW approach is also more accurate, better accounting for actual precipitation

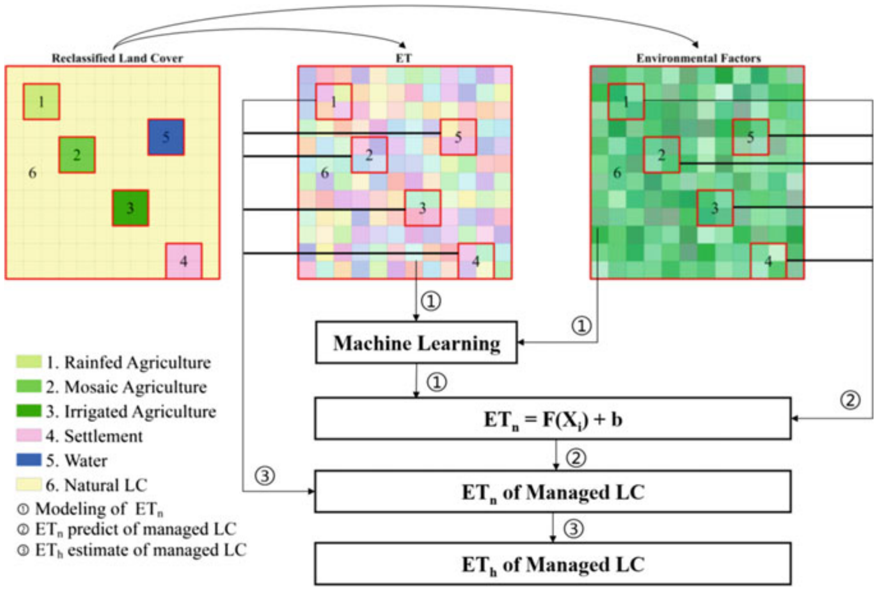

Fig. 2.6 Analytical model for predicting natural ET. The land cover was divided into natural land cover, rainfed, mosaic, irrigated agriculture, and settlement. The environmental factors and ET were also separated into natural and artificial parts. The natural ET prediction model will be built for natural land cover using the machine learning, and then will be used to predict the natural ET for each artificial land cover types

and ET levels for various land-use types in the basin. Also, the RS-ACW approach does not over-estimate available water—unlike the WRA method (Wu et al. 2018). Finally, the water required for sustainable natural environments and ecosystems is 'set aside' before calculating the water available for human consumption.

This approach has been successfully applied in three recent World Bank projects in China. Turpan is among the poorest, extreme drylands in western China. There exists an observed paradox in Turpan whereby modern, large-scale efforts to save water eventually led to total water consumption increase as in other arid regions around the world (Tan et al. 2018; The World Bank 2012; Grafton et al. 2018). In 2008, the World Bank launched the Xinjiang Turpan Water Conservation Project to investigate the issues and recommended innovative and workable solutions (The World Bank 2012). The project's solution to this paradox is to define a sustainable limit on total water consumption at the basin scale and watersheds, set targets for reduction, and then apply irrigation and agronomic interventions carefully to reduce the actual water consumption (ET), monitor and supervise the actual water consumption of farmers, WUAs, townships, and counties using advanced remote-sensing, and reform water rights and water pricing incentives based on ET (Wu et al. 2021a).

The project set a sustainable cap on water use and enforced a strict cap on overall water consumption for Turpan Basin and 11 watersheds, which allocated to 3 counties and down to townships and villages by conducting water balance/budget analysis with multi-stakeholder decision-making to define a prioritized budget for water consumption in the basin. This basin-level balance accounts for all water supplies and all competing water demands (i.e., ecosystem, agricultural, industry, municipal demands, etc.).

The project recognized that only "the reduction of water consumption" can fundamentally solve the water resources problem in the basin. When each watershed within the basin has arrived close to the maximum water consumption limitation then more water consumption is limited, thus it can fundamentally avoid groundwater over-extraction. The project demonstrates that remote sensing provides tools to monitor ET and cropland as independent means of monitoring actual ET vs target of farmers, WUAs, townships, and counties at relatively low cost as compared to expensive water monitoring systems used in countries such as Australia (Grafton 2017).

2.3.4 Hydrological Impacts of Land-Use Change

Land use changes can dramatically modify water dynamics globally (Sterling et al. 2013; Bosmans et al. 2017) and in dryland ecosystems (Lv et al. 2018; Feng et al. 2016; Yin et al. 2017). Changing current land cover types and land management activities in river basin areas will change hydrological processes such as surface runoff, base flow, ET, soil water holding capacity, interception and groundwater recharge, leading the changes of the path from rainfall to runoff, thereby reflecting changes in water demand (Naha et al. 2021; Rogger et al. 2017; Lv et al. 2019; Chen et al. 2019c; Yang et al. 2019a). Therefore, a comprehensive understanding and assessment of the impact of land cover changes on hydrological processes is necessary for watershed management, environmental policy, ecological governance, and the choice of restoration measures.

Distributed hydrological models (e.g., SWAT, VIC, MIKESHE) are used in assessing coupled hydrological processes with analyzing effects of LULC changes on these processes, where the hydrological models are calibrated and validated using ground observational data based on baseline land use scenarios. The calibrated model is then run for various land cover scenarios, and then the changes in the simulation are compared (Naha et al. 2021; Li et al. 2018a). However, there are some uncertainties concerning the input, parameters, and structure of the chosen model (Chen et al. 2019c; Her et al. 2019). Given these uncertainties, it seems reasonable to doubt the reliability of the estimated hydrological response to land cover changes, particularly when the responses are mild or moderate (Yin et al. 2017). The uncertainty induced by the model parameters or structure has unpredictable bias on the impact on the assessment of land use changes (Chen et al. 2019c). Without parameters to be calibrated, remote sensing based approach might be more promising (Wu et al. 2018; Zeng et al. 2019).

Changes from natural land cover types to artificial land cover types will significantly alter the regional hydrological characteristics. Since the development of civilization, agriculture has taken land (and water) from natural ecosystems such as forests, savannas, and grasslands. In the process of agricultural land increase, water and biogeochemical cycles have been significantly changed (Bonan 2008; Davidson et al. 2012; Runyan and D'Odorico 2016). For instance, recent findings have indicated significantly increased ET across global croplands with exceeded water inputs, suggesting that recent increase in food production may be dependent on unsustainable water inputs (Javadian et al. 2020; Pascolini-Campbell et al. 2021). Rainfed farmland keeps lower ET rates with smaller leaf area, smoother surface roughness, shallower root depth, and higher albedo to reflect solar radiation reaching ground surface (Bonan 2008; Perugini et al. 2017). Influenced by more compacted cropland soils, the infiltration rates in cropland are also smaller than natural land cover types from intensive machinery operations and fallow activities. In those regions, higher surface runoff is expected after decreasing ET and surface infiltration (Runyan and D'Odorico 2016). On the contrary, in irrigated cropland, water usage for crop growing can easily dry surface water bodies (Jägermeyr et al. 2017).

Afforestation/revegetation is encouraged worldwide for ecological purposes, particularly in Loess plateau and Sahel. However, the influence of re-vegetation on water resources remains controversial in humid tropics (Lacombe et al. 2016), but decreases water availability in drylands (Wang et al. 2017a; Lian et al. 2021; Feng et al. 2016), which may lead to irreversible catastrophic consequences for dryland ecosystems (Li et al. 2021a). In drylands, the cascade reaction of land cover changes will cause important changes in water resources, including the spatiotemporal pattern of precipitation and ET deduced from regional microclimate conditions, leading to significant changes of human available water resources (Perugini et al. 2017; Wu et al. 2018). The loss of terrestrial water storage in Yellow River was mainly caused by ET increase, which was resulted from higher vegetation cover and more irrigation water use (Lv et al. 2019). The streamflow is temporally companied with agricultural land returning forest activities in Wei River of China (Wang et al. 2017a).

Researches warn that the revegetation on the semi-arid Loess Plateau has already reached the limit of soil water-carrying capacity for vegetation (Feng et al. 2016). The resulting widespread dried soil layer potentially threatened tree mortality (Huang 2019; Wang et al. 2018). Although at present efforts are taken to better understand vegetation productivity thresholds (Feng et al. 2016), equilibrium vegetation cover (Zhang et al. 2018), regional water resource development boundaries (Wang et al. 2018), and soil water-carrying capacity for vegetation (Huang 2019; Jia et al. 2019), it remains a challenge to keep in balance between vegetation productivity and water use in order to sustain a healthy ecosystem.

Another significant land cover change is ecological protection prospective, such as farmland returning to forest and afforestation activities (Li et al. 2021a). In dryland system, ecological restoration significantly influences local hydrological cycling patterns with different water consumption mechanisms. With growing fraction of vegetation cover, plants in revegetated areas are more active in photosynthesis and transpiration processes, thus raising the demand of water consumption (Ma et al. 2019), as the revegetation in China's Loess Plateau demonstrated that the limit of regional water resources requires sustainable ecological restoration planning (Feng et al. 2016). Large land cover shifts in Europe (e.g., deforestation or afforestation, urbanization) from the 1950s are evaluated as the same degree of net impact on the amount and distribution of water resource availability (both ET and streamflow) as precipitation changes from climate changes (Teuling et al. 2019).

The Great Green Wall (GGW) project in Sahara is conducted as a pan-African program with a strong reforestation focus (Goffner et al. 2019). However, farming and afforestation in the African Sahel and Great Green Wall (GGW) regions are constrained by climate variability, water scarcity, and degradation of lands (Mirzabaev et al. 2021). Accordingly, 1,337,535 km^2 (43.5%) and 729,576 km^2 (25.6%) of the Sahel and the proposed GGW region, respectively, not feasible for sustainable planting for rainfed and natural vegetation growth conditions, require supplementary irrigation (Elagib et al. 2021). Certain land use purposes as afforestation and cropland cannot be achieved without sufficient water resources, which is more water consumptive with higher ET.

Conversion of forest or woodland to cropland over large regions (e.g., >100 km) is highly correlated with precipitation reduction (especially the rainfall frequency) and diurnal air temperature increase (Bonan 2008) The regional climate can be changed by LUCC with variant component amounts of the land surface energy balance and ground-atmosphere interaction, influencing near-surface temperature, boundary layer stability, and convective precipitation (Bonan 2008; Perugini et al. 2017). The rainfall regime can also be changed within the same region in which land cover change occurs to even influence adjacent ecosystems (Ray et al. 2006). Another impact of land cover change is the modification of cloud microphysics and cloud processes by altering the biological aerosols emission rate (Pöschl et al. 2010). Regional precipitation highly relying on the regional ET will largely decrease with lower air moisture and decreased ET (Yosef et al. 2018). Although land cover changes have potential impact on precipitation, rainfall is mostly influenced by climate changes and land cover is straightly connected with water consumption processes, i.e., ET (Lian et al. 2021). China and India are leading the greening phenomenon (Chen et al. 2019a). In India, cropland expansion from bare soil or desert grass is in charge of bulk of the ET increase and water consumed in this process can is equivalent to almost ten times the area of the degraded desert grass ecosystem recovery water amounts (Das et al. 2018). Stricter policy to limit cropland expansion is expected in sustainable ecosystem development planning (Das et al. 2018; Zeng et al. 2019). Considering the consequences of India polices, countries shouldn't neglect the significant ET changes accompanying land cover transitions in implementing different future dryland development plans, such as China and Europe for ecological protection with restoration plans, India for food security with cropland expansion plans as well as social and economic development plans with urbanization (Teuling et al. 2019; Wu et al. 2014). ET is the priority element in scheduling the allocations of dryland water resources, which is recommended in China's future water resources policies (Wu et al. 2021a).

2.4 Driving Forces of Dryland Changes

Climate change and human activities are widely considered as major driving factors of dryland dynamics and have intensified the risk of desertification and land degradation in drylands (Li et al. 2021a; Ruppert et al. 2015; Stringer et al. 2021). Climate change mainly refers to the intensifying interannual and interdecadal variability of precipitation and temperature in drylands under the impact of global warming, with wetter weather in the American drylands and drier weather in Eastern Hemisphere drylands as the modelling result of Huang et al. (2017). High variability of precipitation and temperature means more extreme droughts and more extreme rainfall in dryland regions, which further exacerbates water stress and water competition among dryland ecosystem components, reduces the stability and resilience of dryland ecosystem, thus elevates the risk of desertification and land degradation in dryland regions. Human activities mainly include agricultural development, overgrazing, urbanization, and ecological restoration. In the drylands of northern China, large-scale ecological conservation and restoration projects are the main drivers of local dryland dynamics. Such measures reflect the influence of government policies on the dynamic evolution of drylands, enhancing local greening and ecosystem services, but also imposing significant water stress in China's drylands (Li et al. 2021a). Agricultural development and grazing are the most widely distributed human activities, which are more prone to lead land degradation in drylands owing to the low productivity nature of dryland ecosystem (Reynolds et al. 2007). Further, the extent of farmland and pastures are mainly driven by population and agricultural efficiency. Changed land tenure systems and consumption preferences play a key role in reducing the risk of land degradation and food insecurity as well (Stehfest et al. 2019). Understanding the driving mechanisms and extent of natural and human forces to dryland change at multiple spatial scales, and the dynamics of dryland ecosystems under their influence is critical to the sustainable management of dryland ecosystems (Liu et al. 2015a).

2.4.1 Climate Change

As the monitoring results show, global drylands have expanded in the last 60 years and would continue to expand in the twenty-first century (Huang et al. 2017, 2016; Koutroulis 2019; Spinoni et al. 2021). Long-term monitoring also indicates that global drylands are in a state of accelerated expansion (Huang et al. 2016). Evidences have shown that global dryland extent had increased by 4% between 1991 and 2005 (Feng and Fu 2013), and the modeling for future climate scenarios has shown that drylands will experience increase in aridity (Yuan et al. 2019), and more frequent and more severe extreme events, such as extreme droughts and extreme precipitation (Sloat et al. 2018; Gampe et al. 2021; Zhang et al. 2021; Berdugo et al. 2020). As a result, global dryland areas are expected to increase by 11–23% by the end of this century (Huang et al. 2016; Prăvălie 2016). Global drylands are changing in extent, structure, and function, and could expand by 23% by the end of the century under a pessimistic future climate scenario (RCP 8.5), coupling with the impact of aridity (Berdugo et al. 2020), implying that drylands will account for 56% of global area (Huang et al. 2016). However, it is also argued that current projection of dryland expansion is overestimated, which means global drylands will not expand as significantly under a warming climate (Berg and McColl 2021).

Dryland expansion will reduce soil water content, soil organic carbon and gross primary production, further intensifying regional warming, resulting in a warming trend in drylands that is twice that of wet areas (Huang et al. 2016). The warming could lead higher evaporative demand and less soil moisture, which further cause an increased sensible heat flux and declined latent heat flux, meaning a strong impact on temperature extremes (Seneviratne et al. 2014). Thus, the simulated warming trends in drylands is more significant than that in wet areas. Moreover, expanded dryland may result in a declined soil organic carbon storage, as studies show that soil organic carbon storage would decrease with increase temperature and decrease with decline soil moisture (Huang et al. 2016). Furthermore, the lost soil organic carbon would increase emit CO_2 into atmosphere and the declined soil moisture would suppress the photosynthesis activities of plants and gross primary production accumulation, both of which may further reinforce the warming and form positive feedback (Huang et al. 2016).

Extreme climate events such as droughts and heat waves significantly increase the risk of negative changes in dryland ecosystem dynamics and reduce the resilience of dryland ecosystem (Zhang et al. 2021; Gampe et al. 2021; Yuan et al. 2019; Sloat et al. 2018). Aridification leads to systematic and abrupt changes in the ecosystem structure and function, including plant productivity, soil fertility, plant cover, and richness (Berdugo et al. 2020). Warming may reduce soil water availability (Schlaepfer et al. 2017), soil fertility, plant productivity (Berdugo et al. 2020), leaf abundance, and species diversity (Maestre et al. 2016) in dryland ecosystems. The most obvious soil drying occurred over transitional areas between dry and wet regions (Cheng and Huang 2016). Furthermore, the decrease in soil moisture and intense droughts would expand the major deserts in the world, including the Sahara, Arabian, Kalahari, Gobi, and Great Sandy Desert (Zeng and Yoon 2009).

However, desertification is not a global trend as vegetation cover and rainfall are increasing in many drylands. Evidence from time series analysis of satellite images shows some arid lands, such as the Sahel and the Mediterranean basin, as well as China Loess plateau (Li et al. 2021a; Chen et al. 2019a; Gonsamo et al. 2021), are currently greening up (Hellden and Tottrup 2008). This obvious inconsistency is due to different interpretations of aridity, whether in an atmospheric, agricultural, hydrologic, or ecological context. In the semi-arid regions of China, vegetation greening leads to an increase in net primary productivity (NPP), attributing to increased atmospheric CO_2 concentrations (i.e., CO_2 fertilization) (Gonsamo et al. 2021). However, CO_2 fertilization greening increases ET and thus decreases soil moisture in this region (Feng et al. 2016), making existing ecosystem hardly sustainable because the declining soil moisture will limit primary productivity (Peng et al. 2013) and affect photosynthesis in plants that can absorb CO_2 and store carbon, especially in C_4 plants with high photosynthesis levels (Li et al. 2021a), as well as shape the patterns of plant species richness (Sun et al. 2021).

2.4.2 Agricultural Development

Agricultural development is another important factor contributing to the degradation of dryland ecosystems, which consists mainly of overgrazing or unsustainable agricultural practices that exceed the limits allowed by the fragile environment. Cropland expansion has serious adverse effects on biodiversity and carbon stocks through habitat loss and fragmentation (Molotoks et al. 2020, 2018), whereas it provides necessary food for the growing population. However, although unreasonable agricultural measures may temporarily obtain higher input returns, they are at the expanse of ecosystem stability and are not sustainable. The increased frequency of droughts poses a significant risk to agriculture as well, with different impacts on the water balance and crop cropping systems (Sivakumar et al. 2005). Further, overgrazing has been considered to be related to grassland degradation and shrub encroachment in many dryland regions (Gaitan et al. 2018; Fredrickson et al. 2006).

In dryland regions, one of the major unsustainable agricultural practices is the overdevelopment of irrigation infrastructures with the aims of increasing crop or livestock production (Geist 2017). These practices are often based on the misconception that livestock production or crop yield in dryland regions is limited by a lack of water for drinking or irrigation. However, the development of these irrigation systems without considering the vulnerability of dryland soil and water availability intensifies the use of land resource, which may further lead to desertification (Niamir-Fuller 1999; Zeng et al. 2019). Moreover, the construction of these infrastructures limits the mobility of herbs and crop cultivations, thereby reducing the restoration ability of land rotation schemes (Wang and D'Odorico 2008). Furthermore, the initial investments in these infrastructures reduce the short-term gains, which enhances the intensive use of the land. As a result, these investments may lead the entire ecosystem toward an unstable state.

In addition, other unsustainable agricultural practices include residue removal, continuous cropping with limited inputs, and cultivating soils that are marginal for crop production (Blanco-Canqui and Lal 2009), which further degrade the land quality in drylands. Crop residues are used as fuel or fodder, which would accelerate the loss of soil carbon and increase soil erosion. Erosion reduces soil fertility by removing nutrients and organic matter from the land, and degraded soils are not conducive to water infiltration, thus leading to a continued decline in crop productivity and soil quality.

Overgrazing and grazing abandonment also play an important role in the degradation of dryland ecosystems. In the Mediterranean region, overgrazing has led to land degradation in Spain, Greece, and Cyprus (Gaitan et al. 2018; Peco et al. 2017; Concostrina-Zubiri et al. 2017; Riva et al. 2017). In China, both drought and overgrazing are regarded as the major drivers of the decline in grassland biodiversity, ecosystem functions, and services in the northern arid and semi-arid zones (Li et al. 2021a), with overgrazing having a greater negative impact on plant species richness, plant productivity, and soil carbon content than drought (Peco et al. 2017). Some researches indicate that overgrazing is one of the main factors to the shrub encroachment in grassland in arid and semi-arid regions as well (van de Koppel et al. 2002; Fredrickson et al. 2006). Proper grazing speeds up the reproduction and regeneration of vegetation, thus has a positive impact to grassland ecosystems. However, on the one hand, overgrazing will increase the livestock load of grassland and reduce the structure and stability of grassland ecosystem. On the other hand, it tends to alter the community structure of the grassland ecosystem and increase the proportion of shrub by prefer to eat herbaceous and spread shrub seeds (Fredrickson et al. 2006). In contrast, some studies also point out overgrazing may not be the direct cause of shrub encroachment. It is the grazing abandonment after long term overgrazing who triggers shrub encroachment in grassland rather than overgrazing (Sanjuán et al. 2018), yet the mechanism behind it is still unclear. Moreover, the impact of grazing abandonment on dryland ecosystems might not be the expected positive. Studies have shown that although grazing abandonment may have a positive impact on the increased cover of soil crusts (Concostrina-Zubiri et al. 2017), it may also have a negative impact on soil fertility, carbon storage, soil multifunctionality, and soil microbial activity (Peco et al. 2017). In addition, studies have shown that wildfires in abandoned terraces could be the primary reason for severe soil degradation in the Mediterranean (Lucas-Borja et al. 2018) and that fires may be the main driver of the transition from Mediterranean oak woodlands to shrublands (Guiomar et al. 2015).

2.4.3 Urbanization

Monitoring of global drylands shows that the region is maintaining rapid urbanization despite the lack of water and other essential natural resources. The trend of rapid urbanization would be maintained until at least year 2040 as the modelling result shows, which may have a significant negative impact on food production (Chen et al. 2020a). Rapid urban expansion had significantly negative impacts on environmental sustainability, leading to a decline in the comprehensive index of environmental sustainability in northern China (Liu et al. 2019b). Moreover, large scale urban development tends to occupy the original farmland, forestland, and shrubland, which leads to the fragmentation of ecosystem landscapes, resulting in the loss of ecosystem service value (Chen et al. 2020b) and ecosystem carbon stock (Yan et al. 2016). As a result, assessing the environmental impacts of urban expansion is highly important for the sustainable development of drylands.

In the drylands of northern China, rapid urbanization has led to serious challenges to sustainable development due to its influence to ecology and the environment. The water demand in this region accelerated by 50.75% from the 1980s to 2010 (Liu et al. 2015b), while the per capita water resources in the region are very limited, at 68% of the national average. Moreover, the land area undergoing desertification increased to 2.636 million km^2 in 2010 (Xu et al. 2014), while the NPP of grasslands is experiencing a steady decline (Tian and Qiao 2014). Although urbanization poses a threat to the ecosystem stability and water resources tension in some drylands, it also brings more employment opportunities for people living there. Land is no longer the only way for local people to invest in labor. For example, the poverty alleviation projection has greatly reduced the incidence of poverty in the arid areas of Northwest China. People have more opportunities for employment, study and medical treatment, and the quality of life has been improved. In addition, studies indicate that climate change will be the key factor affecting urban expansion in this region, and the area of urban land affected by climate change is expanding (Liu et al. 2019a). To improve the sustainability of cities, effective measures to mitigate and adapt to the impacts of climate change on urban expansion are required.

2.4.4 Population Increase and Poverty Alleviation

Drylands hold to more than 38% of the global population (Reynolds et al. 2007), and increasing population pressure means that the impact of human activities on dryland ecosystems will continue unabated. Population growth has a number of consequences, such as increased demand for food and the expansion and intensification of crop/livestock production systems (Liao et al. 2020). Crop and livestock production is shifting from a relatively extensive, low-input/output production mode to a more intensive, higher-input/output production mode (Powell et al. 2004), which will increase the burden on dryland ecosystem services. Once the limits of ecosystem services provided by dryland are exceeded, irreversible ecosystem collapse may occur. Some studies also indicate the increased demand for agricultural land and forest products from population growth may offset the increased vegetation cover driven by increased CO_2 and precipitation in sub-Saharan Africa (Brandt et al. 2017).

By 2015, 700 million people still lived below the extreme poverty line (Steele et al. 2017). Eradicating extreme poverty and hunger are the first two of the sustainable development goals (SDGs) (Assembly 2015). Developing agriculture and increasing land productivity can directly increase the income of the poor and are essential to achieve the SDG2-zero hunger initiative (Mason-D'Croz et al. 2019). Studies have shown that increasing the crop productivity of land (Pingali 2012) and expanding arable land area are effective methods of reducing poverty rates. However, for fragile dryland ecosystems, the ecological problems caused by agricultural development cannot be ignored. On the one hand, the expansion of arable land leads to the reduction of woodlands and grasslands, crucial for soil and water conservation. On the other hand, the water resources consumed by agriculture further exacerbate the water stress in drylands and intensify the instability of dryland ecosystems. This will increase the cost of local ecological compensation (McLeman et al. 2014). Therefore, sustainable agricultural development is an effective way to address population growth and poverty reduction in dryland regions. In China, labor transfer and relocation have also proved to be effective means of poverty alleviation. By transferring the rural idle surplus labor, land is no longer the only way for people's labor force investment in drylands, and the local ecological pressure has been alleviated. Relocation is one of the ways of land consolidation. Relocation not only increases the employment opportunities for the people in remote areas, but also improves the stability and resilience of dryland ecosystem by ecological restoration after relocation.

2.5 Prediction of Dryland Changes

In the future, the world will face multiple interrelated challenges, including climate change, global income levels and population increases, as well as an increase in food consumption. Therefore, the simulation of future land use will play a vital role in achieving sustainable development goals through more rational land use management and effective global policy and technology improvement (Popp et al. 2017). Modeling is the major way to predict potential land dynamics in the future, such as pattern formation and ecosystem responses to disturbances. Scenario-based land use simulation model could assess the overall situation of future land use based on various scenarios (Liao et al. 2020; Chen et al. 2020a), which is different estimation of future population growth, economic development, and other socioeconomic and environmental variables. However, the spatial distribution of various land use is hard to determine. In contrast, the spatially explicit assignment of predicted land use changes to specific grid cells involves a complex process (Li et al. 2017; Liang et al. 2021), during which the relationship between land use changes and their driving forces must be accurately determined. Existing literature has successfully applied cellular automata, Markov chains, and neural networks to the spatial prediction of land use change and designed a series of models, such as the future land use simulation (FLUS) (Li et al. 2017) and patch-generating land use simulation (PLUS) model (Liang et al. 2021), as well as surface modeling of land cover change (SMLC) and dynamic conversion of land use and its effects (Dyna-CLUE) models (Sun et al. 2012). Although several land use change simulation products have been proposed based on the existing scenario data or actual models (Li et al. 2017; Dong et al. 2018), few concentrates on the land cover projection in dryland regions considered to be the Earth's largest biome.

Population growth, climate change, and economic development will certainly alter land use structure and pattern in future. Based on the IPCC SRES scenarios, Li et al. (2017) produced a high-resolution future land use simulation product with a human–environment interaction system. In this simulation, the farmland area is more related to population, which means increase farmland are more likely to emerge in general and greater increase would occur in an economic emphasizing future. Simulation result Liao et al. (2020) support the same view, in which the cropland in North China Plain is expected to expansion due to rapid population growth and demand for biomass energy. The increased farmland comes at the cost of natural landscapes (Li et al. 2017). Thus, both grassland and forest areas experience significant losses. Further, future changes in pasture in the drylands of western China are highly dependent on changes in the demand for animal products and the efficiency of livestock production systems (Liao et al. 2020). Future pasture shrinkage is most pronounced under scenarios of lower demand and higher efficiency of livestock production.

However, there are still some defects in the model used. Firstly, the description of relationship between driving factors and land use change is too simple. The response of land use to changes in social-economical-environmental conditions are view to be very complicated and follow multiple linear and non-linear patterns with abrupt and gradual shifts. But in the models, this complex correlation is simply descripted by nonlinear, multivariate regression or machine learning based statistical technology. Secondly, the description variable for driving factors is too less. For example, in the simulation of dryland ecosystems, existing models, such as FLUS (Li et al. 2017) and PLUS (Liang et al. 2021), often use indicators of accessibility, such as the distance to settlements and the density of the road network, to reflect the intensity of human activities in the study area. This is logical in practice in that the areas with poor accessibility generally have small populations, weak human activities, and thus low impacts on ecosystems. However, the simple use of accessibility as a factor to characterize the intensity of human activity hinders further improvement of model simulation accuracy, so it is necessary to design more other indicators or their combinations to describe the impacts from various driving factors.

Moreover, the coupling of land use simulation models with other models provides an effective way to study the changes in ecosystem structure, services, and processes in drylands under various scenarios in the future. Coupled PLUS and GMOP models were used to analyze land use changes and their resulting changes in ecosystem service values under different scenarios (Li et al. 2021b), while the trade-offs and synergistic relationships of ecosystem services under different scenarios were explored in Yili Valley based on the coupling of PLUS and InVEST models (Shi et al. 2021). However, the uncertainty of data source and model leads to uncertainty of simulation results. How to eliminate and evaluate this uncertainty becomes very important.

2.6 Perspective

Dryland land cover, as a geographic feature and physical land type, is the synthesis of observable natural and artificial objects within water limited regions of the land surface of the Earth. As one of the significant and special land cover in drylands, vegetation pattern can not only intercept and retain essential resources, indicate the health of local landscape (Ludwig et al. 2007), but also have the potential to indicate the tipping of ecosystem and improve the resilience of ecosystem (Rietkerk et al. 2021; Scheffer et al. 2009). As one of the most characteristic biological landscapes in dryland region, soil biocrusts are the pioneer species of desert vegetation succession and play an important role in promoting the evolution of desert vegetation (Qi et al. 2021; Zhao et al. 2021). The establishment of photovoltaic systems in dryland region is a growing impact of human activities on dryland ecosystems and its impact on local climate and ecosystem function must be more comprehensively determined with future research (Kruitwagen et al. 2021). However, all the issues mentioned above are subject to further study with more and finer resolution data, including field survey and remote sensing.

Dryland ecosystems are water-limited, which is obvious as dryland plants tend to maximize water usage during short rainfall windows and optimize the ratio of the plant growth rate to water consumption (Stringer et al. 2021). Remote sensing technology provides new insights into the calculation of the basin water balance equation and human available water for consumption after the separation of human induced and natural water consumption (Wu et al. 2018). Land cover changes exert a strong impact on the hydrological system, as they can change the amount of ET and precipitation, which significantly alters the water system input and consumption (Lian et al. 2021).

The dynamics of drylands and the response of drylands to environmental change are complex. As mentioned above, environmental changes and irrational land use interact and together lead to the degradation of dryland ecosystems. Aridification due to global warming (Yuan et al. 2019) would lead to the expansion of dryland areas and the degradation of dryland ecosystems (Berdugo et al. 2020). In contrast, global aridification associated with rapid warming since the late 1970s is largely attributed to anthropogenic increases in greenhouse gas emissions (Edenhofer et al. 2013). Mediated by the complex direct and indirect factors, the future trajectory of drylands dynamic remains unclear and is an active research field in the context of rapid climate change and intensifying human activities. Although a variety of models have been proposed and applied to simulate land use transformations, there is still a high uncertainty across models and scenarios (Molotoks et al. 2020). In addition, although models such as structural equations and multiple response-diffusion models have been used to explain the drivers of impact factors on dryland dynamics (von Hardenberg et al. 2001; Li et al. 2021a), the mechanisms by which each driver mediates changes in land use dynamics are still largely unknown. Understanding these complex and interacting drivers of dryland dynamics has a long way to go.

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Chapter 3
Structure, Functions, and Interactions of Dryland Ecosystems

Xiubo Yu, Yu Liu, Shuli Niu, Wei Zhao, Chao Fu, and Zhi Chen

Abstract Understanding the interactions between the structures and functions underlying regime shifts in dryland social-ecological systems (SESs) and how they respond to climate change is critical for predicting and managing the future of these ecosystems. Due to the high spatiotemporal variability and sensitivity of drylands ecosystem to natural and anthropogenic disturbances, it is challenging to predict the state shifts of dryland SESs. This theme delves into the mechanisms and geographical heterogeneity of resilience and the maintenance of the stability of dryland SESs that involve threshold behaviors. We emphasized the importance of considering both biotic and abiotic factors to identify the factors that drive the evolution of ecosystem structures and functions in drylands. The research frontier involves understanding how ecohydrological and socioeconomic processes drive the evolution of dryland SESs in a geographically diverse and scale-dependent context, developing comprehensive indicators, models, and multivariable approaches, and the development of effective management strategies that can maintain the sustainability of dryland SESs in the face of ongoing global environmental changes.

Keywords Ecosystem structure and function · Stability · Resilience · Regime shifts · Climate change

3.1 Introduction

The structure and function of dryland ecosystems, as well as their interactions with the social and economic systems of the inhabitants, are key determinants of the stability, resilience and sustainability of dryland social-ecological systems (SESs) (Fig. 3.1).

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However, the dryland ecosystems are highly heterogenous in space and time and are subjected to a range of natural and anthropogenic disturbances that can pushing ecosystems to tipping points and even cause regime shifts. Understanding the biotic and abiotic mechanisms underlying the structure and function of dryland ecosystems is essential for promoting sustainable development, enhancing the delivery of ecosystem services, and building resilience to global environmental change. In this chapter, we explore the structure, functions, and interactions of dryland ecosystems, as well as the challenges and opportunities for their management and conservation.

Drylands are characterized by a patterned ecosystem structure, spatial heterogeneity in functional attributes, and extensive structure–function interactions (Berdugo et al. 2020; Buxton et al. 2022; Maestre et al. 2021; Mayor et al. 2019; Meloni et al. 2019; Saco et al. 2018; Stavi et al. 2021). In drylands, the extreme climatic conditions and rapid ecosystem processes make the ecosystem highly sensitive to drivers. Both ecosystem structure and function are prone to fluctuations and changes (D'Odorico et al. 2013; Bestelmeyer et al. 2015). Globally, aridification caused by climate change has significantly affected the structure and function of dryland ecosystems, resulting in systemic and abrupt changes in multiple ecosystem attributes (Berdugo et al. 2020; Maestre et al. 2021). As drylands continue to expand, more than 20% of the terrestrial surface projected to cross one or several of these

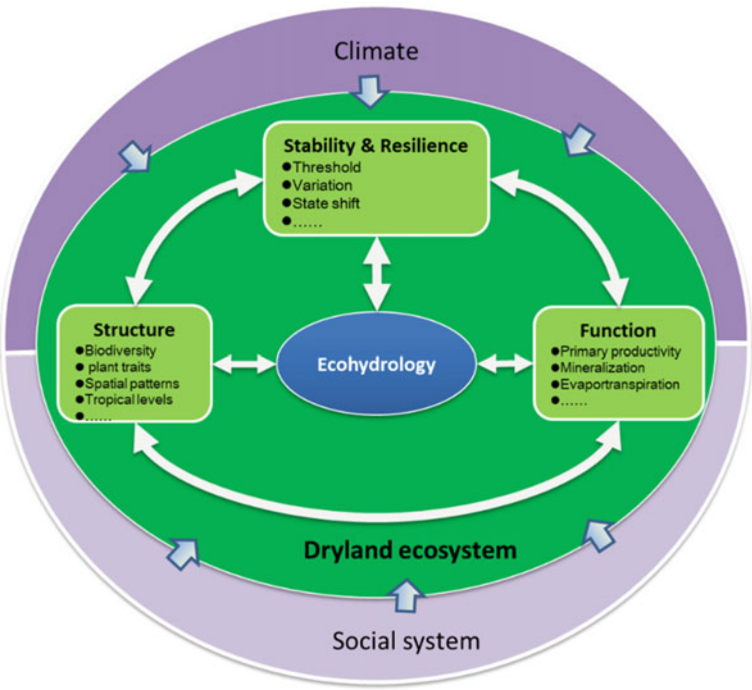

Fig. 3.1 The interrelations among structure, function, and stability in dryland ecosystems

thresholds by 2100 (Berdugo et al. 2020). Understanding the spatial heterogeneity of vegetation assemblages and biodiversity along environmental gradients, particularly those of water availability and aridity, is essential. Moreover, the spatial organization and evolutionary trajectory of natural dryland ecosystems along environmental gradients must be addressed (Berdugo et al. 2020). Given the importance of dryland ecosystems for human well-being, there is growing interest in understanding their structure, functioning, and interactions with human societies, as well as in developing strategies to conserve and sustainably manage these ecosystems.

3.2 Dryland Ecosystem Structure and Functions

3.2.1 Dryland Ecosystem Structure

Dryland ecosystem structure refers to the abiotic and biotic structures and their spatial patterns in arid and semiarid regions. It includes physical and chemical components of the environment, such as spatially heterogenous soil type, topography, and climate, as well as varying biotic components, such as vegetation, animals, and microorganisms (Schowalter 2011). Dryland ecosystems habitat various organisms, including mammals, birds, reptiles, and insects etc. They play important roles in nutrient cycling, pollination, seed dispersal, and other ecosystem processes. The soil in dryland ecosystems is typically low in organic matter and nutrients and can be prone to erosion and desertification. The physical and chemical properties of the soil greatly influence plant growth and ecosystem functioning. Dryland ecosystems are characterized by a variety of plant communities adapt to the unique climatic and environmental conditions of the area. The diversity and composition of vegetation can significantly affect ecosystem functions, such as nutrient cycling, carbon sequestration, and soil quality. Water is a limiting resource in dryland ecosystems. Its availability and distribution greatly influences the structure and function of ecosystems (Hoover et al. 2021; Zhou et al. 2021). Some dryland ecosystems rely on seasonal precipitation and intermittent water sources.

The structure of dryland ecosystems fluctuates or change rapidly (D'Odorico et al. 2013; Bestelmeyer et al. 2015). Dryland ecosystems have evolved over millions of years in response to various environmental factors including climate, topography, soil characteristics, and biotic interactions (García-Palacios et al. 2018; Bestelmeyer et al. 2015). There is great spatial heterogeneity and large variety of plant and animal communities coexist in close proximity. This heterogeneity is driven by the patchy distribution of resources, such as water and nutrients, and the variable climatic conditions that shape the landscape. Consequently, dryland ecosystems feature with a high degree of adaptability and resilience (Maestre et al. 2011). Organisms inhabiting these ecosystems have evolved various strategies to cope with drought, heat, and other environmental stressors. For example, many desert plants have deep roots that allow them to tap into water stored deep in the soil, whereas others have evolved mechanisms to store water within their tissues (Liu et al. 2013).

3.2.2 Dryland Ecosystem Functions

Sustaining the productivity, soil erosion control and mitigating water, carbon and nutrient cycling are important functions of dryland ecosystems. They support the livelihoods of millions of people who depend on them for food, fiber, and fuel. Dryland ecosystem functions are highly interconnected, and changes in one function can have a cascading effect on others. Changes in vegetation cover can affect the water balance of the system, redistribution of materials, which in turn affects ecosystem productivity and nutrient cycling (Turnbull et al. 2012; Maestre et al. 2021; Mayor et al. 2019). Similarly, changes in nutrient availability can affect the diversity and productivity of vegetation, and affects carbon sequestration and water balance.

Functioning of dryland ecosystems is highly sensitive to environmental changes (Tietjen et al. 2010; Maestre et al. 2021; Moreno-Jiménez et al. 2019). Climate change and land-use change alters the ecosystem processes and finally the ecosystem services (Smiraglia et al. 2016). Natural and anthropogenic disturbances (Eldridge and Greene 1994, Bochet et al. 2021) have important role in shaping the dynamic of dryland ecosystem functions (Zika and Erb 2009; Abel et al. 2021). For instance, water and nutrient cycling processes in drylands are highly responsive to environmental changes. Changes in precipitation patterns and temperature regimes affect the productivity and composition of vegetation, which in turn affect carbon and nutrient cycling. Disturbance events such as fires have long-lasting impacts on carbon and nutrient cycling through altering vegetation cover and soil properties. Nitrogen is often a limiting nutrient in drylands. Nitrogen fixation by biological soil crusts has been found to be a crucial process in maintaining the nitrogen balance of dryland ecosystems (Belnap and Lange 2003). Furthermore, inputs of anthropogenic nitrogen from sources such as fertilizer application and atmospheric deposition can have significant impacts on the functioning of dryland ecosystems (Yahdjian et al. 2011). Grazing is fundamental livelihood for residents in dryland over the world. It mitigates dryland ecosystem structure and functions in various avenues (Eldridge and Greene 1994). It affects vegetation cover, species composition, and nitrogen fixation of biological soil crust in dryland steppe (Liu et al. 2009). Overgrazing can led to the reduction of hydrological function (Vandandorj et al. 2017), enhances soil loss and plant invasion (Belnap et al. 2009), which negatively impact the water cycling and productivity of dryland ecosystems at various scales. Understanding the functioning of dryland ecosystems and how they are affected by environmental change is critical for sustainable management. Further researches is needed to address the remaining knowledge gaps in dryland ecosystem functioning and develop effective management strategies that balance conservation and development goals in dryland regions.

3.2.3 Structure-Functions Interactions in Dryland Ecosystems

Understanding the structure–function interactions in dryland ecosystems is fundamental for enhancing ecosystem services supply and reducing the negative impact of environmental change. There is necessity to incorporate structure–function interactions into dryland ecosystem management strategies and call for future research to explore the potential trade-offs among management goals. Ecosystem structure is influenced by ecosystem functions, and can be defined as the minimal or parsimonious pattern of organization required for a function to operate (Müller 1997). Understanding the interactions between ecosystem structure and function at multiple spatial scales has significantly increased our understanding of how terrestrial ecosystem functions respond to global environmental change (Maestre et al. 2021; Turnbull and Wainwright 2019). Among the structure–function interactions, interaction between biodiversity and ecological functions is a critical question in ecology (Peterson et al. 1998). Biodiversity, particularly species richness, has been shown to positively affect ecosystem functions at all spatial scales and trophic levels (Maestre et al. 2021). Biodiversity has a significant impact on ecosystem functions at different hierarchical scales. Therefore, it is crucial to consider biodiversity in ecosystem management (Oliver et al. 2015). A focus on resilience rather than the short-term ecosystem functions and delivery of services, along with the consideration of specific underlying mechanisms, can bridge the research areas of biodiversity-ecosystem function and ecological resilience. This can ultimately aid the development of evidence-based yet flexible ecosystem management strategies. Perennial plant species richness has been found to be crucial for ecosystem function in dryland systems (Midgley 2012), which has global relevance, particularly for developing and least-developed countries facing desertification trends. However, a comprehensive understanding of how biodiversity across trophic levels, in conjunction with abiotic drivers, determines ecosystem function is still needed. For instance, earlier experimental studies and approaches undertaken in temperate grasslands have yet to address this multilayered question.

In dryland landscape, surface cover pattern is of critical importance for the structure–function interaction. The vegetation structure participates in regulating the water balance of ecosystems by mediating the interception and infiltration of rainfall as well as the redistribution of water within the ecosystem (Doerr et al. 2000). Besides, soil structure, which regulates nutrient cycling and carbon sequestration in dryland ecosystems (Lal 2015) is important in sustainable management strategies. Soil organic carbon accumulation in drylands is influenced by a range of factors including climate, vegetation cover, and soil texture (Lal 2004). Restoring the vegetation cover in degraded drylands can improve carbon and nutrient cycling, increase soil water retention, and prevent soil erosion. The conservation of biological soil crusts also help maintain soil stability (Guo et al. 2008; Rodriguez-Caballero et al. 2013) and nutrient cycling (Belnap et al. 2016)in drylands.

The ecosystem structure of drylands interacts with functions through multiple feedbacks, particularly hydrological feedback (D'Odorico and Bhattachan, 2012). As showed by Fu et al. (2021) in Mediterranean drylands, the land-use changes have significant impacts on soil carbon and microbial diversity, which have implications for ecosystem functions, such as nutrient cycling and water regulation. It was illustrated that the composition and diversity of plant communities led to changes in ecosystem functions such as carbon and nutrient cycling, which reshapes soil properties, such as organic matter content and soil texture that affect plant growth and nutrient availability, and finally the ecosystem productivity and resilience to drought (Maestre et al. 2012).

3.3 Structure-Function Interactions Driven by Global Change

3.3.1 Structure–Function Interactions Along Aridity Gradients

Complex relationships exist between vegetation structure, ecosystem functioning, and environmental factors along aridity gradients (Maestre et al. 2015; Hu et al. 2021; Migliavacca et al. 2021). Studies focusing on the vegetation response to climate change and the identification of the aridity threshold have highlighted the importance of monitoring subtype dynamics and transition zones for predicting and mitigating the impacts of climate change on dryland ecosystems. For example, the relationship between vegetation cover and primary productivity is nonlinear and varies depending on the amount and timing of rainfall (Zhao et al. 2021). A aridity threshold of 0.54 has been identified as a critical point beyond which plant productivity undergoes abrupt changes, leading to systematic changes (Berdugo et al. 2020). Aridity has been found to reduce vegetation growth in drylands worldwide, although patterns vary among ecosystems and climate zones (Xu et al. 2018). Regional differences in ecosystem responses to climate change can be attributed to the varied responses of drylands to aridity gradients (Huang et al. 2016). The evolutionary trajectory of vegetation activity in response to variations in aridity in drylands plays an important role in predicting future ecosystem functions under global climate change. The expansion of drylands has negative consequences on carbon sequestration and increases the risk of ecosystem degradation (Huang et al. 2016). However, the response of vegetation to changes in aridity can vary across different subtypes and is regulated by the structure–function interactions within ecosystems. Abrupt changes in each subtype may result in a shift in drylands (Zhao et al. 2021). Drying and wetting trends are common climatic phenomena observed over long time (Zaitchik et al. 2023; Sheffield et al. 2012; He et al. 2019), and have a significant impact on plant growth and vegetation cover in drylands worldwide (Feng and Fu 2013; Huang et al. 2016). Small changes in the aridity may have a significant impact on the vegetation status because the response of vegetation to wetting or drying differ greatly in dryland ecosystems. However, the heterogeneity in the magnitudes and areas of drying and wetting trends may cause misinterpretation of the terrestrial and atmospheric feedbacks, leading to an inaccurate assessment of the "dry gets drier and wet gets wetter" paradigm (Greve et al. 2014; Huang et al. 2016).

The trajectory of vegetation trends in response to temporal changes in aridity was highly dependent on the spatial aridity gradient. Changes in spatial aridity can result in distinct vegetation responses to drying and wetting owing to differences in the capacity for resistance to drought among different subtypes (Xu et al. 2018). Resistance capacity is highly dependent on spatial aridity, with increased spatial aridity index (AI) improving the ability of plants to withstand drought. In drylands, vegetation activity is constrained by water availability (Zhao et al. 2020) and is positively correlated with AI (Huang et al. 2016). Normalized difference vegetation index (NDVI) trends shows regional differentiation along the aridity gradient, with lower vegetation cover and productivity in drier regions, resulting in lower NDVI trends than those in wetter regions (Xu et al. 2019).

Box 3.1 Soil nutrient changes in the northern dryland of China—a long-term observation of the Chinese Ecosystem Research Network (CERN)

Soil comprises the largest carbon and nitrogen pool in terrestrial ecosystems, particularly in drylands dominated by grasslands and deserts. More than 90% of carbon and nitrogen are stored in the soil, making it an essential indicator of ecosystem health (Sharrow and Ismail 2004). The balance of soil nutrient elements is reflected in the soil carbon and nitrogen content, and their ecological stoichiometric characteristics, which are primarily influenced by regional water and heat conditions. The total amount of soil carbon and nitrogen varies greatly due to soil formation factors and human activities, resulting in varying soil carbon–nitrogen ratios across temporal and spatial scales (Wang and Yu 2008). The soil carbon–nitrogen ratio serves as an effective method for evaluating soil quality (Walker and Adam 1958) and ecosystem health (Yang et al. 2017), representing the integration of ecosystem function variability. Therefore, long-term field observations of soil nutrient content are of significant importance for understanding the carbon sink potential of the ecosystem and predicting its response to future climate change.

Long-term monitoring data from the Chinese Ecosystem Research Network (CERN) stations have revealed the spatial patterns and temporal evolution of soil carbon and nitrogen characteristics from 2004 to 2018, including their response to climate change (temperature and precipitation). The content of soil organic carbon and total nitrogen in China's dryland ecosystems exhibited strong spatial differentiation, with an increasing trend from west to east, but the carbon–nitrogen ratio remained relatively stable. Between 2004 and 2018, the carbon and nitrogen content in semi-arid drylands showed an increasing trend, leading to improved soil quality, whereas arid drylands did not experience significant changes. Carbon–nitrogen content and ratio were significantly positively correlated with precipitation and negatively correlated with temperature, indicating the significant influence of hydrothermal control. By continuously monitoring soil carbon and nitrogen characteristics and their temporal and spatial changes in climate sensitivity, it is possible to provide a scientific basis for accurately assessing and predicting soil quality and ecosystem health.

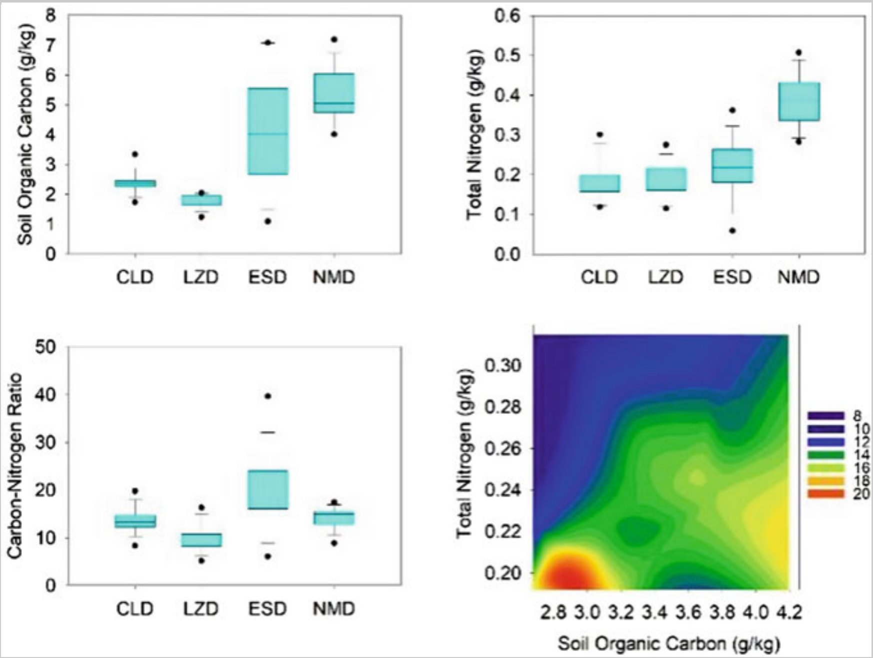

Fig. 3.2 Carbon–nitrogen characteristics of soil in ecological stations in drylands. CLD, Cele Desert station; LZD, Linze Desert station; ESD, Erdos Desert station; NMD, Naiman Desert station

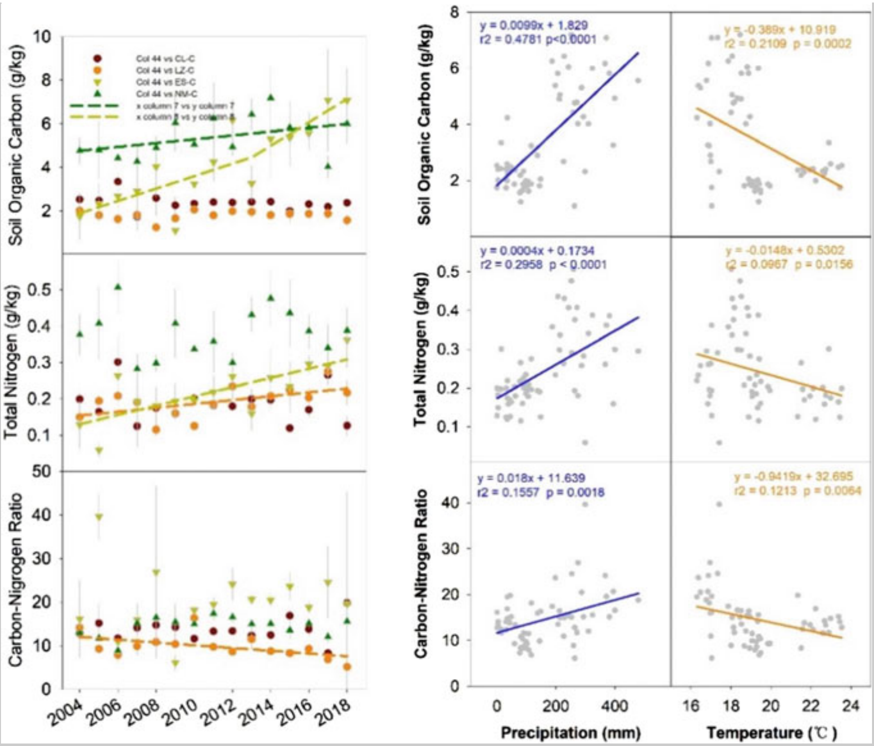

Fig. 3.3 Variation of soil organic carbon content, total nitrogen content, and carbon–nitrogen ratio (dash lines represents the regression with significant trends at 0.05 level); relationship of soil organic carbon, nitrogen, and carbon–nitrogen ratio with climate

The results demonstrate that soil carbon and total nitrogen content increase from west to east with decreasing aridity (Figs. 3.2 and 3.3). From 2004 to 2018, soil carbon significantly increased in semi-arid drylands, including ESD and NMD, while no significant variation was observed in arid drylands, including CLD and LZD. This implies a carbon sequestration potential in semi-arid drylands under climate change. However, the variation in soil nitrogen content did not present an obvious pattern along the aridity gradient, suggesting a potential risk of the impact of soil nitrogen limitation on ecosystem structure and function change. Soil carbon and nitrogen content are significantly controlled by hydrothermal conditions, showing positive and negative correlations with precipitation and temperature, respectively. Future research should explore the impact of the interaction of precipitation and temperature on soil carbon and nitrogen content and the differences in temporo-spatial patterns along the aridity gradient. Furthermore, studying soil-vegetation and soil-microbe activities is necessary to explore the climate-carbon feedback under global change. Investigating the biochemical cycle of the soil–vegetation–atmosphere continuum and exploring the evolution characteristics of the ecosystem and its response to climate change is essential to increase the terrestrial ecosystem carbon sink and ecological health.

3.3.2 Responses and Feedback of Dryland Ecosystem to Climate Change

Recent climate change research largely confirmed the impacts on ecosystems and provides a greater mechanistic understanding and geographic specificity for those impacts (Grimm et al. 2013). Pervasive climate change impacts on ecosystems affect their productivity or ability to process chemical elements. The combined impacts of wildfires and insect outbreaks decrease forest productivity, mostly in the arid and semi-arid West. Forests in wetter regions are more productive owing to warming. Shifts in species ranges are so extensive that by 2100, they may alter the biome composition across 5–20% of the US land area. The accelerated loss of nutrients from terrestrial ecosystems to receiving waters is caused by winter warming and intensification of the hydrological cycle. Ecosystem feedback, especially that associated with the release of carbon dioxide and methane from wetlands and the thawing of permafrost soils, magnifies the rate of climate change (Grimm et al. 2013).

Global environmental changes rapidly alter the structure–function relationships in ecosystems. Changes in precipitation patterns, nutrient inputs and losses, plant photosynthesis rates, and extreme climatic events reshape vegetation cover patterns and can cause unexpected, abrupt, or catastrophic shifts in ecosystems, resulting in the loss or gain of ecological resources (García-Palacios et al. 2018; Mayor et al. 2019; Ursino 2019). Alterations in precipitation patterns can lead to changes in soil moisture and nutrient availability, which can affect microbial activity and nutrient cycling (Huang et al. 2015). These changes can affect primary productivity, carbon storage, and soil fertility (Jobbágy and Jackson, 2000; Schlesinger et al. 1990). Despite the high vulnerability, sensitivity, and fluctuations of dryland ecosystems, there is a lack of verification and predictive power for catastrophic responses to changing environmental conditions, particularly in large-scale ecosystems (Rietkerk et al. 2004; Ursino 2019).

Climate change has been identified as a significant driver of vegetation activity, with reciprocal impacts on land surface cover changes (Schimel et al. 2000; Nemani et al. 2003). The effects of climate change include both interannual and long-term trends that have substantial impacts on vegetation growth (Ryo et al. 2019). In water-limited ecosystems, increased precipitation can lead to enhanced vegetation growth, which, in turn, creates biophysical feedback to the climate system, resulting in increased evapotranspiration, surface cooling, and precipitation (Davin and de Noblet Ducoudre 2010; Yu et al. 2020). In the Patagonian rangelands, along with climate, vegetation structure is also an essential factor in shaping ecosystem functioning (Gaitán et al. 2015). Maintaining and enhancing vegetation cover, particularly grasses, is crucial for mitigating the adverse effects of climate change on ecosystem functioning.

In recent decades, global greening and deforestation have caused significant changes in vegetation dynamics with implications for climate feedback mechanisms (Piao et al. 2015; Chen et al. 2019a; 2019b; Davin and de Noblet-Ducoudre 2010; Strassburg et al. 2012; Seymour and Harris 2019). As global warming surpasses the optimal temperature for vegetation growth, there is a limited safe operating space for these ecosystems (Huang et al. 2019; Xu et al. 2013). Furthermore, dominant climatic factors vary spatially and temporally, leading to potential differences in vegetation activity with climate change and spatial variations. To develop better predictive models for the trajectory of dryland ecosystems under future climate scenarios, research efforts must focus on improving our understanding of these variations (García-Ruiz et al. 2011). Cold and high-latitude ecosystems have experienced a 16.4% decline in vegetated land area limited by temperature owing to rapid warming (Keenan and Riley 2018). In Australia, the precipitation threshold for water limitation of vegetation cover declined significantly from 1982 to 2010 as the vegetation adapted (Ukkola et al. 2015). Deforestation results in surface warming, whereas surface greening leads to cooling effects (Davin and de Noblet-Ducoudre 2010; Yu et al. 2020). Reforestation in Europe has resulted in divergent responses to summer temperature changes (Davin et al. 2020).

Box 3.2 Climate change and vegetation dynamics in the Euro-Asia Transect

The Euro-Asia Transect (EAT) is a crucial component of regional ecological security and the global carbon cycle. Encompasses the Mongolia Plateau, Loess Plateau, Central Asia, and Mediterranean (Fig. 3.4). The majority of the EAT is dryland and has a delicate ecosystem consisting of dry and desert grasslands that are sensitive to climate change. Vegetation cover is critical for preserving environmental stability, providing food and livelihood security, and improving soil quality, among other benefits (Ravi et al. 2010; Xu et al. 2017). However, global warming and extreme climate events have increased the vulnerability of vegetation activity and prompted the degradation of the fragile ecosystems, negatively impacting the sustainability and human well-being in the EAT. Therefore, investigating the contribution of climate change, particularly the interaction of precipitation with temperature on vegetation dynamics, is essential for ecosystem management and predicting the future of global drylands.

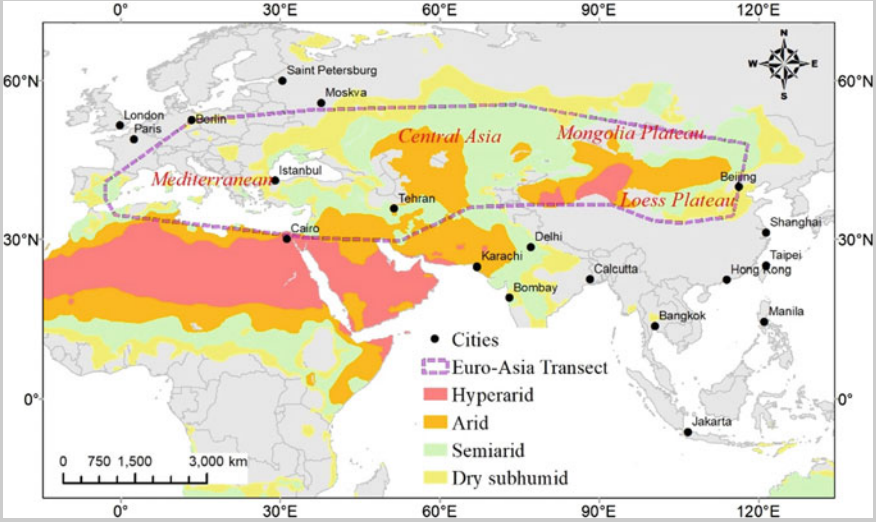

Fig. 3.4 Location of the Euro-Asia transect drylands. Drylands are divided by aridity index into four subregions: hyperarid, arid, semiarid, and dry subhumid

Contribution of climate change to the variability of Normalized Differential Vegetation Index as the proxy of vegetation dynamics in the Euro-Asia Transact (EAT) are quantified using the GeoDetector Model (Fig. 3.5). Results demonstrated that precipitation (PRE) contributed much more than temperature (TEM) to the EAT vegetation dynamics, and the interaction of PRE with TEM had an enhanced positive effect on the NDVI variability compared to the sum of PRE and TEM contribution. Regions with PRE contribution below 10% account for more than half (53%) of the EAT dryland, and that between 10 and 30% account for 40% of the studied area. Patterns of TEM contribution is similar to that of PRE, regions below 30% account for about 98% of the EAT dryland. Importantly, the interaction of PRE with TEM exerted a significant breakthrough impact on the vegetation index variability. Regions of the interaction contribution of PRE with TEM that are above 30% account for 46% of the EAT. Moreover, those that are between 10 and 30% account for 51% of the area. Furthermore, 86% of the EAT dryland showed the enhanced positive effect of the interaction of PPT with TEM which is even larger than that of the sum of PRE and TEM contribution, remarkably improving the interpretation of climate change impacts on vegetation dynamics.

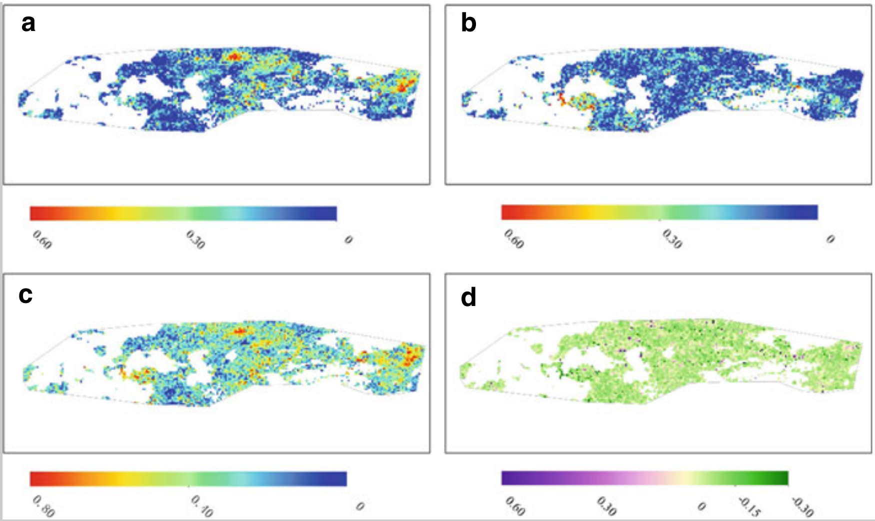

Fig. 3.5 Contribution of climate change to vegetation dynamics in the Euro-Asia transact drylands. **a** Contribution of PRE to NDVI variability; **b** Contribution of TEM to NDVI variability; **c** Contribution of the interaction between PRE and TEM to NDVI variability; **d** Difference between contribution of PRE-TEM interaction and the sum of PRE and TEM to NDVI variability

Recent studies reveal that the present 'hot model' is poor in performance in climate simulations and exaggerates the earth and ecosystem impacts (Hausfather et al. 2022; Voosen 2022). The EAT dryland occupies a large area as a widely inland ecosystem. It is substantially relevant because of the biochemical cycles and geo-physical feedback between the terrestrial and atmospheric domains through carbon-climate responses. Quantifying the contribution or the association between climate change and vegetation dynamics is important to understand the mechanism of the terrestrial carbon sink effect through plant physio-ecological process and surface physical-feedback. Multi-site comprehensive and comparative studies based on long-term field monitoring would provide confident parameter modules for the ecological processes and earth models to improve the accuracy of carbon-climate interaction simulations and predictions.

Anthropogenic disturbances, such as land-use change, overgrazing, and desertification, affect the impacts of climate change on drylands, highlighting the need for improved monitoring and modeling tools to accurately capture ecosystem dynamics in response to climate change (Verwijmeren et al. 2013; He et al. 2011; Richardson et al. 2019). Understanding the complex interplay between the structure and function of drylands, as well as their responses to global changes, is critical for the sustainable management and conservation of dryland ecosystems. Interdisciplinary research and multiscale approaches along with the integration of new technologies and methods are necessary to achieve this goal.

3.3.3 The Geographical Diversity of the EvolutionTrajectory of Dryland

The trajectory of dryland evolution is influenced by various geographic factors, including climate, topography, soil characteristics, and biotic interactions. These factors vary widely across regions, resulting in diverse patterns of vegetation cover, nutrient cycling, and ecosystem functions (Maestre et al. 2021). In addition, the interactions between vegetation and soil properties can influence the ability of the ecosystem to resist and recover from disturbances, such as drought, fire, and grazing. Furthermore, the effects of climate change on dryland ecosystems can vary across spatial scales, from individual plant responses to regional changes in precipitation patterns (Schlaepfer et al. 2017). This variability can also be influenced by interactions between different components of the ecosystem, such as plant–herbivore interactions, feedback between vegetation and soil processes, and the effects of climate on biotic communities (Schwinning and Sala 2004). Because of the effects of climate change on drylands vary across biophysical and socioeconomic contexts (Doerr et al. 2000; Maestre et al. 2016), as well as the specific adaptive capacity and resilience of each system (Bardgett et al. 2021), the geographically diverse structures and functions of different drylands bring high uncertainty in the trajectory of local dryland changes under climate change (Berdugo et al. 2022).

The geographically diverse human activities also participated in making the diverse trajectory of dryland ecosystem dynamic. Overgrazing, land-use change, and drought, which are geographically diverse, can also affect the trajectory of dryland evolution (Belnap et al. 2009, Eldridge and Greene 1994). Grazing can lead to soil erosion, vegetation loss, and changes in nutrient cycling, contributs to desertification and land degradation (Osem et al. 2013). However, well-managed grazing systems that take into account the ecological processes of dryland ecosystems can positively affect biodiversity, soil health, and carbon sequestration (Teague et al. 2016). Agriculture can also have significant impacts on dryland ecosystems, particularly through the use of irrigation, which can lead to soil salinization and waterlogging, and the use of agrochemicals, which can negatively impact soil health and biodiversity (Lal 2015; Cherlet et al. 2018). However, sustainable agricultural practices such as conservation agriculture and agroforestry can contribute to the restoration of dryland ecosystems by enhancing soil health, biodiversity, and carbon sequestration (González-Sánchez et al. 2016; Mbow et al. 2014). Urbanization can also have significant impacts on dryland ecosystems, including habitat loss, fragmentation, and changes in microclimate (Grimm et al. 2008; Seto et al. 2012). Green infrastructure, such as parks and urban forests, can contribute to the restoration of dryland ecosystems by providing habitats for wildlife, improving air and water quality, and reducing the urban heat island effect (Nowak et al. 2006).

Predicting the trajectory of local dryland ecosystem changes under climate change can be challenging because of the high uncertainty associated with the interactions among these diverse factors (Verwijmeren et al. 2013). Since the trajectory of dryland evolution is complex and influenced by multiple factors, including climate change, land-use changes, and social and cultural dynamics, the effective management and restoration of dryland ecosystems require interdisciplinary and integrated approaches that incorporate the ecological, social, and economic aspects of these systems. By combining scientific knowledge with local knowledge and engaging local communities and stakeholders, sustainable and effective strategies for the conservation and restoration of dryland ecosystems can be developed.

3.4 Stability and Resilience of Dryland Ecosystem and Implications for Restoration

3.4.1 Ecosystem Stability and Resilience in Drylands

Ecosystem stability and resilience are intrinsically determined by the interactions between ecosystem structure and function (D'Odorico et al. 2013; Maestre et al. 2021). The presence of alternative stable states can have unexpected and serious consequences due to anthropogenic environmental changes and natural perturbations (Schröder et al. 2005). Dryland ecosystems exhibit slow changes in structure but rapid responses to external drivers and feedback (Bestelmeyer et al. 2015; Saco et al. 2018). The critical role of ecohydrological feedback in driving state changes in dryland ecosystems has been previously highlighted (Saco et al. 2018). Multiple types of feedbacks, particularly ecohydrological feedback, couple dryland ecosystem structures with their functions (D'Odorico et al. 2013). The relationships between structure and function at various spatial scales are indicative of how terrestrial ecosystems respond to global change (Maestre et al. 2021) and are intrinsic determinants of ecosystem state change (Mayor et al. 2013; Saco et al. 2018). Descriptions of ecosystem stability depend on the scale of observation, including physical and temporal scales and subjectively chosen indicators. Resilience refers to the ability of an ecosystem to recover from disturbances and maintain its functions over time (Folke et al. 2004). It describes the ability of an ecosystem to resist changes in response to environmental stressors (Holling 1973). Descriptions of ecosystem stability also depend on the scale of observations, with different indicators and processes becoming relevant at different spatial and temporal scales (Huston 2014). Aridity gradients have a significant effect on ecosystem stability (Maestre et al. 2012). Dryland ecosystems may experience frequent disturbances, such as droughts and fires, which can disrupt their functions and reduce their stability (Reynolds et al. 2007). However, these ecosystems may also exhibit unique adaptations to water scarcity, such as efficient water use and high carbon storage in soils (Maestre et al. 2021). The resilience to different magnitudes of drought and different ecosystems to environmental stressors should be addressed. Comprehensive indicators that cover both "slow" and "fast" variables are needed for describing the resilience of dryland ecosystems. Spatio-temporal variations in resilience, coupled with water input pulses, should be included.

Structure–function interaction plays fundamental role in determining the stability and resilience of the dryland ecosystems. The physical structure of dryland ecosystems, including soil composition, vegetation cover, and hydrological patterns, plays a crucial role in determining ecosystem functions, such as carbon sequestration, nutrient cycling, and water retention. These ecological functions affect the structure of the ecosystem, creating a feedback loop between structure and function. One of the central questions regarding the interactions between ecosystem structure and function is how biodiversity relates to ecological functions (Peterson et al. 1998). Increasingly diverse ecosystems have a greater probability of including species with disproportionate positive or negative effects on ecosystem functioning (Naeem et al. 2009). Mounting evidence indicates that biodiversity enhances multifunctionality (Chen et al. 2018; D'Odorico et al. 2013; Maestre et al. 2021) and increases the stability and resilience of ecosystems in changing environments (Folke et al. 1996). However, ecosystem structure and function in drylands rely heavily on water availability and feedback to water redistribution and cycling by changing the physical environment (Saco et al. 2020; Urgeghe et al. 2010) or the physiological capacity for water transpiration (Fisher et al. 2011). Among all the structural features, the spatial and temporal patterns of vegetation cover in drylands are crucial for ecosystem functioning, and mitigates ecosystem stability and resilience. Restoring vegetation cover in a degraded rangeland ecosystem improved the soil structure and water infiltration, leading to enhanced plant growth and carbon sequestration (Doerr et al. 2000; Valone et al. 2002). Structure–function interactions in a degraded grassland ecosystem showed that increasing plant diversity through the restoration of native plant species enhanced ecosystem functions, such as soil carbon sequestration and nutrient cycling (Zhang et al. 2020). Increasing soil organic matter through the incorporation of cover crops in a degraded dryland ecosystem in the Great Plains of the United States improved the soil structure and water retention, leading to enhanced carbon sequestration and increased resilience to drought (Blanco-Canqui et al. 2020). Thus remote sensing derived information on vegetation cover, biomass, and productivity are important indicators of ecosystem stability (Zhang et al. 2013).

Soil carbon stability and greenness are two key indicators adopted to assess ecosystem stability along the aridity gradient. Soil carbon stability is an important measure of ecosystem health as it reflects the amount of carbon stored in soils over long periods and contributes to global carbon cycling (Köchy et al. 2015). Greenness, on the other hand, is a measure of the amount of vegetation cover and photosynthetic activity in an ecosystem, which can be indicative of its overall productivity and health (Zhu et al. 2015). By examining how these indicators vary across an aridity gradient, we gain a better understanding of the factors that contribute to ecosystem stability and how they respond to environmental change. The diversity and abundance of plant and animal species provide important information regarding the ecosystem's functions and services (Díaz et al. 2016). Additionally, the functional traits of species, such as their water-use efficiency and tolerance to drought, can help explain the patterns of ecosystem stability along the aridity gradient (Siefert et al. 2015). By examining ecosystem stability along the aridity gradient at different spatial and temporal scales and using a range of indicators, we can gain a comprehensive understanding of how these ecosystems respond to environmental changes and the factors that contribute to their stability.

Dryland ecosystems can exhibit high resilience to climate change. However, this resilience may vary across spatial scales and may be influenced by factors such as land-use change and management practices (Fu et al. 2021). In addition to changes in vegetation, climate change has led to alterations in nutrient cycling, soil moisture, and other ecosystem processes in drylands. These changes have been shown to have cascading effects on entire ecosystems, ranging from microbial communities to larger mammals (Eldridge et al. 2016). Owing to their unique combinations of stressors, regional ecosystems may differ considerably in their normal ranges of primary and secondary productivity, species composition, diversity, and nutrient cycling, making the patterns of their responses to stressors highly variable and unpredictable (Rapport and Whitford 1999). Therefore, understanding the mechanisms and geographical heterogeneity of resilience and sustaining the stability of dryland ecosystems are critical scientific tasks that should be included in dryland ecosystem research.

3.4.2 Mechanisms of Maintaining Resilience and Stability

Dryland ecosystems are characterized by limited water resources. Thus, ecohydrology plays a crucial role in connecting ecosystem structures and functions. The ecohydrological status of a dryland ecosystem determines its resilience and stability based on water availability (Fig. 3.6). This suggests that, in areas with high water availability, water inputs can be increased through water management or more efficient water use through improved ecosystem structure and function. Because water availability is critical in dryland environments, the relationships between ecosystem

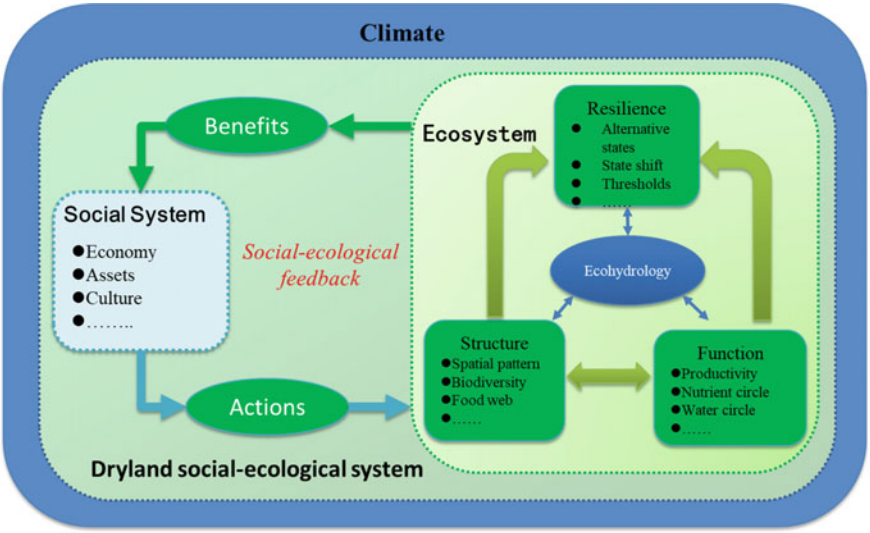

Fig. 3.6 Ecohydrology is essential in understanding the interaction among structure, function, and resilience of dryland ecosystems, and their coupling to the social system

structure and multifunctionality in drylands are unique compared to other ecosystems. Interactions relevant to ecohydrological behaviors should be focused on, such as water redistribution among vegetation patches, the effect of vegetation organization on runoff redistribution, and rainfall portioning. The organization of vegetation at different spatial scales influence the redistribution of water and nutrients, ultimately affecting ecosystem productivity and resilience. Temporal variability in water availability can lead to shifts in plant community composition and function, highlighting the need for long-term monitoring and evaluation of ecosystem responses to environmental change in drylands.

Plant traits, such as root length, hydraulic conductivity, and stomatal conductance, strongly influence water-use efficiency and drought tolerance of dryland plant communities (Díaz et al. 2016; Pillar et al. 2018). Dryland ecosystems exhibit complex patterns of connectivity and feedback between vegetation and hydrological processes that have important implications for the resilience and stability of these systems (Feng et al. 2016). Zhou et al. (2016) investigated the impact of vegetation organization on runoff generation in semiarid grasslands, whereas Zhang et al. (2020) examined the effects of shrub introduction on soil properties and implications for revegetation in a dryland.

Restoration activities that focus on enhancing soil health and water storage capacity can also help build resilience to impacts of climate change such as drought and desertification (Hoover et al. 2021). This highlights the importance of understanding the ecohydrological behavior of dryland ecosystems to develop effective management and restoration practices.

Understanding the interactions between the structure and function of dryland ecosystems at multiple spatial scales is essential to manage these ecosystems and mitigating the effects of global environmental change (Maestre et al. 2021; Turnbull and Wainwright 2019). Vegetation plays a critical role in regulating dryland ecosystem processes such as carbon and nutrient cycling, water availability, and soil erosion control. The spatial arrangement of vegetation cover, including the distribution of vegetation patches and gaps, can have a significant impact on ecosystem processes such as water infiltration and runoff (Doerr et al. 2000; Wang et al. 2015). Nitrogen-fixing plants or deep-rooted plants can enhance ecosystem processes and resilience to environmental stressors such as drought or nutrient limitation (Maestre et al. 2021; Ochoa-Hueso et al. 2019). The effectiveness of vegetation in regulating ecosystem processes may depend on the spatial scale of observations and the characteristics of the landscape context (Maestre et al. 2012; Zhang et al. 2016). Vegetation cover can affect soil microbial communities and nutrient cycling, which, in turn, can affect plant growth and productivity (Delgado-Baquerizo et al. 2016). Management strategies promoting the maintenance of healthy and resilient dryland ecosystems should be developed.

3.4.3 Ecological Restoration of Dryland for SES Sustainability

Dryland SESs are complex systems that involve interactions between natural resources, ecosystems, human activities, and institutions. They are highly sensitive to climate change. Restoration of degraded drylands has gained increasing recognition as a potential approach for enhancing ecosystem resilience and stability. Restoration practices that increase plant functional diversity can enhance ecosystem resistance and resilience to drought, nutrient limitation, and other stressors (Irob et al. 2023; Lepš et al. 2018; Hallett et al. 2017). Restoring soil organic matter can improve soil water-holding capacity and nutrient availability, as well as promote soil aggregation and microbial activity (Zhao et al. 2023). Plant diversity can enhance ecosystem resilience to climate variability and disturbance and increase soil nutrient cycling and carbon storage (Maestre et al. 2012; Soliveres et al. 2014).

Restoration practices that increase plant functional diversity include a range of techniques such as planting diverse mixtures of native species, using seed mixes that incorporate a variety of functional traits, and promoting the natural regeneration of vegetation. In arid and semi-arid ecosystems, restoration projects that incorporate a mix of deep- and shallow-rooted plant species can enhance soil water availability and reduce erosion (Maestre et al. 2021). In degraded grasslands, restoration efforts that promote a diverse mix of grasses, forbs, and legumes can enhance nutrient cycling and improve soil fertility (Hobbs and Norton 1996). Martínez-Vilalta and Lloret (2016) found that the functional diversity of vegetation was positively correlated with the resistance of Mediterranean shrublands to drought and that restoration practices increased functional diversity and improved their resistance. Increased functional diversity also enhance ecosystem resilience by promoting faster recovery after disturbances (Lavorel and Grigulis, 2012). By promoting plant functional diversity, restoration efforts can help build more resilient and productive ecosystems that can better cope with environmental stressors and provide valuable ecosystem services to human communities.

Developing new approaches to evaluate and monitor the success of restoration efforts is critical for promoting effective restoration practices. New approaches for monitoring and evaluating the success of dryland restoration efforts have emerged, including the use of remote sensing and machine learning to assess changes in vegetation cover, soil properties, and other indicators of ecosystem health (Wang et al. 2018). Remote-sensing technologies are used to monitor vegetation dynamics and ecosystem processes at different spatial and temporal scales (Brandt et al. 2018; Wang et al. 2023a). Detecting changes in vegetation cover, biomass, productivity, soil moisture, and other important variables can indicate the effectiveness of restoration efforts. Social-ecological monitoring frameworks have been developed to assess the impact of restoration on both the ecological and social components of dryland ecosystems (Verwijmeren et al. 2013). Such integrated approaches can help ensure that restoration efforts are not only ecologically effective but also socially just and sustainable. These approaches have the potential to provide more accurate and timely information on the effectiveness of restoration practices and support adaptive management and decision-making.

3.4.4 Ecosystem Management and Structure–Function Inter Action in Drylands

The management of SESs has been scrutinized because of the rising degradation of ecosystems caused by human activities, which poses a threat to human well-being (Clark and Dickson 2003; Folke, 2006; Ostrom 2009). Drylands face the challenge of low water availability, which limits both the material support services of ecosystems and human livelihoods. To address these issues, scientists, governments, and organizations have launched numerous international programs aimed at developing transdisciplinary research-based strategies for problem-solving and SES management (Carpenter et al. 2012). Recent research has shown the potential of nature-based solutions to combat desertification and enhance the resilience of dryland ecosystems to climate change (Bekele et al. 2021; Seddon et al. 2021). These approaches include agroforestry, soil conservation measures, and water-harvesting techniques, which can improve soil quality, enhance biodiversity, and provide multiple benefits to local communities. By combining traditional and modern knowledge systems, context-specific solutions that build on the strengths and resources of local communities can be developed. Other recent studies have highlighted the importance of understanding the socio-ecological dynamics of dryland ecosystems and the potential for collaborative governance approaches to support effective management and restoration efforts (Bawa et al. 2021; Fu et al. 2021; Schlüter et al. 2019). These approaches prioritize the engagement of local communities and stakeholders, the incorporation of multiple knowledge systems, and the building of social networks and partnerships to support ongoing learning and adaptation.

Interactions between various components of dryland SESs can be nonlinear and exhibit thresholds, making it difficult to anticipate abrupt changes and tipping points that have significant consequences for ecosystem services and human well-being (Folke et al. 2010; Scheffer et al. 2012; Allen et al. 2016; Berdugo et al. 2022). Impacts of climate change on dryland SESs differ substantially and depend on factors such as the availability and distribution of water resources, land-use patterns, soil quality, biodiversity, socioeconomic conditions, and governance arrangements, which vary greatly across different dryland regions (Intergovernmental Panel on Climate Change (IPCC) 2019; Millennium Ecosystem Assessment 2005). Furthermore, human activities, such as land-use change and water management, can have significant impacts on the structure and function of dryland ecosystems, further complicating the predictions of their trajectory under climate change (Bastin et al. 2017). Efforts have been made to develop effective management and conservation strategies for the species. For example, global assessments, such as the Intergovernmental Panel on Climate Change (IPCC) and the Global Drylands Initiative (GDI), have identified key drivers of dryland degradation and potential pathways for restoration and adaptation (Intergovernmental Panel on Climate Change (IPCC) 2019; Global Drylands Initiative (GDI) 2021). In addition, advances in remote sensing and modeling techniques have allowed for more accurate mapping and monitoring of dryland ecosystems at regional and global scales (Brandt et al. 2018). The potential feedback between climate change and human activities, such as land-use change and water management, can create additional uncertainties and complexities that are difficult to capture in predictive models (Dearing et al. 2015; Wang et al. 2023a). Emerging technologies such as remote sensing and big data analysis offer new opportunities for monitoring and understanding the dynamics of dryland ecosystems, including the impacts of climate change and human activities (Wang et al. 2023b; Omuto et al. 2010). These technologies can support more accurate and timely monitoring of ecosystem health and inform the design and evaluation of management and restoration strategies.

As explored before, the study of SESs is an expanding scientific field (Colding and Barthel 2019), and it is important to establish a shared analytical framework and consistent standards to compare effective management strategies. An assessment strategy is required to ensure the practical application of effective SES management strategies. However, quantifying the complexity of social, economic, and ecological factors remains challenging due to the lack of long-term social survey data and accurate mathematical models of SESs. In dryland areas, the structure and function of the SESs are closely tied to water availability and are highly vulnerable to natural and human-induced disturbances. There are three associated themes need to be addressed to restore and manage dryland ecosystem for SES sustainability: (i) the spatial and temporal pattern of the evolution of SESs; (ii) the response and feedback of SESs under climate change and the implementation of management strategies; (iii) the assessment of the interaction of natural and societal measurements and their mechanisms. Together, these aim to explore the nexus trajectory of the nature and society coupling mechanism and determine early warning indicators for identifying SES regime shifts (Fu et al. 2021), which will help provide a basis for SES management strategies and human well-being. Further work is also needed to draw on other disciplines to develop appropriate indicators for the simultaneous resilience of multiple ecosystem functions. Efforts towards ecosystem restoration in drylands should strive to increase resistance to environmental stressors while promoting long-term stability. This can be achieved through a combination of measures, such as planting drought-resistant species, improving soil health, reducing erosion, and managing human activities. The successful restoration of dryland ecosystems requires an integrated approach that addresses both the ecological and social dimensions of the system.

Box 3.3 Restoring dryland ecosystems through a social-ecological framework: the smart grassland management system and community-based agriculture reorganization in Inner Mongolia, China

The subregions of global drylands exhibit variations in ecosystem structure and functions, land degradation, and human dependence. Nature-based solutions provide the basic principles for adaptive and sustainable management of Social-Ecological Systems. To cope with future climate change and promote the achievement of the United Nations Sustainable Development Goals, countries worldwide are taking practical actions by adjusting livelihood strategies and ensuring ecosystem stability. In Inner Mongolia, China, activities such as afforestation for carbon sequestration, smart grassland management, and community-based agriculture have been implemented, resulting in significant improvements in carbon sequestration, reduction of land degradation, and enhancement of people's livelihoods.

In 2010, the Horinger Ecological Restoration Project was launched in Inner Mongolia, China by the Laoniu Foundation and The Nature Conservancy. The project aimed to enhance carbon sequestration capacity and ecosystem services through ecological restoration and livelihood optimization, and to explore trade-offs between natural ecological protection and utilization, and technical engineering for system restoration. The project includes afforestation, gully management, slope restoration, community-based agriculture reorganization, and smart grassland management (Figs. 3.7 and 3.8). The forestry carbon sequestration project in Shengle International Ecological Demonstration Zone in Inner Mongolia was certified with a gold certificate in climate, community, and biodiversity in 2013 under the Clean Development Mechanism (CDM). The carbon storage of this planted forest is expected to reach 220,000 tons of CO_2 by 2041, of which 160,000 tons have already been traded and subscribed by Warner Disney.

Fig. 3.7 Restoration of grassland ecosystem structure and function can be achieved through a combination of engineering and biointegration techniques for gully management and smart grassland management. This involves implementing measures such as enclosure and controlled grazing to improve surface vegetation cover and biomass, which in turn helps to maintain the health and stability of the grassland ecosystem

Fig. 3.8 Applying technology to increase and stabilize agricultural yields, and developing feeding and animal husbandry practices to reduce damage to the ecosystem

The project aims to plan ecological restoration, protection, and sustainable development and prioritize protection to comprehensively restore and manage the natural ecosystem and artificial production system for carbon sink enhancement. In conjunction with regional development and the needs of residents, the project promotes systematic ecological restoration and social development cooperation. Activities such as compensation, environmental education, and technical training are carried out to promote scientific farming, develop water-saving techniques, climate-smart agriculture, and explore the tradeoff between food production and economic prosperity in local communities. The project implements precise and intensive management of seasonal grazing and establishes an information platform for comprehensive evaluation of the grassland. Based on the balance of grass and livestock, a seasonal grassland management system is established to determine grazing area and time according to the sustainable utilization potential of the grassland and to promote the balanced development of animal husbandry and grassland production in society.

3.5 Mechanism for Regime Shifts in Dryland SESs

3.5.1 Overview of the Regime Shift and Its Impact on SESs

Regime shifts are associated with critical transitions in ecosystem functioning and structure, which can occur in ecosystems due to changes in environmental conditions or management that cross a threshold, leading to large, sudden, and often undesirable changes in the system (Carpenter et al. 1999; Scheffer et al. 2015). According to Scheffer et al. (2001), regime shifts can occur when a complex system is pushed past a tipping point, where the system suddenly reorganizes into a new state with different ecosystem functions and structures. Bestelmeyer et al. (2015) suggested that regime shifts in dryland SESs can occur because of slow or rapid responses to changes in external drivers and feedback within SESs. It is important to address context-specific socio-ecological feedback in drylands that involves threshold behaviors to prevent or mitigate the negative impacts of regime shifts.

Regime shifts can be triggered by a variety of factors such as climate change, land-use change, and changes in biotic interactions (Scheffer et al. 2015). In drylands, regime shifts likely occur due to the vulnerability to land degradation, which can lead to decreased ecosystem resilience and increased susceptibility to droughts and other disturbances (Doerr et al. 2000). Furthermore, dryland SESs are often characterized by complex feedback between social and ecological processes, which can make predicting and managing regime shifts challenging (Folke et al. 2004). Prolonged droughts or changes in precipitation patterns can cause a decline in vegetation cover, leading to soil erosion, increased water runoff, and reduced soil fertility. These changes can create a positive feedback loop in which the loss of vegetation cover leads to further degradation of land and a shift towards a new ecological state with different dynamics and feedback. Socioeconomic factors such as population growth, urbanization, and globalization can also contribute to regime shifts by creating new demands for natural resources and changing the way in which they are managed. By addressing the underlying drivers of regime shifts, such as climate change, land use, and socioeconomic conditions, it may be possible to reduce the risk of ecological degradation and promote more sustainable and resilient dryland ecosystems.

The responses and feedback of SESs to regime shifts are complex and context-specific. In some cases, the negative impacts of regime shifts can trigger further changes in the social and ecological components of a system, leading to a spiraling decline (Cumming et al. 2005). In other cases, regime shifts can lead to positive transformations, such as the emergence of new social norms, institutions, and practices that promote sustainability and resilience (Folke et al. 2010). The dynamics and drivers of regime shifts in SESs need to be paid more attention. Recent studies have highlighted the importance of cross-scale interactions and teleconnections in shaping the vulnerability and resilience of dryland ecosystems (D'Odorico et al. 2013; Wang-Erlandsson et al. 2018). Other studies have emphasized the role of social networks and learning processes in enabling adaptive governance and collective action in the face of regime shifts (Bodin and Prell, 2011; Ernstson et al. 2010). A better understanding of the social dynamics of dryland SESs is necessary for the effective management of regime shifts. Social dynamics, such as resource competition, conflict resolution, and trust among stakeholders, can strongly influence the responses and feedback of SESs to regime shifts (Bowker et al. 2012). Hence, stakeholder engagement and participation in the management of dryland SESs are crucial to ensure the long-term sustainability and resilience of these systems. The response and feedback of SESs to regime shifts requires a comprehensive and multidisciplinary approach to understand and manage effectively.

3.5.2 Approach and Indicators for Early Warning of Regime Shifts

Vulnerable dryland ecosystems readily fluctuate at large magnitude because of their intrinsic biological and physical structures and interactions, as well as the combined impacts of climate change and human activities. Consequently, the structure, functions, and interactions of fragile dryland ecosystems may change significantly (D'Odorico et al. 2013), leading to shifts among alternative stable states. When a critical threshold is surpassed, ecosystems can undergo catastrophic changes and reorganize into different states (Angeler and Allen, 2016; Turnbull and Wainwright, 2019). However, the mechanisms underlying these interactions remain controversial and poorly understood (Loreau and Mazancourt, 2013). Early warning for regime shifts in drylands can help prevent or mitigate the negative impacts of such shifts ecosystems and communities that depend on them.

Early warning of a state shift is still an issue that needs to be addressed. Recently, many early warning signals have been extracted from the spatial patterns of vegetation assembly at the local scale (Berdugo et al. 2017; Bestelmeyer et al. 2013; Nijp et al. 2019; Saco et al. 2020). Single aspects of regime shifts are often addressed because it is easier to handle the relationship between variables (Corrado et al. 2014; Zurlini et al. 2014). However, predictions based on a single indicator cannot describe the whole story of state shifts in dryland ecosystems, owing to great fluctuations, high sensitivity, and vulnerability to natural and/or anthropogenic disturbances. Additionally, increased variance and autocorrelation are potential early warning indicators that are readily used (Kéfi et al. 2014). However, they fail to predict nonlinear changes (Burthe et al. 2016), although nonlinearity (McGuire and McDonnell, 2010; Sarah 2004), feedback (Saco et al. 2018; Turnbull et al. 2012), and behavioral thresholds (Eby et al. 2017; Schwinning et al. 2004; Zehe et al. 2007) are common in dryland ecosystems. These characteristics of dryland ecosystems lead to a low predictability of state shifts caused by changes in ecosystem structure, and the difficulty in determining the tipping point for state shifts does not mention alternative stable states. Thus, understanding how state shifts because of changes in ecosystem structure, and what the tipping point is, requires an understanding of the biotic and abiotic mechanisms underlying state shifts and stability from a more holistic perspective. Furthermore, comprehensive indicators and models should be developed and multivariate approaches are necessary.

To develop effective early warning systems, it is important to consider indicators that can provide insights into the potential for a regime shift to occur. The AI can serve as an important early warning signal of ecosystem shifts in drylands. Vegetation productivity responds to moderate drying and wetting trends with increased greening; however, excessive drying and wetting can impede an increase in the NDVI (Berdugo et al. 2020; Zhao et al. 2020). The relationship between vegetation dynamics and climate change is nonlinear and complex. One reason for greening is the strong acclimation of grasslands, which cover most drylands, to climate change, allowing the development of high resilience and the ability of plants to recover from adverse conditions (Marcolla et al. 2011; Reichstein et al. 2013). However, grasslands are highly sensitive to drying, and drought events can significantly decrease terrestrial carbon sequestration (Xu et al. 2019). It is important to note that the effect of a single drought event on vegetation productivity may not be significant if it contradicts long-term trends owing to the nested hierarchical structure of complexity (Ryo et al. 2019). In addition, wetting promotes increased carbon sequestration in water-constrained drylands. However, NDVI does not increase indefinitely, as shifts in the dominance of other environmental factors can affect plant growth (Keenan and Riley 2018; Zhao et al. 2020). By utilizing the AI threshold and analyzing long-term trends, researchers can develop a better understanding of the relationship between vegetation dynamics and climate change, ultimately helping to protect vital ecosystems (Berdugo et al. 2020; Zhao et al. 2020). It is important to note that the effect of a single drought event on vegetation productivity may not be significant if it contradicts long-term trends owing to the nested hierarchical structure of complexity (Ryo et al. 2019). As aridity increases in drylands, the availability of soil water decreases, limiting photosynthesis and the ability of ecosystems to sequester carbon (Peng et al. 2013; Doughty et al. 2015; Frank et al. 2015; Xu et al. 2019). However, in wetter years, drylands can quickly become significant carbon sinks because of the less constrained availability of soil water, which leads to increased vegetation productivity (Poulter et al. 2014). Recent research suggests that "vegetation decline", which is observed in satellite data, is a key feature in the initial stage of ecosystem transition (Berdugo et al. 2020).

3.5.3 Prediction Models in Sustainable SESs

Because the regime shifts of dryland ecosystems involve changing spatial organization of vegetation, ecohydrological processes, soil loss, and their interactions (Peters et al. 2015; Hoover et al. 2021; Grünzweig et al. 2022), it is necessary to develop a comprehensive model describing how regime shifts occur. Understanding these complex relationships is critical for predicting ecosystem shifts in the drylands and developing effective conservation strategies. The biotic and abiotic mechanisms (Mayor et al. 2013, 2019) underlying the state shifts, particularly the ecohydrological determinants of dryland ecosystem regime shifts (Hoover et al. 2021), need to be explored. There is growing recognition of the need to address context-specific socio-ecological feedback in drylands that involves threshold behaviors. Prediction models should account for the impacts of climate change and their potential effects on the resilience and sustainability of SESs. Dryland vegetation distribution is determined by the spatial pattern of precipitation, and the controlling drivers vary based on climatology, terrain, and ecological regions. Soil moisture-atmosphere feedback dominates land carbon uptake variability, and biogeochemical and biogeophysical feedback is important for predicting ecosystem carbon cycles (Yuan et al. 2019; Humphrey et al. 2021; Windisch et al. 2021). The association between climate change and vegetation dynamics is modulated by water availability, which can be tracked by monitoring the soil moisture (Zhou et al. 2021; Miralles-Wilhelm, 2022; Erofeeva, 2021). Changes in soil moisture-precipitation feedback and soil moisture-carbon coupling link precipitation and vegetation growth. The asymmetrical and nonlinear relationship between precipitation and aboveground net primary productivity significantly affects the global carbon cycle (Quan et al. 2019; Moon et al. 2019; Maurer et al. 2020). Variations in soil moisture could signify changes in this association, which affects vegetation-climate coupling and dryland carbon sequestration (Zhao et al. 2021; Martínez-Fernandez et al. 2021; Zhou et al. 2021; Humphrey et al. 2021). Moderate soil moisture can maintain reasonable surface-atmosphere feedback, strengthening the association (Özkan and Gökbulak 2017). Tracing changes in soil moisture is crucial for predicting the impact of climate change on future vegetation (Fig. 3.9) and can guide the development of climate-vegetation association models.

The development of prediction models for sustainable SESs in drylands is a rapidly evolving interdisciplinary field. The integration of remote sensing, multicriteria assessments, resilience, and participatory modeling approaches is critical for addressing the complex and context-specific feedback that underlies the sustainability and resilience of these systems. A multicriteria assessment framework for evaluating the sustainability of dryland ecosystems that incorporates social, economic, and environmental indicators has been provided for policy interventions to promote sustainable land use (Wang et al. 2023b). Machine learning techniques for predicting the future states of SESs based on historical data are used. For example, using data on water flow, temperature, and other environmental variables to accurately predict water quality levels up to several days in advance, which could help inform management decisions to maintain sustainable water resources (Islam et al. 2021). Integrated model that combines social and ecological factors was developed to predict the impacts of land-use change on ecosystem services in the Yucatan Peninsula of Mexico, such as carbon sequestration, water regulation, and biodiversity (Mendoza-Ponce et al. 2018). Participatory modeling approach was used to develop a predictive model for sustainable agriculture or rangeland ecosystems that incorporated local knowledge and expertise and was able to predict the impacts of different practices

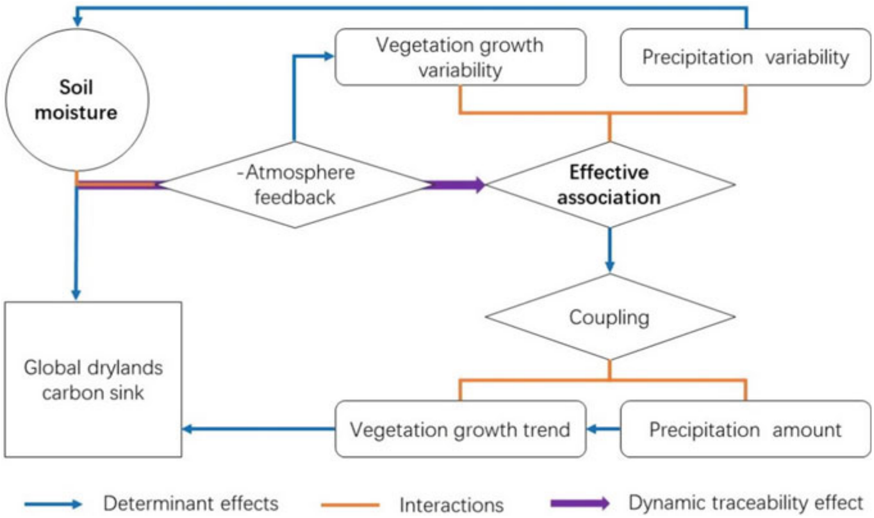

Fig. 3.9 Schematic diagram of the effect of soil moisture on carbon sinks in drylands through the traceability effect. The effective association with sufficiently high correlation coefficients indicates the coupling of vegetation growth and precipitation. Blue arrows indicate the determinant effects. Orange lines indicate the interactions among elements. The purple arrow indicates the traceability effects of soil moisture on the PRE-VI association. PRE, precipitation. VI, vegetation index (Zhao et al. 2022)

to promote sustainability (Gorripati et al. 2023). These models should be multidisciplinary and transdisciplinary and should account for the complexity, variability, and uncertainty of these systems. Trade-offs and synergies are often present in complex SESs, and prediction models should account for these interactions to avoid unintended consequences and optimize the sustainability and resilience of these systems. Since the dryland SESs are water-limited, the incorporation of impacts of climate change and the uncertainty into prediction models, along with a multidisciplinary and transdisciplinary approach, is critical for developing sustainable SES models for drylands.

3.6 Summary and Perspectives

Dryland SESs are complex and highly sensitive to both natural and anthropogenic disturbances, making them vulnerable to regime shifts that can have negative effects on the delivery of ecosystem services. To understand the mechanisms underlying regime shifts and the maintenance of stability and resilience in dryland SESs, it is essential to consider the interactions between the structure and function of these systems at multiple spatial scales. Hydrological feedback is particularly important in drylands, and climate change adds another layer of uncertainty to predicting the trajectory of local dryland SES changes. Therefore, related researches have significant implications for the management and conservation of dryland SESs, ultimately helping maintain the provision of ESs and promote sustainable development in these regions.

We highlight the importance of considering context-specific socio-ecological feedback that involves threshold behaviors in drylands. This feedback can be responsible for slow or rapid responses to changes in external drivers and feedback within SESs, ultimately leading to irreversible or persistent regime shifts. Thus, developing comprehensive indicators and models and introducing multivariate approaches are crucial for predicting the possibility of future regime shifts in dryland SESs. Understanding the biotic and abiotic mechanisms underlying regime shifts in dryland SESs and their stability from both holistic and context-specific perspectives is essential for identifying how dryland SESs change in specific contexts. This understanding can help address questions about tipping points and regime changes that may significantly affect the delivery of ESs in geographically different dryland SESs.

Future research priorities should include understanding the mechanisms and geographical heterogeneity of resilience and stability, developing comprehensive indicators and models, and introducing multivariable approaches to predict the possibility of regime shifts in dryland SESs, which identified in this theme, are critical for unraveling the complexities of dryland SESs and their interactions, predicting the possibility of regime shifts, and developing strategies to maintain their stability and resilience. The multidisciplinary nature of these research highlights the need for collaboration across various fields, including ecology, hydrology, economics, and the social sciences, to achieve a comprehensive understanding of dryland SESs and inform management decisions that promote sustainable development.

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Chapter 4
Dryland Ecosystem Services and Human Wellbeing in a Changing Environment and Society

Nan Lu, Dandan Yu, Lu Zhang, Yihe Lu, and Bojie Fu

Abstract The framework of the Global Dryland Ecosystem Programme (Global-DEP) combines the ecosystem service (ES) research paradigm and system dynamics thinking. The core of the framework is the resilience of social-ecological systems (SESs) in drylands. This resilience depends on the interaction between ecological and social subsystems. Water shortages, desertification, and poverty are currently the biggest challenges to maintaining resilience and realizing sustainable development in dryland SESs. However, the internal links between ecosystem degradation/restoration and poverty/eradication remain unclear. ESs bridge ecological and social subsystems by forming a "bonding concept" that connects environmental goals and socioeconomic goals, as ESs can directly or indirectly promote almost all land-related sustainable development goals (SDGs). Clarifying the change of ESs and their contributions to human well-being (HWB) is the key to the entangled dryland challenges, promoting the resilience of SESs and finding solutions to coordinate ecological protection and socioeconomic development. This chapter summarizes the research progress in dryland ES and its relationship with HWB in a changing environment and society. It outlines research priorities, focusing on the concept of ES and how its methodologies contribute to dryland research and management for realizing SDGs. The priorities are as follows: ES quantification; the interactions among ESs; mechanisms of ES contributing to HWB; landscape optimization for ESs; and ecological compensation.

Keywords Dryland · Ecosystem services · Human well-being · Global-DEP · SDGs

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4.1 Background and Significance of the Theme

Drylands provide important yet under-appreciated ecosystem services (ESs) that are essential to sustain the well-being of local residents and beyond. These include key provisioning services, such as the production of food, fiber, medicinal and pharmaceutical plants, timber, and biofuels. They also include a variety of regulatory services, such as water purification, pollination and seed dispersal, and climate regulation by sequestering and storing vast amounts of carbon in the soils (Yirdaw et al. 2017). The cultural services deeply rooted in people's lifestyles and beliefs in drylands are an important part of human civilization. Due to the vulnerability of drylands to climate change and land disturbances, it is critical to protect and sustainably manage the ecosystems. So far, many countries still regard biodiversity and ecosystem protection as an obstacle to economic development, ignoring nature's contributions to people (NCP). Their actions related to protection and development usually conflict (Pires et al. 2018).

ESs are not only an object of research but also an object of management. They form a common language for communication and dialogue among researchers, managers, and stakeholders, and they are a source of human well-being (HWB). Thus, it is important to develop a clear understanding of ESs in studying and governing the dryland SESs, which are close combinations of society and nature (e.g., pastures, agropastoral ecotones, agroforestry systems, desert-oasis composite systems, etc.). ESs bridge ecological and social subsystems as a "bonding concept" that connects environmental goals (e.g., ecosystem integrity and biodiversity maintenance) and socioeconomic goals (e.g., sustainable livelihood, poverty reduction and cultural heritage) (Pires et al. 2018), as ESs are directly or indirectly related to almost all land-related sustainable development goals (SDGs) (Preez et al. 2020). Since the Millennium Assessment (MA) in 2005, studies on ESs have gradually increased. Early ES studies mainly focused on the description and quantitative analysis of ESs, including biophysical quantification, valuation, and modeling. Since the efforts of TEEB in 2010 and IPBES in 2012, more studies began to shed light on the contribution of ESs to HWB. Many new keywords emerged in publications from 2010 to 2015, including payment for ESs, willingness to pay, economic valuation, and poverty. After 2015, hotspots shifted to perception, trade-offs, cultural ESs, ES flow, and protected areas (Wang et al. 2021).

Biodiversity in drylands, represented by plant, animal, and microbial diversity and diversified cropping practices (e.g., polycultures, crop rotations, cover crops, agroforestry, etc.), is considered fundamental for the resilience/stability of dryland ESs (Naeem et al. 2012). Countries with high biodiversity have great potential to promote the resilience/stability of ecosystems and socioeconomic systems through the sustainable use of biodiversity and ESs (IPBES 2019). Up to now, dryland biodiversity and ESs have been inadequately evaluated due to limited data availability, the lack of a systematic ES valuing approach, and discord between decision-makers and researchers. Dryland ESs have high temporal and spatial variations, associated with the fast-changing variables (technological change, crop production, rainfall variability, etc.) and slow-changing variables (demographics, land use, annual mean precipitation, soil fertility, etc.) that simultaneously regulate the dynamics of social-ecological processes. As the distribution of the population also presents the characteristics of decentralized aggregation, the supply and demand of ESs usually do not match across different temporal or spatial scales. Poverty, remoteness, inadequate management, and imperfect market systems of drylands all contribute to the high dependence of human livelihood on such land. Land degradation and poverty form a vicious circle, hindering the socioeconomic development.

The widely used ES cascade conceptual framework (emphasizing how ecosystems benefit human society) and the supplementary NCP framework (developed on the basis of the concept of ESs and emphasizing social and cultural attributes of ES demand) provide us with different theoretical angles for understanding the links between nature and society. Specifically, ES indicators can link biophysical and socioeconomic analysis (Boyd et al. 2015). The methodologies of ES, including trade-off analysis and supply–demand (mis)matching analysis, are favorable tools for identifying the problems in dryland SESs, and landscape optimization and payment for ES are providing solutions to managers (Dean et al. 2021). Therefore, it is of great significance to incorporate the ES concept into resilience and sustainability studies in dryland SESs.

4.2 Quantifying Dryland ESs in the Changing Environment

Quantifying ESs is an important and a basic step in understanding the spatiotemporal changes in ESs, their driving forces, and ecosystem management (Lu et al. 2018). ES valuation can reflect human needs, perspectives, and market dynamics, further linking ESs to the social domain. Spatial mapping and scenario simulations can help identify ES degradation and deficit spots under changing climate and socioeconomic conditions, and guide risk management via spatially explicit monitoring of the ecosystems (Everard and Waters 2013; Hauck et al. 2013).

4.2.1 Biophysical Modeling of ESs at Multiple Scales

Modeling is a powerful tool to quantify changes in ESs at different scales. The low availability and high variation of water are the foremost factors in ecosystem processes and functions in drylands. The variation of annual rainfall in drylands can exceed 50% of the annual average, whereas this is only 5–10% in mesic areas (Barnes et al. 2021). Some existing ecosystem models or integrated ES models (such as InVEST) have been applied to quantify and predict ES changes in dryland ecosystems. However, methods of quantifying dryland ESs are still lacking, as drylands are usually regarded as marginal areas. As few models have been developed for dryland ecosystems, simulations of dryland processes are usually poor, particularly in addressing water cycling processes (Turner et al. 2016). They can provide useful information on large-scale patterns, but the fine changes and temporal dynamics on smaller scales require single-ES models or the validation of modules in integrated evaluation models using observations of drylands.

For water-based services, accurate simulations of water balance components are fundamental. Water supply and flooding regulations are important provisioning and regulatory services, respectively, corresponding to evapotranspiration (ET), runoff, soil water storage, groundwater, and reservoirs. Improved hydrological models require the identification and inclusion of key hydrological processes of drylands at different scales, which are usually ignored in existing hydrological models (D'Odorico and Bhattachan 2012; Quichimbo et al. 2021). Accurate characterizations of the hydrological variability are particularly important. Soil water deficit could be a more important driver than atmospheric drought in terms of influencing dryland vegetation and ET, given that vegetation change in the past decades showed no significant correlations with the atmospheric aridity index (i.e., the ratio of annual precipitation to potential ET) (Berg and McColl 2021).

Soil erosion is a significant concern to land managers in global drylands, as it can lead to a reduction of soil organic matter, declining of crop yields, loss of biodiversity, and the intensification of water pollution and dust storms, further affecting food security and exacerbating poverty (Li et al. 2017). Soil retention is a key ES to the residents in drylands and beyond. The USLE Model is the most widely used model for assessing soil erosion at different scales and in different regions, based on which many other erosion models have been modified. However, these models do not perform well for some areas because of the different development purposes and applicable conditions. For example, USLE cannot accurately simulate the erosion process of gully landforms, as it is not well described for the erosion process on steep slopes (Li et al. 2017). Describing the topographic and geomorphic features of drylands and the key hydrological processes at a specific scale is crucial to better describe the erosion process in model development (Sidle et al. 2019). In recent years, new technologies such as hyperspectral remote sensing provide support for optimizing the accuracy of key parameters (such as DEM in meters).

Dryland ecosystems play a very important role in the global carbon cycle (Poulter et al. 2014). Dryland carbon flux is the main driver of variation in the global carbon flux. The greening or browning of dryland vegetation has contributed to significant changes in global ecosystem carbon sequestration over the past 30 years, especially in the southern hemisphere (such as Australia), in which the carbon sink flux increases sharply during La Niña years. By comparison, the decrease of gross primary productivity (GPP) is even greater during El Niño years. This indicates that the responses of dryland vegetation to drought and rainfall pulses are different from those of other ecosystems (Barnes et al. 2021). Understanding carbon–water (both initial water conditions and water constraints) coupling is still the key to improving modeling and spatiotemporal predictions of carbon-related ESs in drylands.

Cultivated lands are a substantial part of the dryland landscape, especially in semi-arid areas. Crop yields largely depend on rainfall and its time allocation. In current crop growth models, water-driven models show better performance than radiation- or carbon-driven models (Lu et al. 2021a, b). However, sensitivity analysis of the key parameters is lacking for dryland crop models. In addition to rainfall variability, phenology and irrigation are important factors affecting the sensitivity and uncertainty of crop models (Plaza-Bonilla et al. 2014), which include ET, soil water, and vegetation parameters (e.g., root development under high water stress and maximum canopy coverage under low water stress) (Hui et al. 2022). Only with an in-depth understanding of sensitivity is it possible to explore the best management practices for dryland agriculture.

In a word, an in-depth understanding and quantification of the hydrological dynamics of dryland ecosystems and the coupling relationships with vegetation and soil processes are crucial for accurately quantifying the critical provisioning and regulatory ESs (including water provision, soil conservation, carbon sequestration, food production, etc.) in this highly variable environment. New simulation methods need to be further innovated to improve ES quantification across spatial and temporal scales. Particularly, the model structure needs to be supplemented by finely describing the characteristics of landforms, soil hydrology, and vegetation in drylands, as well as the ecohydrological and ecophysiological responses to long-term trends and the short-term variability of rainfall (extreme rainfall and drought) to reduce uncertainties when modeling dryland ecosystems. As for non-material services, there is no model to directly simulate cultural services, and indirect simulations merely simulate future scenes.

4.2.2 ES Valuation: More Than Monetary Value

ESs have values beyond biophysical value. They have market value, non-market value, option value, and non-use value. Quantifying the social values of a specific ES can be difficult, particularly for regulatory and cultural services (Martín-López et al. 2014). Currently, ES valuation research most widely focuses on the economic or monetary value of ESs, wherein ESs are regarded as an asset that can be consumed by people and that can be considered in economic accounting. In the view of TEEB, ESs contribute to the economy by creating income and welfare, and by avoiding social impairment (TEEB 2013). In the SEEA framework, ESs that directly contribute to human society are defined as "final services", and the services flowing within the ecosystems are "intermediate services" (Hein et al. 2016). The differentiation between "final" and "intermediate" ESs is to determine the direct/indirect link between ESs and HWB and avoid double counting. Economic valuations of ESs can arouse peoples' concern regarding nature and they can provide insights on the outcomes of a specific policy or management intervention according to the marginal change of the ES value and cost effectiveness (TEEB 2013). In a quantitative review of the ES value in drylands, Schild et al. (2018) found that the monetary value of dryland ESs depends on the type of ecosystem assessed and the assessment method adopted. Farmland and forest are regarded as high-value ecosystems because they can provide food or wood. In comparison, for grasslands and semi-desert ecosystems, the monetary value of ESs is very low. Although the total amount of provisioning services that these marginal ecosystems can provide is small, they are of great significance to the livelihoods of local residents. Particularly, under some circumstances, possessing some ESs (such as crops or woods) may indicate social recognition or cultural identity, beyond merely goods for consumption or monetary value. Therefore, economic evaluations alone are not completely reliable for making predictions, tending to cause biased management actions that neglect the sustainable use of ESs with low market value. This is especially true for regulating and cultural services. Future research needs to integrate monetary and non-monetary value methods to uncover the full spectrum of values of these undervalued ecosystems and ESs, so as to avoid further neglecting and destroying these ecosystems.

A framework that considers the multidimensional value of dryland ESs is needed because the multiple values of ESs usually provide different and complementary information for ES assessments (Martín-López et al. 2014). Studies are also needed for developing more appropriate valuation tools that link the biophysical value and social value of ESs in order to obtain dynamic predictions. Some progress is being made towards integrated ES valuations by constructing integrated evaluation frameworks (Boerema et al. 2017). Such approaches introduce ecosystem dynamics into the natural capital account and evaluate the value change of the expected final service flow or the change in ecosystem capacity. Such research connects the value of ESs with the concept of ecosystem dynamics, taking into account multiple stable states, thresholds, and lag effects, with positive and negative feedback in ecosystem dynamics.

4.2.3 Drivers and Scenarios

Climate change, land cover change, urbanization, livestock grazing, biological invasion, and the economy are the main drivers of dryland ecosystems. The increases in temperature, rainfall variation, CO_2 concentrations, duration of drought periods, climate extremes, and their interactions not only have significant direct effects on the ecosystems, but also indirectly affect the processes and services of the ecosystems by changing their phenology and stoichiometry (Burrell et al. 2020; Li et al. 2021). Human activities, including urbanization, sedentarization, land-tenure change, and cropland expansion, fracture drylands into spatially isolated pieces, discouraging mobile livestock herding and accelerating land degradation (Li and Huntsinger 2011). Ecological restoration plans have brought certain ecologically positive effects to project areas, but the pressure of vegetation restoration on regional water resources cannot be ignored (Li et al. 2021). Multiple natural and anthropogenic drivers impact ecosystems at different scales. For example, at broad spatial scales, climate variables determine the distribution and dynamics of vegetation; at finer spatial scales, the successional pathway of the rangeland diverges from the regional trajectory under the pressure of livestock herbivory. That is, the mosaics of foraging suggest decoupling between climate and vegetation (Liao and Clark 2018). In the temporal dimension, the dynamics of ESs are the outcomes of the interwoven influences of the faster and slower drivers. It is fundamental to identify and monitor these driving variables to understand and predict ES changes.

Land degradation or desertification is a comprehensive representation of dryland ecosystem deterioration in responding to the interactions of multiple pressures (Box 4.1). Desertification exists widely in global drylands, although vegetation has become greener in some regions in the past decades. Prăvălie (2021) summarized 17 paths of global land degradation. The first five are drought, water erosion, salinization, soil carbon loss, and vegetation degradation. Among them, drought is the foremost factor for desertification, as it relates to 70% of the agricultural degradation of drylands. Desertification usually leads to losses of biodiversity and wildlife habitat, degradation of ESs, the decay of traditional culture and social identity, and the loss of management practices and knowledge that could help halt and reverse land degradation. It also has strongly adverse impacts on non-drylands, which may be located thousands of kilometers away from the degraded areas (i.e., spillover effects). The cascading or cumulative impact of the multiple stressors may not be additive, but rather magnified by their interactions, leading to abrupt transitions in the ESs, possibly followed by catastrophic changes in the SESs. Therefore, it is significant to understand the impacting mechanisms of the multiple stressors on dryland ESs at local, regional, and global scales (Lucatello et al. 2020), as well as the different drivers of ES modeling and the relative contributions and ecological thresholds (Wu et al. 2015; Hauck and Rubenstein 2017).

Box 4.1 Causes and Consequences of Land Degradation in Drylands

Land degradation, namely desertification in dryland, is a pervasive, systemic phenomenon, which occurs in all parts of the terrestrial world and can take many forms. Combating desertification and restoring degraded land is an urgent priority to protect the biodiversity and ecosystem services in drylands.

- **The causes of land degradation**
  - Climate change; Rapid expansion and unsustainable management of croplands and grazing lands; High consumption lifestyles; Widespread lack of awareness of land degradation; Reactive and fragmented institutional, policy and governance responses to address land degradation.
- **The consequences of land degradation**
  - The biophysical impacts include biodiversity loss, crop yeild reduction, losses of soil fertility and stability, aggravating dust storms, downstream flooding, impairment of global carbon sequestration capacity, and regional and global climate change.
  - The societal impacts relate notably to human migration and economic refugees, leading to aggravated poverty and political instability, threatening the long-established resource-use patterns.

- **Aspirations for addressing land degradation and possible actions and pathways**
  - ***Safeguarded biodiversity***. Strengthen protection of biodiversity through enlarged and more effective protected systems, halting conversion of natural land, and large-scale restoration of degraded land;
  - ***Low-consumption lifestyles***. Lower per-capita consumption patterns, including the adoption of more vegetable-based diets and low- and renewable-energy-based housing, transportation and industrial systems;
  - **Circular economy.** Reduced food loss and waste, sustainable waste and sanitation management systems, reuse and recycling of materials;
  - ***Sustainable land management***. Sustainable land management practices in croplands, rangelands, forestry, water systems, human settlements, and their surrounding landscapes, specifically directed at avoiding, reducing, and reversing land degradation (IPBES, 2018).

Scenarios that examine a range of potential futures for one or more components of a system, instead of attempting to predict just one future, have become an important tool to study the sustainability of SESs (IPBES 2019). They provide a useful tool for treating distinct possible scenarios and exploring plausible future trajectories of the direct and indirect drivers of environmental and social changes. Climate scenarios are currently mostly used to predict the extent and ES consequences of drylands. Using different drought indices, conclusions regarding whether the range of global drylands will expand in future climates are inconsistent. Huang et al. (2016) predicted that drylands would expand by 11% and 23% by the end of this century under different climate scenarios (RCP 4.5 and RCP 8.5, respectively) according to the atmospheric aridity index. In contrast, Berg and McColl (2021) found that the scope of global drylands would not expand, based on an ecohydrological aridity index. The results are also different when using other aridity variables (e.g., vapor pressure deficit, runoff, and soil water) (Lian et al. 2021). Nevertheless, with a large amount of evidence, one consistent view is that under the condition of climate warming, the frequency and severity of extreme events (including drought and fire) will be increasingly likely in drylands in the future.

Demographic, social, economic, and technical factors are also bound to change significantly in the future, and they must be taken into account in scenario assessments of dryland ESs. Population growth and agricultural expansion will be accompanied by an increase in water demand. Intensified livestock grazing and large-scale afforestation may further aggravate water shortages and trade-offs with other ESs; although the application of water-saving technology may somewhat alleviate these shortages (Lian et al. 2021). So far, studies on dryland ES predictions under socioeconomic scenarios are inadequate. Current research on causal mechanisms with modeling and controlled experiments rarely considers socioeconomic feedback (Briske et al.

2015). This knowledge gap makes it difficult to judge whether the economic development path and ecological protection measures to resolve social conflicts and environmental degradation in drylands are reasonable. Groups of scenarios, such as the representative concentration pathways, shared socioeconomic pathways, and the Global Environmental Outlook of the United Nations Environment Programme (UNEP), have many common aspects in the underlying assumptions and can be regarded as "archetype scenarios", which represent synthetic overviews of a range of assumptions about the configuration and consequences of the direct and indirect drivers adopted in the scenarios. It is necessary to simulate dryland ES changes under archetype scenarios that reflect the values and guiding principles of society, i.e., the scenarios representing the regional socioeconomic and sociocultural context (IPBES 2019). The resilience and adaptability of dryland SESs in coping with future climate and socioeconomic conditions can be informed by these scenario simulations.

Future ES assessments of drylands should be directed toward an integrated operating model to examine the mechanisms that lead to the joint outcomes of multiple drivers, how their interactions affect system transitions, and how alternative strategies may depend on socioeconomic contexts and traditional knowledge (Liao et al. 2020). To do this, site observations, modeling, remote sensing, and socioeconomic investigation must be integrated to quantify the temporal dynamics and spatial heterogeneity of ESs and to connect cross-scale findings. Spatial modeling in ES evaluations is particularly important because it can provide key information for spatially explicit decision-making and for monitoring the outcomes of decisions (Everard and Waters 2013; Hauck et al. 2013).

4.3 Interactions Among ESs

A key challenge for balancing the protection and development of drylands is to coordinate economic, social, and environmental benefits. This is important for any region, but particularly pressing in drylands (de Araujo et al. 2021). The 2.1 billion dryland residents face water shortages, and half of them are poor and dependent on cropland, rangeland, and natural systems. This requires positive interactions among the ESs provided by the ecosystems. The interactions among ESs include (1) a broad range of trade-offs or synergies between different types of ESs, or between different locations or time periods for a certain ES, and (2) the relationships between ES supply and ES demand, noting that the supply–demand balance is a (mis)match but not a trade-off. In a world of resource constraints and uneven distributions, trade-offs and supply–demand mismatches occur everywhere. ES trade-off is the core of all trade-off issues in the SESs (the others are the conflicting relationships between ecosystem multifunctions, multidimensional HWB, and management goals) (Lu et al. 2021a, b). Along the cascade from ecosystem to HWB, trade-offs are transferred from the biophysical domain to the social domain. With spatially heterogeneous and temporally dynamic human needs, the trade-offs and mismatches between ESs can be enlarged, causing complex interactions among multiple beneficiaries, locations, time periods and even human generations (Seppelt et al. 2011). In order to serve human needs and improve decision-making for a better nature as well as HWB in drylands, it is necessary to explore ways to improve positive interactions among the ESs, i.e., higher synergies among ESs and better supply–demand matches.

4.3.1 ES Trade-Offs

In dryland ecosystems, water, soil, and nutrients are limited. The trade-offs between multiple ESs can be fierce, especially for food provisions, water yield, sediment control, biodiversity, carbon sequestration, and biofuels, which are the most important conflicts for land-use choices. Social factors such as population growth, economic development, and the transition from a nomadic to sedentary lifestyle further affect ES trade-offs, and sometimes lead to ES degradation. Ecosystems are vulnerable to disturbances when their carrying capacities are exceeded. As a single result can seldom be optimized without affecting the other components of the system, trade-off analysis is required in system modeling and management practice. Understanding the main trade-offs can provide effective solutions for the decision-makers and managers.

By using correlation analysis, scenario analysis, spatial association, or overlap analysis, trade-offs have been sporadically evaluated in some studies. Most of these studies focus on the biophysical value of ESs (Dade et al. 2019). The foremost challenge for future studies is to navigate the trade-offs, i.e., tracking the change of ES trade-offs from the biophysical domain to the transformation into human needs and well-being, and trying to tackle them at different knots of the ES cascade. ES trade-offs are derived from ecosystem functions and their spatial distributions and temporal dynamics. It is difficult to define a win–win situation even for the functional traits of plants. In complex SESs, the trade-offs among stakeholders and the different dimensions of HWB can be more complex. Market systems, sociocultural preferences, and management goals all affect ES trade-offs in varied ways. To some extent, the ES valuation method shapes the trade-off outcomes. That is, the output information of the trade-off can be greatly different when using an inconsistent method to quantify the biophysical, sociocultural, and monetary value of ESs (Martín-López et al. 2014). So far, no theoretical or empirical studies have explored the mechanism of trade-off changes from the biophysical to social value of ESs in drylands (Howe et al. 2013).

Driver analysis is another challenge in ES trade-off research. Different action paths may lead to different trade-offs or synergistic consequences under the same driving factor. The failure to include mechanism analysis in trade-off assessments may lead to the mis-identifications of the effect of policy options (Dade et al. 2019; Turkelboom et al. 2018). Driving analysis has not been used in most ES trade-off studies. Existing studies usually consider changes in land use, biophysical conditions, and policy as the most commonly examined drivers, but cultural factors are rarely investigated (Dade et al. 2019). In the Loess Plateau of China, for example, afforestation in abandoned cropland led to increased soil organic matter and soil nitrogen content but decreased soil water content, and the trade-offs varied along the precipitation gradient (Lu et al. 2014). In drylands, social (e.g., water resource management and restoration policy) and environmental (e.g., climate) factors affect ES trade-offs, but this needs to be further explored at different scales. Alternative scenarios and causal inference methods can be used. A multi-process coupled ES model is advantageous in that it provides the driving mechanisms behind the trade-offs among multiple ESs by conducting scenario and causal analyses.

The SES framework originates from system thinking. However, in reality, it is usually impossible to consider all elements at the same time, and compromises are needed when considering overall benefits. The food, energy, and water nexus (i.e., the FEW nexus) has been used as a concept for addressing the key resource and environmental issues in drylands (Olawuyi 2020; Yadav et al. 2021). It is a useful tool to coordinate several ESs and a great improvement in system studies. Recent research has expanded this concept to include ecological integrity (i.e., FEWI nexus) (Müller et al. 2015), which can be used as a more developed framework for dryland trade-off solutions and sustainability. The FEWI highlights not only provisioning food, water, and energy, but also the overall ecosystem integrity and health, fundamental for regulatory and cultural services. In this sense, ecosystem management should consider not only human needs for food, water, and energy, but also the maintenance of biodiversity and natural habitats (Müller et al. 2015). FEW or FEWI does not represent three or four ESs, but bundles of ESs. However, a common caveat of these nexus frameworks is that they miss the varied value dimensions of ESs and their driving forces. It is necessary to develop more advanced frameworks that consider the trade-offs in the biophysical value as well as the socioeconomic value in order to clarify the spectrum of trade-offs from ecosystems to HWB and the driving mechanisms that regulate the interactions of the ES bundles.

4.3.2 ES Demand and ES Flow

Due to the spatial heterogeneity of dryland ecosystems and the population distribution, the supply and demand of ESs have high spatial variability and mismatch (Castro et al. 2014). Several large-scale famines in human history indicated the lack of food supply was not due to insufficient production, but rather inequitable food distribution. In recent times, the social demands have changed from dependence on provisioning services to the need for more regulatory and cultural services (Geijzendorffer et al. 2015). These changes in human needs intensify the contradiction between humans and land.

Research has also shifted from solely focusing on the aspects of ES supply (including ES quantification and trade-off analysis) to understanding the dynamic relationship between ES supply and demand. Early supply–demand analysis emphasized ES surplus and deficit analysis. The ratio or difference between ES supply and demand as well as their changing trends are used as an index for risk evaluations (Maron et al. 2018). Through risk classification, the risk grades (e.g., security, existing risk, and insufficient supply) can be identified spatially to provide a decision-making basis for risk management. For example, by establishing the dynamic and spatially explicit monitoring system of the water supply–demand balance, managers can obtain information about water deficits and abundance and then use engineering such as artificial or semi-artificial canal systems, inter-city water pipelines, and dam regulations to regulate the spatiotemporal allocation of water in a watershed or region.

Supply areas and demand areas of ESs are usually separated. With urbanization, people are migrating from rural areas to cities. Urbanization has become an important driving force that has affected dryland SESs in recent decades. Of the 1692 cities with a population of more than 300,000 across the globe, 35% (586) are located in drylands, and this number is still rising (Cherlet et al. 2018). Urban areas occupy only about 2% of the area of drylands, but they contain nearly 45% of the dryland population. The spatial connection between ES supply and demand areas has changed significantly. Cities and towns become the demand centers of ESs, while suburbs are the main supply areas of ESs (e.g., grain and livestock). Suburban residents rely on ESs provided by local ecosystems and ES flows from other supply areas. However, cities and towns rely on a variety of substantial service flows from the suburbs. ES flow, which refers to the spatial delivery of services from the supply area to the benefit area, has become a popular research interest in recent years. Besides changing the distributions of ES flows, urbanization also alters the balance of resources between rural and urban populations, as it usually encroaches on natural or agricultural lands (Seitzinger et al. 2012).

ES flow is becoming a critical concept and subject of management for alleviating mismatches in quality or quantity between the supply and demand of ESs in space and time. ES flows can be classified into four categories in terms of transportation paths: biophysical flow through species migration and dispersal, biophysical flow through processes in air, water and soil, biophysical flow of traded goods and embedded ESs through an artificial carrier, and information flow through information networks (Schröter et al. 2018). ES flows can be classified into another four categories in terms of the spatial and directional characteristics of the flows: non-proximal ES flow such as climate change mitigation, directional ES flow such as water yield, omni-directional ES flow such as pollination, and ES flow related to user movement such as cultural services (Xu et al. 2019a, b). These classifications are potentially useful for managers to make correspondingly appropriate strategies of ES delivery, but more empirical studies are needed to explore the mode and mechanism of ES flow transportation and allocation. The ES flow concept is also useful in ecological protection and the restoration of drylands to expand the areas from only those with high biodiversity and ES provisions to those with ES flow paths (e.g., vegetation corridors, waterways, and air channels).

“Telecoupling” refers to socioeconomic and environmental interactions over distances (Liu et al. 2013). It is also used to describe the occurrence of ES flow at large spatial scales (e.g., regional or global). Ecosystems are ever more affected by distant interactions among countries or regions in globalization. The telecoupling analysis framework provides a new method for analyzing the spatial correlation between ES supply and demand. In this framework, multiple supply and demand areas can be regarded as interrelated nodes in a network. The effects of local actions on systems in distant places can be noticed in ecosystem management. Spillovers are a result of these telecouplings whereby effects of seemingly unrelated events in one region can be experienced in other regions. Some studies have demonstrated the substantial impact of telecouplings on environmental benefits in distant countries, such as international trade. Another example is carbon sequestration, which has regional spillovers (i.e., improving agricultural productivity) and global spillovers (i.e., mitigating climate change) (Plaza-Bonilla et al. 2015). Network analysis is expected to become a new technical tool to better reveal the size, direction, and changes of ES flows in time and space (Liu et al. 2013). Establishing and evaluating the ES flow network is an important research direction to deal with supply–demand mismatches. Future studies should combine ES flow or the telecoupling framework with trade-off analysis (noting that it deals with multiple rather than single ESs) and investigate the spillovers.

4.4 Contributions of ESs to HWB

The internal relationship between ESs and HWB is a challenging topic. By clarifying the mechanisms between ESs and HWB, we can explain the interaction and feedback in the "circle" of poverty and land degradation in drylands (Barbier and Hochard 2018). Recent theoretical studies and sporadic empirical studies show that the key is to determine which dimensions of HWB are most relevant to ecosystems (Leviston et al. 2018).

4.4.1 Mediating Factors from ESs to HWB

HWB is multidimensional and includes basic materials, health, safety, good social relations, and freedom of choice and action. Poverty is essentially the lack of wellbeing, and it is also multidimensional. A high percentage of people living in drylands are still reliant on basic needs for survival, and poverty is the largest obstruction to social and economic development.

Although it is commonly understood that HWB depends on natural capital and services, little empirical research has been conducted to explore the mechanism of how ESs contribute to HWB. According to the review of Suich et al. (2015), there are about 250 research papers detailing the relationship between ESs and HWB. Of these, 39 articles offer a quantitative analysis, of which 21 focus on farming systems and only four on dryland ecosystems. The ESs most widely associated with poverty usually include water supply, the diversity of wildlife and crops, species and quantity of livestock, green vegetation, and peatland. For dryland SESs, soil conservation and available habitats are also highlighted (Suich et al. 2015). Some of the internal mechanisms of the transformation from ESs to HWB are more intuitive, but some may be hidden in multiple paths and processes and not easily identified.

Cruz-Garcia et al. (2017) reviewed the relationship between ESs and HWB in Africa, Asia, and Latin America. Of the 462 publications, 71% assumed that there was a link between ESs and HWB, but only 29% reported an empirical test of this hypothesis. The analyses were mainly for European and North American countries, with very few for Asia, Africa, and Latin America. Ten ES-HWB relational frameworks were used in these case studies, but 82% of the studies used the simplified framework of MA. The rest were applied only once, indicating that the current ES-HWB framework is still theoretically oriented and difficult to apply in empirical studies, especially of fisheries, wetland, and grassland systems. Also, studies on the ES–HWB relationship mainly focused on provisioning and regulatory services, with relatively little attention to cultural services (Leviston et al. 2018).

ESs and HWB relations are regulated by a range of overlapping factors in the SESs at different scales (IPBES 2019). Mediating factors are the variables that affect how ecological processes bring benefits (and their values) to people (Mandle et al. 2020; Duraiappah 2011). They are similar to the indirect drivers of ESs, including the market access mechanism, macroeconomic conditions, power and governance, tenure security, institutions and rights, and financial assets (Horcea-Milcu 2015). Mediating factors are important to consider for an accurate representation of ESs in decision-making. In dryland SESs, the core goals of coordinating all the relevant mediating factors should be combating desertification and restoring degraded land and soil. This is related to a range of SDGs. These mediating factors may affect the change and benefit distribution of ESs, ultimately affecting the realization of well-being (Suich et al. 2015). More empirical research is needed to test the connections and reveal their internal mechanisms.

4.4.2 Quantifying the ES-HWB Relations

The relationship between ESs and HWB is not one-to-one correspondent. Some methods are used to quantify the ES–HWB relationship in some sporadic studies, including ecosystem accounting, unified indicator (i.e., using a specific ES flow, carbon flow or water flow, as a unified indicator to measure ES and HWB) (Xu et al. 2019a, b), the structural equation model (SEM) (which identifies the direct and potential ES variables that affect well-being) (Santos et al. 2015), the relative rate of change (i.e., the ratio of change in HWB to the change in ecosystem services) (Daw et al. 2016), and Nexus Webs approach (Levistona et al. 2018).

One difficulty in quantifying ES contributions to HWB is that many ES and HWB indicators have different units. Ecosystem accounting aims to quantify the value of ESs to understand how much the value of ESs is involved in social capital (Lavorel et al. 2020). It is intuitive to estimate the economic value of ESs and analyze its contribution to social economy. Challenges to using the economic value of ESs include determining the economic end, avoiding double counting, and reducing uncertainties in valuation methods. Xu et al. (2019a, b) drew on a similar idea, but they directly used biophysical quantities instead of economic value, i.e., carbon flow, as a link for a variety of services and well-being indicators in a "mountain-oasis-desert" system. Santos et al. (2015) used the SEM method to quantitatively analyze the relationship between biodiversity, ESs, and HWB in a national-scale study of Spain. SEMs can incorporate many indicators (including driving forces, biodiversity, ESs, and HWB) into a Driver-Pressure-State-Impact-Response (DPSIR) conceptual framework and analyze the direct and indirect quantitative relationships among indicators, but their disadvantage is the lack of an explanation of the internal mechanism of the relationships. Within the resilience framework, Daw et al. (2016) developed the concept of ES resilience to describe the sensitivity of HWB to ecosystem changes. A high ratio of ΔHWB/Δ ecosystem stocks indicates a close relationship between ESs and HWB, and a low ratio indicates low resilience and decoupled correlations. This resilience method can be applied to compare the elasticity of different benefiting groups at different scales, which is helpful to understand the vulnerability of different social actors to ecosystem change. Some studies also found that the sensitivity of HWB change over ES change depends on the scarcity of the ESs. When the supply (relative to demand) of ESs is sufficient, a marginal increase in ESs can only lead to small changes in HWB; however, when an ES is lower than a threshold, small changes in the ecosystem may lead to a significant reduction in HWB (Liu et al. 2007). However, the application of this elasticity method in a highly dynamic environment is challenging because it is hard to determine under what circumstances the threshold of ESs will be transmitted to HWB and cause abrupt changes. Levistona et al. (2018) employed a Nexus Webs framework to investigate the inter-dependencies of ES and HWB. The Nexus Webs framework provides a method for integrating biophysical and socio-economic modeling and the assessment of HWB. Each Web contains a number of components (e.g., water, energy, and biodiversity), organized sequentially via system dynamics. The challenge of this model is to construct the linkages between the components.

Some theoretical studies suggest that ES value chain analysis and system dynamics should be combined to identify the chain reactions with biophysics and the social economy in each value chain of ESs. For example, it is unclear whether the grassland landscape improves the well-being of residents through the production of animal husbandry or tourism income. The pathways are multidimensional and nonlinear. How ESs affect people's identity cognition, values, spiritual feelings, traditional beliefs, and overall well-being remains unknown (Suich et al. 2015). Due to the complexity of SESs, the behavior of the system is often difficult to predict. System dynamics is relatively simple when analyzing supply services and regulatory services, but for some ESs (cultural services and some other regulatory services) that lack an understanding of the intermediate processes, system analysis is more difficult. More developed methods that include legacy effects, slow effects, and the complementary behavior of ecosystems are needed to better describe and predict the contribution of ESs to the welfare of humans (TEEB 2013), considering that the scale and boundaries of ESs that impact HWB. The resilience framework brings our attention to system dynamics. This framework has the potential to advance ES science and solve complex nonlinear issues in the SESs.

Efforts need to be made to refine variables that represent different dimensions of HWB corresponding to the SDGs of drylands, the demand preferences of residents in drylands for ESs (food security, water security, health, income, assets, and employment), and the influencing factors. Traditional methods to quantify HWB variables include statistics, questionnaires, and social surveys. These methods all have uncertainties associated with a small sample size, poor timeliness, low data availability, and low accuracy. A challenge and opportunity for HWB quantification is to establish a big data platform of indicators and a database of drylands. It is necessary to integrate the existing data and build a data interface for dynamic evaluations of HWB with the help of modern internet technology and artificial intelligence. At present, research of the ES–HWB relationship is mostly theoretical. Many open questions must be answered by empirical studies (Box 4.2). It is also necessary to conduct an in-depth mechanism analysis of the relationship between ESs and well-being by conducting empirical studies so as to test the validity of currently proposed methods and provide clear guidance for ES management practices. ESs and HBW (especially poverty reduction) also need to be effectively integrated into national and global sustainable development agendas and mainstream policies (Pires et al. 2021). Biodiversity is the basis of ESs and HWB, but correlation analyses with biodiversity are still insufficient. For countries with high biodiversity yet drought and poverty, it is particularly important to combine biodiversity, ESs, and HWB (Pries et al. 2018).

Box 4.2 Human Wellbeing Indicators and Key Questions

- ***HWB indicators:***
  - Food security and domestic water security (basic human needs); energy security, economic security, and sense of security (community resilience to change, connection, migration, gender, social cohesion); environmental security (sustainability); health (mental and physical health, spiritual/aesthetic value, peace, free will).
- ***Key questions:***
  - Are primary dimensions of HWB the same across different SESs?
  - Are some dimensions of HWB more critical than others? Are there trade-offs between these dimensions?
  - Which indicator of ES or NCP contributes to well-being in what way? Is this mode diversified among different SESs?
  - Are the relationships between ES and HWB direct and linear, or are there optimal ranges?
  - What roles do aspects of personal sense of control and place attachment play in moderating relationships between HWB and ES?

– What are the ‘threshold points’ beyond which ES decline has a significant, meaningful, lasting impact on dimensions of HWB, and vice-versa?

4.5 Landscape Optimization for ESs

Improving the resilience of the whole SES depends on improving the resilience of both ecological and social subsystems (Cumming 2011). In complexity theory, it is assumed that there are common potential mechanisms in different systems. We expect that the interactions between patterns and processes of social systems and ecosystems may have similarities in terms of the spatial principles and mechanisms (Cumming 2011). Spatial resilience is an important component of resilience theory. A new area of research involves applying resilience theory at the landscape scale (Allen et al. 2016). The landscape scale is a more operable scale in resilience management than local and global scales, as the local scale is too small to be included in the structure and process of the SDGs, and the global scale is too large to describe the fine mechanisms that can guide management strategy. As a geographical unit with the closest combination and the strongest interaction between humans and nature, landscapes are the proper working unit for ES optimization and sustainable path selection in drylands (Wu 2013). Understanding landscape processes, including both natural and social processes and their correlation with the landscape structure, is crucial for forecasting landscape changes and their consequences for ESs and HWB (Yirdaw et al. 2017).

4.5.1 Spatial Resilience

Spatial resilience refers to the interactions between the spatial variations of internal variables (corresponding to spatial heterogeneity), external variables (corresponding to driving feedback factors), and the resilience of the whole SES on multiple spatiotemporal scales (Cumming 2011). It is currently one of the most advanced concepts in ecology, aiming to explain the elasticity and convertibility of heterogeneous and dynamic systems. Identifying disturbances, defining boundaries, quantifying diversity, and identifying connectivity are some important procedures in spatial resilience assessments (Allen et al. 2016).

In dryland SESs, the typical concepts of “patch” and “connectivity” in landscape ecology have the potential to deepen our understanding of pattern–process relations and improve system resilience. The spatial distribution of vegetation patches and connectivity dynamics has a significant impact on ES supply, demand, and flow (the flow of ESs from a “source patch” to a “sink patch”), and also trade-offs (/synergies) and the supply–demand (mis/) matches of ESs. In the biophysical domain, ESs such as carbon sequestration, soil erosion, and crop yields are all affected by vegetation connectivity and hydrology connectivity in drylands. For example, increasing vegetation connectivity in cropland can promote pest movement and reproduction and then potentially reduce crop production, but decreasing the connectivity of natural vegetation can impede pollination; and increasing hydrological connectivity in the vegetation-bareland mosaics can increase soil erosion and water loss, leading to positive feedback between the loss of vegetation patches and an increase of bare soil patches. A review paper suggested that ESs can be negatively affected by decreasing connectivity, especially for regulatory services such as pollination (Mitchell et al. 2013). This indicates that connectivity may have multiple impacts on ES depending on ecosystem type, the expected ES, and connectivity metrics. In fact, dryland residents have been managing the connectivity of their lands throughout history, with runoff control in agricultural practices, no-tillage, farmland shelterbelts, and straw checkerboard fences for vegetation restoration. However, these practices have not been comprehensively evaluated or raised to theory (Okin et al. 2018). Similarly, in the social domain, social exclusion—that is, the unavailability of resources, ESs, and markets—is the manifestation of the fracture of connectivity in the social system. Therefore, social governance is required to strengthen the connections between the key elements that affect ES flows, such as between the locations of ES supply and demand, ES production and the market, residents and green infrastructure, and power and rights. All of the elements and relationships in both domains of SESs (i.e., ecological and social domains) have relevant spatial locations and spatial properties. The concept of connectivity provides new insight to understand dryland ecology and socioecology.

Scaling is a typical challenge in ecological and SES studies. With the spatially hierarchical structure in the SES, spatial resilience at a finer scale can provide spatial countermeasures for optimal regional layouts (Li et al. 2021). Field and Parrot (2022) conducted pioneering research to quantify the functional connectivity of three types of ES (water flow, food, and landscape aesthetics). They explored how the change of one ES provision can affect another by altering functional connectivity. Landscape ecology has the potential to apply its principles, such as corridor theory, to enrich ES flow and spatial trade-off studies and to advance resilience science (Beller et al. 2019). Spatial resilience should be one of the major considerations in landscape optimization. Landscape management and dryland restoration should be designed from the perspective of spatial resilience by establishing a multi-center and multi-scale governance system that considers inter-patch relations and connectivity (Cumming et al. 2017).

4.5.2 Landscape Optimization

Land-use management is one of the basic factors for improving the structure and multifunctionality of landscapes (Plaza-Bonilla et al. 2015). Limited to small scales, earlier ecosystem management and governance inadequately considered the concept of space (Cumming 2011). Agricultural production and many other ESs need to make the best use of the structure of land systems. This requires coordinating integrated designs of landscapes with livelihood acquisitions. Such designs are called “land system architectures” (Verburg et al. 2013). They represent the application of ecological theory to management practice for optimizing land use at the governance level. Although traditional land-use planning objectively reflects the economic function of land use, it ignores the value of multiple ESs.

Landscape optimization originates from the concept of land-use structure optimization, which aims to achieve an optimal ecological and economic solution. As a new research and management tool in ES management, the purpose of landscape optimization is to increase the resilience of the SES by optimizing the landscape, improving the provisions for ESs, decreasing trade-offs, and facilitating ES flow delivery to users. It is impossible to maximize all ESs, and this it is not the nature of optimization. In theory, it is more resilient and more effective if nothing reaches the maximum so that a certain degree of redundancy can be maintained. Such a system is more resilient to environment variability and more economically cost effective. Focusing on optimizing one specific ES is dangerous and insufficient. Rather, the focus should be on the trade-offs of multiple ESs and their connectivity (Nguyen et al.2018; Wu et al. 2018; Field and Parrot 2022).

Landscape-level ecological restoration is considered an effective way to enhance both biodiversity and the provisions of ESs (Schiappacasse et al. 2012), and it pertinent to the rehabilitation of degraded drylands. Identifying appropriate restoration methods to induce short- to long-term recovery is often hindered by inconsistent value systems, knowledge systems, and ruling institutional systems (Gorddard et al. 2016). The empirical work led by the International Network for Sustainable Drylands suggested that it is crucial to promote a transformative framework for sustainable land management considering multiple SDGs, their synergies and trade-offs (Huber-Sannwald et al. 2020), and multiple sectors or actors who determine an optimal combination and compromise of multiple ESs (Lucatello et al. 2020). Combining participatory and spatial optimization modeling can help determine the priority of investment locations to mitigate degradation, and map the supply of ESs by prioritizing the ES of a region. Then, according to the vulnerability of ESs to land degradation, the priority of important investment areas can be determined (Willemen et al. 2017). Combining biophysical and socioeconomic perspectives will help local or regional decision-making by organizing ideas and determining key system attributes (Verón et al. 2017).

Landscape assessments are the basis for landscape optimization. They are used to determine whether the spatial arrangement of the key elements of a landscape is appropriate for ES synergy and delivery before further modifications are made. Landscape optimization and assessment form a feedback process: the landscape can be further optimized based on the results of an assessment. Network analysis is a useful tool to assess the composition of local species, biogeographic modes, and social relations. Bayesian networks have been used to assess ES trade-offs and hydrological connectivity, and to support decision-making and planning in water use in drylands

(Crossman and Pollino 2018). The advantage of this method is that it integrates different forms of data, particularly in relating the potential outcomes of management interventions to a defined set of endpoints by integrating non-commensurate data values and types (McVittie et al. 2015). Spatial scenario modeling is another option in which a large number of landscape scenarios can be tested to select the most favorable ones according to varied optimization goals. For example, restoring cropland to grassland is effective to produce more water, but restoring cropland to a mosaic of grassland, forest, and shrubland is a compromise that offers relatively abundant water and higher carbon sequestration in a semi-arid watershed (Wu et al. 2018). So far, research to identify and evaluate disturbances and boundaries, diversity and redundancy, and the connectivity of multiple ESs—the main aspects for resilience assessments—is still rare (Allen et al. 2016). By only focusing on the flows of individual ESs, previous studies did not consider the interactions and feedback among ESs and how these relationships might influence landscape resilience (Field and Parrott 2017). The procedure of optimization becomes more complex when the goal becomes more oriented to improve system resilience and SDGs. Landscape optimization modeling that includes the elements of ES interaction and spatial connection will be an important research direction for future dryland ES studies.

4.6 Ecological Compensation and Payment for ESs

Ecological compensation is a positive conservation action to counter-balance the loss of ES value in resource use and management (Brown et al. 2013). The relevant projects include compensatory mitigation, biodiversity offsets, mitigation banking, habitat banking, species banking, wetland mitigation, etc. (OECD 2016). Payment for ESs (PES) occurs when a beneficiary or user makes a direct or indirect payment to the provider of ESs (for maintaining or avoiding decreases in ESs) (Nelson et al. 2008), or where the government acts on behalf of the ES buyer and makes payments as a third party (Schomers and Matzdorf 2013). While the terms "ecological compensation" and "PES" are often used interchangeably, ecological compensation is a broader term that includes PES-like policies/programs and a variety of other policy/program types (Zhang et al. 2010). Ecological compensation or PES is theoretically an effective way to achieve the "win–win" goal of coordinating ecosystem protection and socioeconomic development based on the market mechanism or financial transfer mechanism. Increasing investments in drylands is financially promising and socially rewarding. In certain circumstances, PES can create new revenue streams for conservation and has been interpreted as "making trees worth more standing than cut down" (Salzman 2011).

Ecological compensation internalizes ES externalities, and it has been applied in many countries and regions. Most cases involve national compensation plans based on government financial transfer (i.e., the Pigovian concept). Although different terms are used to describe the practices, relatively few cases are PES-like programs based on market economics through private negotiations between stakeholders (i.e., the Coasean concept) (Sommerville et al. 2009). For example, China has implemented large-scale ecological compensation in the Natural Forest Protection Project (NFPP), and the government has provided compensation to areas that experienced economic losses caused by logging restrictions and offered compensation for reforestation and sustainable forest management. In the Green for Grain Project (GfGP) with a more extensive scope, the Chinese government provided grain and living subsidies to farmers for the sake of returning farmland to forests or grasslands. This kind of conceptualized ecological compensation for PES reflects a compensation mechanism limited by national legislation (Schomers and Matzdorf 2013). On a trans-regional or transnational scale, the global environment facility (GEF) and international PES (IPES) may contribute to ecological protection and restoration on the global scale. For example, the IPES can help mitigate deforestation in the regions that contribute significantly to global climate mitigation (e.g., three-quarters of Brazil's carbon emissions come from deforestation).

By developing institutions, expertise, and market infrastructure, government-financed payments, the private sector, and NGOs have driven a rapid increase in market-based PES (Bremmer et al. 2016; Vogl et al. 2017). PES-like programs in watersheds are regarded as the most mature PES in the light of transaction value and geographical distribution (Salzman et al. 2018). However, some studies indicate that most of them are unable to demonstrate the effectiveness of PES on water-related ESs in watersheds (Brouwer et al. 2011; Yan and Joachim 2018). This is because water-related PES studies are usually based on empirically untested assumptions about the relations between land use and water flow. They lack baseline data and a control design, which are required to analyze the externalities and to determine which beneficiaries need to be paid and how much. The root cause is that most PES studies are not originally designed for a rigorous evaluation of PES effectiveness (e.g., comparison between PES and non-payment) (Salzman et al. 2018).

Ecological compensation or PES was proposed as an important measure to combat desertification and land degradation by 2030 (Li et al. 2018). However, the theoretical regime has not been well established. Assessments of ecological compensation for restoring degraded lands are complicated. It is difficult to obtain accurate estimates of the potential costs of avoiding desertification or restoring degraded drylands. Therefore, a common argument in favor of action is to add up the "damage costs" or foreclosed revenue, including the loss of ESs due to degradation, and the approximate cost of restoring a particular area. This usually generates a large amount of monetary value (Nkonya et al. 2016). Existing ES valuation methods still cannot reasonably estimate the value of all ESs, and in fact PES captures only a small part of the value of ESs. Existential value, option value, and many public goods interests are considered to be outside the scope of the PES mechanism. PES actions are often questioned for having the adequacy of the levels of compensation involved (Franco et al. 2013; Dell'Angelo et al. 2018) because inadequate PES level could reverse the initial expected potential benefits due to natural disasters (such as severe drought), reduced policy support, or greater profits through other management alternatives (Plaza-Bonilla et al. 2015). In the cases where the income flow of PES itself is not enough to motivate land owners to adopt beneficial land practices (Salzman et al. 2018), the combination of PES with other strategies such as subsidies is needed. Therefore, fairness and efficiency must be balanced under specific conditions (Bellver-Domingo et al. 2016). PES represents a relatively new policy instrument for drylands but offers great potential as an income generator. Motivated buyers, motivated sellers, metrics, and low-transaction-cost institutions are the important features for PES up-scaling (Salzman et al. 2018). Other options for a better PES design include creating new markets for ESs, such as carbon and water, and establishing subsidy programs that help land users overcome the initial costs of changing land use and management. With improved PES plans, investment in drylands can be promoted (Thomas et al. 2014).

Some researchers argue that ecological compensation or PES is unlikely to be successful for drylands if the action does not consider the goal of poverty reduction, particularly for developing countries (Plaza-Bonilla et al. 2015). Drylands are a global economic community, providing important services for life-support systems worldwide. Like biodiversity and tropical forests, drylands should be treated as global environmental commons (Stafford-Smith and Metternicht 2021). It has been argued that local or regional sustainable development policies for drylands must be included into global development agendas, by mainstreaming and coordinating funds from multiple policies and initiatives to support ecological compensation in dryland restoration (Plaza-Bonilla et al. 2015).

Importantly, targeted governance and management countermeasures should be put forward according to the characteristics of drylands. For example, the total carbon sequestration of drylands is large but distributed in a very large area that is not as concentrated as the carbon storage of forest ecosystems and thus relatively uneasy to measure. Integrating measurements, evaluations, and telecoupling analysis of ES flow and ES value is critical for drylands. A recent study reported an impressive example of payments for wind erosion control services considering regional differences. The physical quantity of wind erosion maintenance services was calculated according to weight factors such as regional GDP, population density, and dust concentration in the atmosphere, combined with the willingness to pay of the people in the beneficiary area. Then, the biophysical quantity of trans-regional and transnational ES flows are transformed into the flow of economic value, and the reference line of PES is given (Xu et al. 2019a, b). The novelty of this study is that it establishes a quantitative relationship between ES flow and PES, and provides a spatially clear visualization tool for PES policymaking from the perspective of ES flow, in which both contributors and beneficiaries are clear.

Theoretically, more rigorous metrics that align with conservation goals and accurately capture ES values and transaction costs need to be further developed (Salzman and Ruhl 2000; Maron et al. 2018). Practically, PES is feasible when the metrics are easily accessed and the exchanges and assessment mechanism are efficient for identifying ES holders (Salzman et al. 2018). Furthermore, approaches and models are needed to guide practices of PES programs to support sustainable development by integrating linkages between influencing factors, livelihood activities, and socioeconomic outcomes (Wu et al. 2021). PES still has some defects, but it can be solved by improving the design. From a research perspective, the challenge is to design PES plans from a multidisciplinary and interdisciplinary perspective, with long-term outcomes as the priority. This design cannot be limited to too small a scale when applied to drylands. It should instead deal with ES externalities with a large span of ES flows and trans-regional impacts. In addition, a deep relationship between cultural services and relationship values and land should be established, which is the internal driving force for landowners to manage ecosystems (Chan et al. 2017). A reasonable PES payment standard can promote the restoration and protection of ecosystems and maintain the sustainable supply of ES biophysical flow and value flow. And, it can close the gap between ecology and regional economic development and provide a poverty reduction path for poor groups who provide ESs, even if it cannot completely solve the problem of poverty. It also opens up a scheme that can be further designed for the realization of the goal of poverty eradication. Therefore, PES is a promising tool of environmental policy to tackle and understand the feedback between social systems and ecosystems.

4.7 Summary

An ES paradigm provides a perspective and method for analyzing the relationship between nature and people in drylands, but many theories and assumptions need to be confirmed by empirical research. Existing research on dryland ES mainly focuses on the evaluation of single services. The trade-off between various ESs, the relationship between the supply and demand, the transfer path of ESs, and the mechanism of the ES–HWB relation are still weak. Cross-scale ES trade-offs and the driving factors of the dynamics and distribution of ES flows remain poorly understood. From the perspective of system feedback, there is also a lack of sufficient practical experience on how to better formulate land use strategies and ecological compensation strategies. As global drylands contain a variety of SES types, each type has specific land use and livelihood characteristics. It is necessary to carry out systematic comparative research across different SES types on different scales, summarizing the general laws and regional dependence characteristics of the relationship between ESs and HWB. Regional comparisons and multi-site syntheses are needed to improve global modeling and the knowledge base of drylands. This is favorable for developing the connotation, methods, and paradigm of ES science, and to provide systematic experience and scientific support for formulating a sustainable development path of drylands from local to global. Future research needs to (1) establish a long-term socioecological monitoring network, (2) further develop the quantitative method of dryland ESs, optimizing the parameters of ES models and strengthening verification and scenario analyses, (3) explore the mechanism of ES change under multiple pressures, (4) clarify the path and direction of ES flow, and (5) make overall land use and PES planning at the policy and management levels. These are all important for understanding the ES–HWB relationship and for combating dryland degradation and reducing poverty.

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Chapter 5
Ecosystem Management and Sustainable Livelihoods in Drylands

Yanfen Wang, Yali Liu, Liwen Shan, Jianqing Du, Yuexian Liu, Tong Li, and Xiaoyong Cui

Abstract Drylands are very vulnerable ecosystems because of their resource constraints and environmental pressures etc. They are sensitive to a range of pressures, including climate change and human disturbance in many forms. The livelihoods of people in dryland regions must be made sustainable if the stability of dryland social-ecological systems is to be maintained. Human livelihoods in drylands are characterised by a single structure, great dependence on natural resources, and vulnerability to disruption by disturbance. In the context of global climate change and the associated expansion of arid biomes, livelihoods in drylands face growing challenges. Maintaining and rebuilding sustainable livelihoods are inseparable from good ecosystem management. However, ecosystem management is recognised as a "wicked problem" without clear-cut solutions because of the complexities involved. This chapter identifies the issues and challenges facing human livelihoods in drylands and proposes a research framework for dryland ecosystem management and sustainable livelihoods. The framework clarifies the core characteristics of sustainable livelihoods and the principles and strategies of ecosystem management while proposing a research philosophy to guide future enquiry.

Keywords Ecosystem management · Resilience · Sustainable livelihoods · Drylands

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5.1 Introduction

5.1.1 Research Background

Global climate change has exacerbated many of the common problems of drylands, including water resource shortages, soil erosion, desertification, low ecological stability, and habitat fragility. Driven by changing climate and other environmental pressures, the social-ecological systems in drylands may undergo gradual or sudden linear and non-linear changes (Fu et al. 2021). The stability of dryland ecosystems can be maintained by applying the principles of sustainable development, and the key to achieving sustainable development lies in that dryland livelihoods are sustainable. Sustainable livelihoods require a combination of appropriate skills and capabilities, capital, and activities to maintain a sustainable way of life. Maintaining and strengthening the capital and capabilities of a community without destroying natural resources is considered sustainable (Serrat 2017). Because of the fragility of dryland social-ecological systems due to resource constraints, harsh climates, and low levels of economic development, pressures, such as extreme climate events and high impact developments, seriously affect their stability, and they rarely recover quickly. Dryland livelihoods are fragile as they are typically based on crop cultivation or animal husbandry, which are highly dependent on stable environmental conditions and reliable access to a range of resources (Middleton and Sternberg 2013; Moreno-Jimenez et al. 2019). Disturbances such as extreme weather events directly affect the reliable provision of water, food, energy, and ecological security for residents, severely restricting the sustainability of dryland livelihoods (Keesstra et al. 2018; Sibhatu and Qaim 2018). In contrast, low levels of education, limited economic and technological development, and rigid social and cultural traditions limit the development of new industries. These problems work against the development of sustainable livelihoods in drylands.

Livelihoods in drylands are uniquely shaped by climate change, prolonged droughts, variability in resource availability, remoteness, and the prevalence of human mobility and informal economic networks (Asfaw et al. 2019; Robinson et al. 2015). Distinct livelihood vulnerability patterns were identified in developing/transitional and industrialised regions based on the combination of the five indicators: poverty, water stress, soil degradation, natural agro-constraints, and isolation (Sietz et al. 2011). Vulnerable livelihood patterns occur mainly in developing/transitional regions of Africa, Afghanistan, the Middle East, India, and Latin America which contain 84% of all global drylands (Reynolds et al. 2007). In these regions, climate-related shocks such as drought and depletion of natural resources combined with socioeconomic hardships (Asfaw et al. 2019; Robinson et al. 2015) present significant challenges to achieving household- and community-level livelihood resilience (Shackleton and Shackleton 2012). Compared with developing and transitional regions, industrialised arid regions with more potential for livelihood diversification are less vulnerable. These include the Negev region of Israel, Central Spain, Australia, and the Southern Great Plains of the United States. The inhabitants of these regions have more opportunities to participate in non-agricultural economic activities, which helps conserve and reduce dependence on marginal natural resources. Increased urbanisation and associated business developments in these regions are opportunity ("pull") factors enabling livelihood diversification. They offer better employment and business prospects, increased food security, the opportunity to acquire new technical skills and education, and improved physical security and health (Biglari et al. 2019; Mekuyie et al. 2018), greater infrastructure development (e.g., roads, electrification, schools, health institutions), and more growth in local markets (Li et al. 2019). However, rapid urbanisation, such as the development of tourism and gaming in Las Vegas, has presented severe challenges for water resource management (Mauget et al. 2020). Drought and consequent land degradation have also resulted in large economic losses including crop failure and livestock deaths in the Southern Great Plains (Mauget et al. 2020; Smith and Katz 2013). High-risk biophysical issues like these are relevant to the main livelihood activities of the indigenous inhabitants of Australia's rangelands, most of which are in dryland regions (Feng et al. 2020; Foran et al. 2019).

Ecosystem management is recognised as a "wicked problem" without clear-cut solutions because of the inherent complexity of ecosystems and the impossibility of predicting all the consequences of interventions across different spatial, temporal, and administrative scales (DeFries and Nagendra 2017). Wickedness may be worse for the management of dryland ecosystems, which are characterised by vulnerability due to low volumes and high variability of precipitation, in combination with unfavourable temperature, wind, and soil conditions. The productivity of these ecosystems is generally low, whereas the human demand for resources in many regions is usually high. Because of this, land degradation is extensive in dryland regions. However, during the long history of the human presence in dryland regions, considerable experience and indigenous knowledge about the effective management of local ecosystems have been accumulated. These management practices should be studied to determine if they can be effectively applied to other regions. Ecosystems in global drylands are diverse in terms of their type, degradation levels, land use, and human presence, so management regimes and objectives differ as well. "Nature-based solutions" may seem promising, but they may not be well tested in dryland ecosystem management, or applicable to all of the diverse dryland regions that exist (Keesstra et al. 2018).

Under the impact of climate change, unsustainable development and exploitation, dryland regions and countries are generally lagging behind in achieving the UN Sustainable Development Goals (SDGs). The global drylands are challenging but critical for comprehensively achieving the SDGs. The stagnation and decline of arid area development is inconsistent with the UN imperative to "leave no one behind" (United Nations 2015). Therefore, it is important to promote a standardised ecosystem management model and develop related theoretical research methods with the aim to improve the adaptability and resilience of dryland livelihoods. It will require the cooperation of all stakeholders, including academics, the public and policymakers, to eventually realise sustainable economic development and management in dryland regions.

5.1.2 Research Progress

Social and ecological systems are dynamic and mutually influencing one another, thus ecosystem management and the development of sustainable livelihoods are closely interconnected issues. Research on sustainable livelihoods from 1990 to 2020 primarily focused on management, conservation, sustainability, biodiversity, climate change, poverty, resilience, vulnerability, and adaptation. At present, nearly half of the existing research on livelihoods in drylands is related to sustainable livelihoods, including specific case studies on the characteristics of regional livelihood sustainability and how to reliably maintain them. This research generally focuses on rural areas, with families as the most common research subjects. Early research was often narrowly focused on one aspect of sustainable livelihoods and had trouble achieving its goals (van Ginkel et al. 2013). Recent research has generally adopted a more comprehensive perspective for studying dryland social-ecological systems.

Livelihoods are closely related to the living conditions and experiences of people. These investigations generally use structured questionnaire surveys, interviews, and participatory rural evaluation methods which engage research more actively. Most of the research data were collected through household questionnaires, interviews with key information providers, and other related methods. Specific research questions were then answered through data comparison and logistic regression analysis (Antwi-Agyei et al. 2015; Brottem and Brooks 2018; Yobe et al. 2019). The results show that household surveys are highly representative of the attitudes and experiences of the communities as a whole; therefore, decision-makers should consider issues at the scale of households when dealing with livelihood-related policies in drylands (Yobe et al. 2019).

Disturbances such as climate change, population growth, economic development, and new policies cannot be ignored by residents of drylands, who may adapt their livelihoods and respond in other ways that have a range of positive and negative impacts. For example, farmers and herders may engage in deforestation and overgrazing to increase their income. Although measures such as intensifying grazing, dominant pasture cultivation, and supplementary feeding can meet demands for increased production, they are often implemented in ignorance of the carrying capacity of local ecosystems, leading to large-scale and long-term ecological degradation (Brottem and Brooks 2018; Qi et al. 2017; Yobe et al. 2019). If these issues remain unaddressed, herders will lose the capital required to maintain their livelihoods and end up in poverty.

Changes in government policies can have dramatic effects on livelihoods. When local governments imposed strict regulations on floodplain resources in the Okavango Delta of Botswana, local farmers who originally relied on transitional farming methods to cope with flooding and rainfall patterns permanently switched to dryland agriculture. This policy intervention led to the loss of responsive local livelihood strategies and may have caused a decline in the long-term adaptability of residents (Shinn 2016). In contrast, management and technological innovation can also be used to promote and support the sustainability of livelihoods when resources are scarce. Laporiya Village in the semi-arid salt lake region of Rajasthan, India, innovatively used local people as participants and beneficiaries of interventions through community-based shallow groundwater management. These dryland water management measures were adapted to the local system, alleviating water shortages and other threats to livelihoods, and also increasing jobs, and provide a strong example for other dryland areas facing similar challenges (Everard and West 2021). Managers of semi-arid land in Kenya have implemented Sustainable Land Management (SLM) technologies including replanting forages, rain harvesting, soil conservation, and dryland agriculture and forestry compliance technologies to prevent land degradation. SLM technology has contributed to reversing land degradation trends in the local area, improving agricultural production and food security, and subsequently improving the livelihoods of communities in drylands (Mganga et al. 2015). Therefore, it is important to understand the abilities of residents to adapt to environmental changes. Researchers suggest that before regional managers introduce new livelihood interventions, they should accurately assess local capabilities and adopt targeted strategies to truly help families in the region obtain funding (King et al. 2018).

With dryland ecosystems under increasing pressures, currently stable livelihoods may also face various problems. Ecosystem management in drylands is particularly important for maintaining sustainable livelihoods. In recent years, research attention to ecosystem management in arid regions has increased, while the research emphasis varies according to geographical location, climatic conditions, and socioeconomic development status. The main livelihoods in drylands are agriculture and animal husbandry, so their ecosystem management focuses on agricultural irrigation and animal husbandry development. For example, there are a large number of studies on ecosystem management related to agricultural irrigation in Africa, and countries in the Americas have systematically studied agricultural ecosystem management. Some arid regions have also focused on ecological restoration and biodiversity protection. Their research purpose was to restore the natural environment and maintain the stability of regional ecosystems. Some areas consider biodiversity protection and agricultural production to be complementary, such as the Mediterranean region. Dryland ecosystem research in the Mediterranean has taken three directions in recent years, focusing on biodiversity issues, economic crop variety optimisation issues, and ecosystem management and optimisation issues. The unique problems that arise in each region lead to the development of distinct management systems. For example, researchers in Africa have studied critical research issues such as the invasion of local wetlands and grasslands by exotic plants and support ecosystem management by analysing the causes of the invasion and its subsequent direct and indirect effects.

Understanding and predicting changes are the basis of management, which is divided into two steps: literature-based research methods in the early stages and data-based modelling to predict changes in the research subjects in later stages. The research data used at this stage included remote sensing data, temperature and precipitation data, Normalised Difference Vegetation Index (NDVI) imagery, satellite time sequence data, and gridded climatological data. For example, researchers have used spectroscopy and remote sensing technology to detect forage quality that needs to be understood to prescribe effective supplementation in livestock (Serrano et al.

2020). A three-dimensional finite evolution hydrodynamic model has also been used to assess changes in estuarine hydrodynamics (Huang et al. 2020a, b).

At present, application of the relatively rich practical experience in dryland ecosystem management is largely limited within regions. The main reason is that ecosystems in different regions have their own specific characteristics, and the results of ecosystem management vary. For example, continuous hydrological and hydrodynamic feature detection helps to understand the ecology of regional aquatic biota and provide new directions and solutions for dryland river management (Mallen-Cooper and Zampatti 2020). Landsat TM and ETM + surface water time series have been used to determine key factors to ensure landscape connectivity in surface water habitats and to detect out-of-control surface water dynamics to guide irrigation and biodiversity management (Bishop-Taylor et al. 2017). A chemical application framework has been developed for dryland planting field experiments to ensure that managers reduce damage to non-target populations (Umina et al. 2015). Therefore, it is necessary to formulate special management measures based on the distinctive attributes of different study areas and subjects.

5.1.3 Challenges to Livelihoods in Drylands

Livelihoods in drylands face severe challenges. Climate change has caused a continuous expansion of arid areas worldwide through the process of aridification. Aridification puts large areas of land at risk of serious degradation, which is likely to exacerbate poverty in drylands. Human activities, including urban expansion, water and air pollution, and biodiversity loss, also cause aridification, put pressure on water resources, and worsen the effects of drought. Drought restricts the development of agriculture, especially in relatively poor countries with few natural resources, and the livelihoods of residents cannot be guaranteed. Poverty and the social instability caused by it are key problems affecting the sustainable development of drylands.

Following the outbreak of COVID-19 at the end of 2019, many economically undeveloped arid countries in Africa face severe food security problems. Such as Ethiopia, simultaneously affected by civil war and a desert locust plague, has received widespread attention regarding its domestic food security issues. Although many arid countries urgently need to deal with a series of problems caused by climate change, including aridification, worsening droughts, land degradation, resource scarcity, poverty, and the food security issues mentioned above, most lack the capacity for systematic research. An incomplete understanding of the driving factors, structure, and functions of dryland ecosystems restricts the implementation of ecosystem management and development of livelihood capital. In addition, due to the complexity of ecosystems themselves and regional differences, the management level of arid areas is generally low, especially for the less developed arid countries in Africa. The dryland social-ecological system is fragile, and livelihoods within it are specialised and highly dependent on the natural environment and resources, making it extremely vulnerable to disturbance.

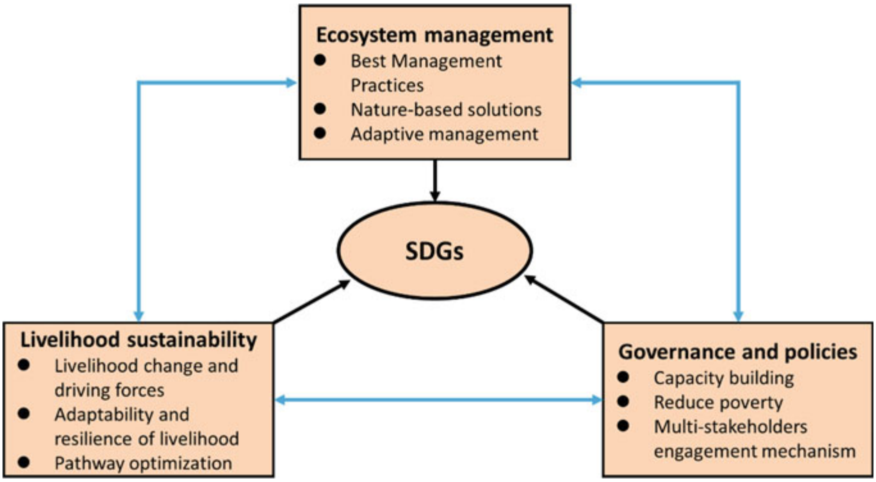

Fig. 5.1 Research framework of ecosystem management and sustainable livelihood in drylands

The main issues facing drylands today are: (1) the lack of an accurate understanding of dryland ecosystems in the context of social development and environmental changes; (2) the lack of capacity of most dryland areas to support the maintenance and improvement of livelihood stability and resilience; and (3) the inadequacy of ecosystem management to adapt to the challenges of supporting livelihoods. To address these problems, we propose the following research framework (Fig. 5.1).

Ecosystem management and livelihoods are part of a complex system that can be thought of as a "panarchy", which is a dynamically organised and structured system arranged across multiple scales of space and time (Allen et al. 2014). They are also directly controlled or indirectly influenced by top-down actions of governments. Hence, it is essential to build the capacity for adaptability and resilience for residents to cope with this complexity and unpredictability, especially for people in less developed countries. In this process, ecosystem management focuses on maintaining sustainable livelihoods. With nature-based solutions as the basic principle, specific measures include case-based construction, and implementing best practice management and adaptive management strategies for disturbance. It is now well recognised that coordination, negotiation, and collaboration among multiple stakeholders are fundamental to effectively implement sustainable ecosystem management and livelihood schemes, and yet the difficulty may vary dramatically across global drylands. Thus, effective mechanisms must be explored to facilitate decision making involving multiple sectors and spanning administrative boundaries (DeFries and Nagendra 2017). As dryland countries lag behind in achieving the SDGs adopted by the United Nations (Sachs et al. 2020, 2021), emphasis should be placed on promoting local development in these places that make ecosystem and livelihood sustainability central to their purpose.

5.2 Building Adaptability and Livelihood Resilience

5.2.1 Ecological Capacity of Livelihood in Drylands

The sustainability of dryland livelihoods depends on a range of variables. It has been proposed that these biophysical and socio-economic variables may be divided into the categories “fast and slow”, with the key dynamics in dryland ecosystems determined by the “slow” variables. Fast variables such as advanced household disposables cannot reflect land degradation or indicate that intervention is needed, while slow variables such as household capital wealth turnover time tend to reflect the key dynamics much more accurately (Reynolds et al. 2007). Therefore, the development of sustainable livelihoods in drylands should be based on the consideration of the “slow” variables.

Dryland social-ecological systems are fragile, having difficulty to bounce back if resource use exceeds their ecological carrying capacity (Reynolds et al. 2007). Extreme weather occurs frequently in drylands due to climate change, causing water shortages, food shortages, and decreased income. The resulting fluctuations may mean that it takes a long time for a family to “rebound” and recover, and returning to their previous socioeconomic status may be impossible (Morecroft et al. 2019; Muricho et al. 2019). Accurate measurement of the carrying capacity for livelihood-related variables is important for maintaining sustainable livelihoods. Current research on dryland social-ecological systems generally relies on models, and the most advanced of these uses dynamic system simulation to determine the limits and behavior of a system (Yu et al. 2021). Due to the diversity of the social-ecological systems in drylands, improving livelihood capacity needs to be developed for specific regions. As farming and animal husbandry are the primary forms of livelihood in drylands, methods of improving livelihood capacity may include the development of drought-resistant crop varieties (Menkir et al. 2020), hydrological ecosystem service management (Porras et al. 2018), increasing access to climate risk information (Satishkumar et al. 2013), and infectious disease control (Wilcox et al. 2019). If cross-scale studies are to be carried out, two obstacles still need to be overcome: the lack of availability and high cost of data with high spatial resolution, and the lack of adequate resources for processing such data (Yu et al. 2021).

5.2.2 Impact of Climate Change on Dryland Livelihoods

Climate change is an important factor affecting residents in drylands. Important climate variables are changing significantly from their historic ranges and cycles. Studying the short-term and long-term impacts of climate change is essential for stabilising dryland livelihoods.

(1) Responses of livelihood-related indicators to climate change in the drylands

Social-ecological systems are formed by long-term interactions between nature and human societies. Changes in external environmental conditions have impacts on the stability and functioning of the system. Global warming and extreme weather events are important climate-related factors affecting ecosystem stability, especially in dryland regions. They have exacerbated the perennial environmental problems of dryland ecosystems, including water scarcity, soil erosion, desertification, and environmental fragility. For example, they have exaggerated the gap between regional water supply and demand in dryland agricultural areas that require intensive artificial irrigation. Dealing with climate change and maintaining sustainable livelihoods have become a severe challenge for residents in drylands.

Global warming is climate change on a relatively long time scale, while the effects of extreme weather are brief but dramatic. To effectively manage the effects of both, it is important to construct a livelihood indicator system specific to drylands, study their response to climate change based on long-term and short-term monitoring data, and interpret the results with the distinction between sensitive and insensitive indicators of climate change in mind. Then, the livelihood-related indicators of stable or easily affected individuals can be highlighted. Specific and locally-important indicators could then be managed in a more targeted and effective way.

(2) Livelihood resilience to climate change in drylands

Because of the fragility of the social-ecological systems in drylands, climate change and human activities have direct and immediate effects on livelihood stability. Understanding the level and quality of household assets provides a clear picture of the household's resource base. Livelihood resilience is measured by assessing financial, physical, natural, social, and human assets. Industrialised arid regions generally have a higher level of resilience than rural area. Dryland inhabitants in developing/transitional regions are highly dependent on climate-sensitive natural resources and ecosystem services, having limited adaptive capacity in terms of the assets that they can mobilise in response compared with those in industrialised regions.

Integrating SDGs with adaptation strategies is an integral part of moving toward a resilient world. Locally, this requires the identification of locally-contextualised entry points to enhance viable livelihood pathways in the drylands. For example, achieving the sustainable use of natural resources is the principal entry point to improve livelihood resilience for inhabitants in less-developed drylands. Knowledge-based entry point interventions, such as water governance targeted at providing solutions and some innovative technologies, are the best options for building resilient livelihood pathways in these areas (Porras et al. 2020; Sietz et al. 2011).

Although livelihoods depend on the ownership or availability of resources, they are ultimately determined by factors such as cultural preferences, education, inheritance, and gender. Industrialised regions with higher levels of human knowledge are considered to have greater adaptive capacity than emerging nations and those in transition (Cohen et al. 2016). Increasing the overall literacy level is a reasonable entry point to reduce livelihood vulnerability in developing/transitional regions by increasing inhabitants' capabilities and access to information. This in turn increases their ability to cope with adversity. The integration of local and traditional livelihood choices with interventions of scientific knowledge is a promising entry point for advancing sustainable dryland livelihoods.

5.2.3 Strategies to Enhance Livelihood Capital

Livelihood capital includes the five aforementioned components: financial, physical, natural, social, and human capital. The core goal of sustainable livelihood strategies is to improve the livelihood capital of drylands. At present, it is urgent to assess, contextualise, and meet the challenges of developing livelihood capital in the drylands. More efforts are needed to assess livelihood vulnerability, analyse influencing factors, and identify the challenges and opportunities for dryland livelihoods caused by climate change and human activities (Muricho et al. 2019). Nevertheless, the vulnerability and resilience of livelihoods may change over time, requiring effective and dynamic policies to support dryland ecosystem self-regulating properties and tackle the degradation of dryland ecosystems.

In recent years, with the deepening of livelihood research, information, cultural (traditional), and institutional capitals have also been regarded as organic components of livelihood capital, attracting extensive attention from scholars in various countries (Odero 2006; Reed et al. 2013). Scholars have conducted research on many aspects of livelihood changes in agricultural households in arid areas, including the spatial differentiation of livelihood (Coetzer et al. 2013; Martin et al. 2016; Wu et al. 2020), livelihood and policy processes (Harihar et al. 2015; Lan et al. 2021; Nepstad et al. 2013), organizational structure change (Wendiro et al. 2019), and livelihood strategies (Adhikari et al. 2021; Ellis and Freeman 2004; Kiptala et al. 2018).

Livelihood strategy refers to the way in which farmers use and combine livelihood assets to pursue goals relating to production activities, investment strategies, and reproductive arrangements. Scoones identified that farmers dynamically used expansionary, intensive, and diversified livelihood strategies (Scoones 1998). When environmental conditions, available livelihood capital, or government policies change dramatically, farmers usually change their livelihood strategies actively or passively to adapt to the new human-land relationship and gain more income, increase their welfare, reduce their vulnerability, and use natural resources more sustainably (Huang et al. 2020a, b).

During the long periods of time that it takes for residents to adapt to dryland ecosystems, they gradually explore strategies for making a living by using the available natural resources. During periods of disturbance, residents of areas with limited natural resources adopt new livelihood strategies, which may include out-migration for work, industrial restructuring, and tourism developing. This strategic adaptability is an important way to achieve sustainable livelihoods in drylands. In addition, the integration of local and traditional livelihood choices with interventions of scientific knowledge could be a good entry point for developing sustainable dryland livelihoods (Bautista et al. 2017; Mauget et al. 2020; Stringer et al. 2017). Livelihood diversification strategies should be adjusted according to the opportunities and resources available depending on the socioeconomic position of each household. All these strategies constitute a resident-centered and rights-based approach that is important in supporting and enhancing the adaptive capabilities of livelihoods in drylands.

5.3 Ecosystem Management and Sustainable Livelihoods

5.3.1 Evaluation and Priorities for Achieving Sustainable Development Goals

Dryland socio-ecological systems are highly complex and vary around the world. Only by truly understanding and evaluating the problems in each region can we formulate targeted ecosystem management strategies to alleviate the challenges they face. Understanding and evaluating the current developmental status of drylands is the first step in ecosystem management. Drylands are different from other regions in the world in that water resource limitations and habitat fragility are critical limiting factors (Dougill et al. 2010; Reynolds et al. 2007). At present, there is no sustainable development evaluation system designed for the special characteristics of drylands. The resulting inability to accurately identify the unique problems and constraints of drylands seriously impedes progress towards sustainable development goals. The lack of a specific dryland evaluation system may be due to a lack of research interest. There are more than 120,000 papers with the theme of "Sustainable development" listed on the Web of Science, while less than 400 can be found with the themes of "Dryland AND Sustainable development".

The UN SDGs provide a template for an evaluation system based on a set of indicators (United Nations 2015). Scholars have conducted a series of studies on indicator selection, standard determination, and correlation analyses between indicators, but research specific to the sustainable development of arid areas is limited. Most research proposes solutions to specific problems. For example, the method of Farmer-Managed Natural Regeneration (FMNR) applied to the restoration and reforestation of cultivated land in drylands to ensure the sustainable production of crops (Weston et al. 2015), and the practice of sustainable pasture management used to solve feed shortages and pasture degradation by investments and economic incentives to improve pasture management and the livelihoods of herders (Louhaichi et al. 2016). Sustainable development strategies can only succeed if all significant factors are considered in the development process (Qi et al. 2017). It is impossible to comprehensively understand the problems faced in the sustainable development of drylands through case studies in individual academic disciplines, making it difficult to formulate appropriate policies. The lack of a systematic sustainable development evaluation system with a good record of successful implementation supported by

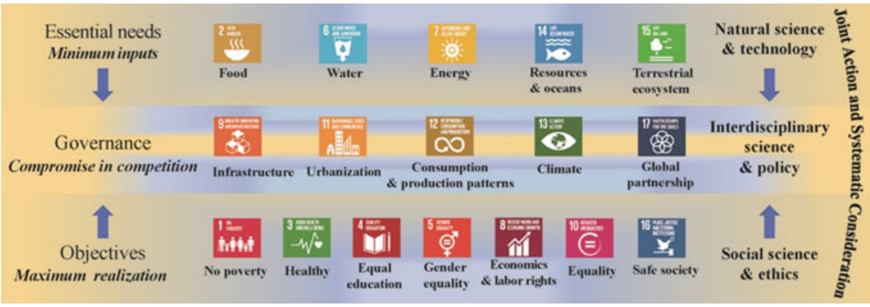

Fig. 5.2 SDG categories: essential needs, governance, and objectives, reprinted from ref. Fu et al. 2019, licenses are CC BY 4.0

case studies is the greatest challenge facing sustainable development research and practice in drylands at present.

With less than 10 years remaining before the planned achievement of sustainable development in 2030, a timeline that will be challenging to achieve for most countries, the efficiency of plans to reach the SDGs must be maximised. To this end, the relationships between sustainable development goals and indicators have become a popular research topic in recent times. Stafford-Smith et al. (2017) proposed the concept of integration through cultivating cross-sectoral links between fields, such as finance, agriculture, energy, and transportation, as well as links between developed and developing countries, in the hopes of promoting sustainable development. Subsequently, Fu et al. (2019) reviewed the complexity and relevance of 17 sustainable development goals. They divided the 17 UN SDGs into three categories, essential needs, governance, and objectives and analysed the interaction between goals (Fig. 5.2). For most regions in the drylands, essential needs are an urgent requirement for residents. Evaluating the development status of drylands provides the basis for proposing priority development goals and optimising future development strategies, which is essential for promoting global sustainable development.

5.3.2 *Principle of Ecosystem Management*

Ecosystems contains many elements that interact and influence each other in complex ways. Owing to differences in spatial, temporal, and administrative scales, the results of interventions are difficult to foresee. It is not surprising that there is still no established, effective management system to deal with complex ecosystem problems. The combination of low productivity and higher dependence on primary production for livelihoods increases the potential for degradation and presents significant additional challenges for ecosystem management in drylands. Although current research on the interaction between livelihoods and ecosystem management is not extensive, it does show that nature-based solutions should be preferred for sustainable livelihood development in drylands. Nature-based solutions are defined by the International Union for Conservation of Nature (IUCN) as "actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham et al. 2019). These actions can effectively and flexibly respond to various social challenges while contributing to human well-being and biodiversity. The concept of nature-based solutions can provide a foundation when formulating sustainable management strategies for arid ecosystems, guiding strategy makers, strengthening the link between regional environmental restoration and socio-economic development goals, and ultimately achieving the goal of sustainable development of livelihoods in drylands (Fu et al. 2021).

Climate warming and water shortages are problems common to all arid regions but there are existing nature-based solutions specifically designed to alleviate them. These solutions ask for reducing the concentration of greenhouse gases in the atmosphere, limiting deforestation, restoring wetland systems, and improving land use. Water shortages are an important limiting factor in drylands, and effective water resource management is critical for realising the social and economic development of drylands. Inner Mongolia, a typical drylands area in northern China, implements strict water regulations based on sustainable development principles. Studies have found that water regulations in Inner Mongolia have promoted industrial transformation, reduced environmentally harmful industries like coal and steel production, and increased the presence of tertiary industries such as tourism. Economic development in Inner Mongolia is no longer strongly dependent on water resources and environmental protection has been achieved at the same time (Liu et al. 2022). This is a prominent example of the remarkable success of nature-based solutions.

Nature-based solutions can act as a framework to reverse the degradation of natural resources by increasing the alignment between conservation and development objectives. They can be implemented alone or integrated with other solutions to societal challenges, depending on the natural and cultural context of a site, and drawing on traditional, local, and scientific knowledge. Interdisciplinary research is the best way to determine the effects of human use on ecosystems and any subsequent changes in ecosystem structure or functions in human social groups. Case studies on nature-based solutions have rapidly accumulated, but few have examined their effectiveness in dryland ecosystem management. In addition, nature-based solutions may lead to the production of ecosystem disservices (having harmful effects on people) (Schaubroeck 2017). Some essential concepts are missing or weakened in nature-based solutions, such as adaptive management/governance, effectiveness, uncertainty, multi-stakeholder participation, and consideration of time scales (Cohen-Shacham et al. 2019). Their relationship with other approaches, such as the ecosystem approach, also requires clarification.

5.3.3 Case Studies and Pathway Exploration

Under different cultural, historical, and social backgrounds, the types of livelihoods and strategies employed in drylands are different, but the variation is based on adaptation to local environmental and social conditions. Local conditions and factors have the potential to threaten the sustainability of dryland livelihoods. The main biophysical constraints include ecosystem degradation, water scarcity and aridification. Social and economic limitations, such as poor access to markets and inputs, weak governance, and lack of information about alternative production technologies also limit the options available to residents in drylands. Sustainable ecological management strategies for individual regions do exist, but there is still no effective system suitable for universal implementation in arid regions around the world. Facing the diversity, variability, and unpredictability of livelihoods in drylands, it is very challenging to present a consistent analysis of different case studies and establish a common theoretical understanding of dryland livelihoods.

Because of the different socioeconomic development characteristics of drylands worldwide, there are significant differences in the development pathways available for restoring and promoting sustainable livelihoods. Agriculture is one of the main livelihoods of dryland residents, in addition to management methods to guide sustainable development of dryland ecosystems, some advanced technologies are also important for the sustainability of livelihoods (Zhao et al. 2014). Regenerative agriculture is used in the Mediterranean. It improves soil quality without affecting the stability of indigenous agricultural ecosystems and thus enhances local capacities to adapt to climate change (Luján Soto et al. 2021). In Africa, in situ rainwater harvesting has helped to increase soil nutrients and crop production (Vohland and Barry 2009). A range of ecological restoration projects have been carried out in Africa, such as the Climate Resilient Agriculture (CRA) projects (Amadu et al. 2021) and Africa's Great Green Wall program (Sacande and Berrahmouni 2016), having had a positive impact on agricultural production and biodiversity conservation. To contain desertification, China has adopted a combination of technologies and management measures. Grass barrier and Ectomycorrhizal (ECM) fungal inoculation techniques are used to promote plant colonisation in China's drylands (Taniguchi et al. 2021), and scientific planting technologies have been developed to take advantage of seasonal cycles through management measures to maintain reasonable ecological water use and groundwater levels (Zeng et al. 2020).

The construction of a cross-scale and multilevel arid region case database would be a great advantage to dryland research and management. A case database based on different function zones (agricultural and pastoral zone, industrial zone, financial zone), aridity levels, income levels, and other characteristics would be useful to analyse particular issues and establish common ground between regions. In this way, researchers will be able to identify broadly applicable rules and lessons that can be used as a reference for sustainable development pathways.

5.4 Summary and Perspectives

Under the influence of global climate change and human activities, drylands are facing severe livelihood sustainability challenges. Global drylands are found in many different countries and regions, and their social-ecological systems are diverse. The key to maintain sustainable livelihoods in drylands is to maintain and strengthen the adaptability and resilience of livelihoods. Ecosystem management based on long-term monitoring and evaluation is necessary for maintaining sustainable livelihoods. The principle of ecosystem management is nature-based solutions, considering the constantly changing external environment and the peculiarities of different arid areas. To construct a cross-scale and multi-level case database and identify best practices will help summarise the general laws that can be referenced for development pathways. Enhancing regional cooperation to achieve holistic development may be beneficial for achieving sustainable development goals.

Maintaining sustainable livelihoods in drylands is an ambitious goal that requires in-depth cooperation among stakeholders. This chapter puts forward the research status and priority research direction of ecosystem management and sustainable livelihood in drylands. In the future, more scholars and managers are expected to implement these research and management strategies to protect the ecological environment and sustainable livelihood in drylands.

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Part II
Dryland SESs in Different Regions

Chapter 6
Socioeconomic and Environmental Changes in Global Drylands

Shilong Piao, Yangjian Zhang, Zaichun Zhu, Xu Lian, Ke Huang, Mingzhu He, Chuang Zhao, and Dan Liu

Abstract Drylands are a pivotal component of Earth's biosphere and provide essential ecosystem services to mankind. Over the past several decades, with rapid population growth, global drylands have been experiencing quick socioeconomic transitioning. Such socioeconomic changes, together with fast climate change, have dramatically altered dryland ecosystem functioning and the quality and quantity of ecosystem services they provide. In fact, complex interactions among climate, vegetation, and humans, involving multiple biophysical, biogeochemical, societal, and economic factors, have all played important roles in shaping the changes in global dryland environment. A comprehensive review of socioeconomic and environmental changes of global drylands and their underlying mechanisms would provide crucial knowledge informing ecosystem management and socio-ecological capacity buildup for a more sustainable future of global drylands. In this chapter, we would begin with summarizing the characteristics of socioeconomic changes in drylands. We then presented and discussed past and future projected changes in dryland ecosystem structure and functioning (e.g., vegetation growth, land cover changes,

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carbon sink, water-use efficiency, resistance/resilience to disturbances) and hydrological cycles (e.g., soil moisture, runoff, and groundwater storage). We also discussed new understandings of mechanisms underlying dryland eco-hydrological changes.

Keywords Dryland ecosystems · Gross primary productivity · Land cover change · Crop yields · Carbon–water coupling · Hydrology · Socioeconomic changes · Vulnerability

6.1 Changes of the Socioeconomic System in Drylands

6.1.1 Human Population and Its Regional Variation

Among the 7.3 billion (in 2015) people in the world, 2.56 billion lived in drylands (Fig. 6.1). A striking characteristic of the global population distribution is strong spatial unevenness at the continental scale (Fig. 6.2). In Sub-Saharan Africa, South Africa, and West Asia, over 75% of the population is distributed in their dryland part. In countries like Mexico, Peru, Bolivia, and Australia, this proportion typically exceeds 50%. Generally, human populations in drylands are more sparsely distributed than those in non-dryland systems at the national scale. As of 2015, more than half of the human population living in drylands was located in African countries (Fig. 6.1). The 10 countries with the most densely populated drylands in the world are located in Africa (Ethiopia, South Africa, Burkina Faso, and Morocco), Asia (India, Iran, China, Turkey, and Afghanistan), and North America (Mexico). It is predicted that the human populations in drylands in Turkey, Iran, China, and Burkina Faso will double by 2050.

6.1.2 Net-Migration in Dryland Regions

Net population migration (immigration minus emigration) is indirectly estimated as the difference between population change and natural population growth. Hotspots of negative net migration from drylands, indicating population loss due to migration, are found in Asia (Pakistan, Syria, Iran, and Uzbekistan), Africa (Nigeria, Somalia, and Morocco) and South America (northeast Brazil) (Fig. 6.3). Migration losses may be driven by environmental factors, such as increased drought frequency, severe heatwaves, or other extreme climate events, and by non-environmental factors such as land degradation or limited technological resources (Neumann et al. 2015). Hotspots of positive net migration into drylands, indicating population growth due to migration, are found in the United States, Zimbabwe, India, and China, with a total net

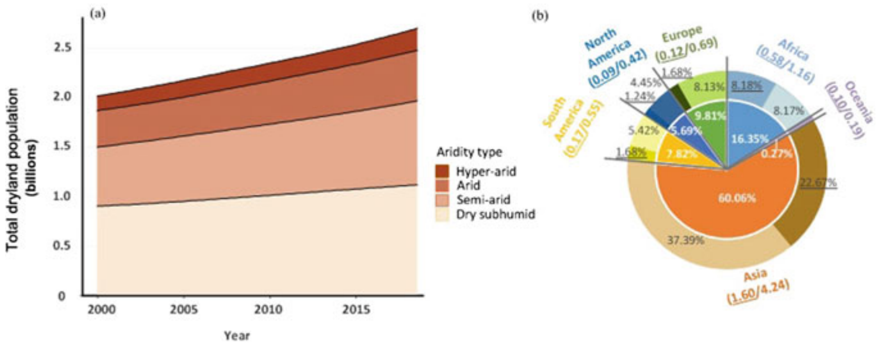

Fig. 6.1 **a** Changes in the human population in dryland regions of different aridity levels, over 2000–2018. **b** Percentages of continental populations living in drylands in 2015. Numbers below each continent label represent the population in billions. Numbers in the inner pie chart indicate the percentage of the global total, and those over the outer ring indicate the proportion of the continental population living in drylands and non-drylands. Underlined values represent drylands

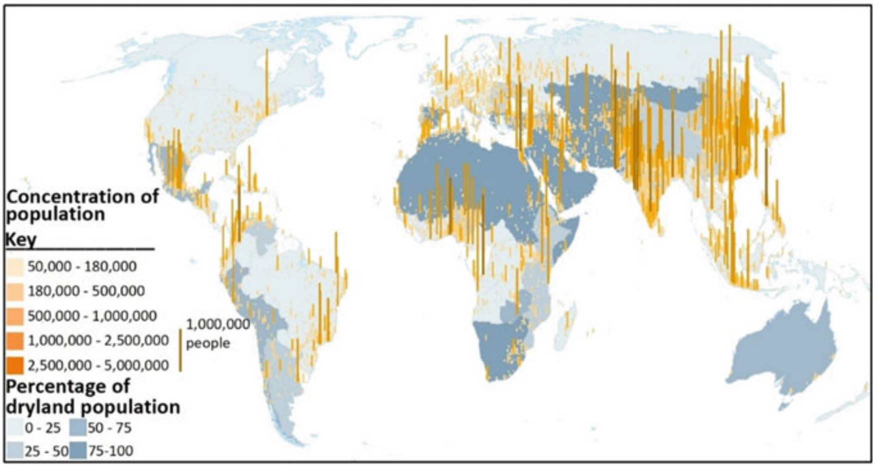

Fig. 6.2 Global distribution of human population in 2015. Areas with <50,000 individuals per cell (10 × 10 km) are not shown. Bar heights reflect population size

gain of over 100 million people. Countries with positive net migration generally have greater and more varied employment opportunities, higher incomes, more developed technology, and stronger government policies aimed at sustainable land development.

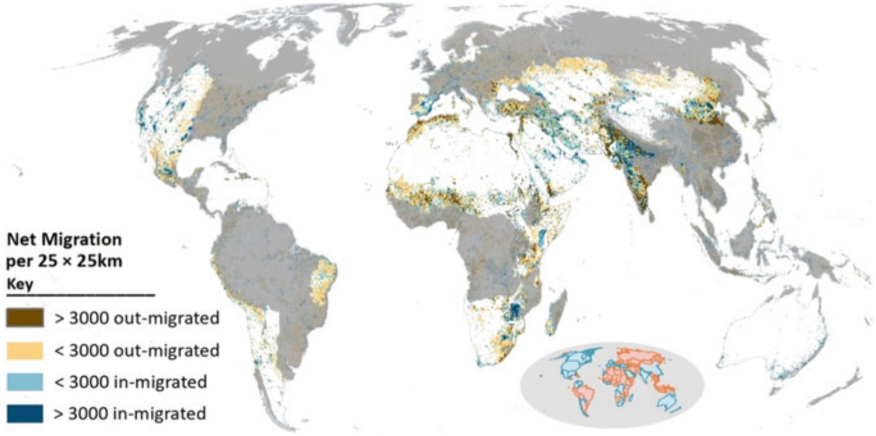

Fig. 6.3 Global pattern of net migration during 2010–2015. Dark gray shading indicates humid regions. The inset panel shows countries with net emigration in pink, and countries with net immigration in blue. Countries without drylands are not shown

6.1.3 *Projected Population Growth*

The Shared Socioeconomic Pathways database suggests that the global population in drylands will peak by the 2060s and then undergo a slight decline, leading to an overall increase of 25% by the end of the century (Fig. 6.4). During 2010–2050, the human population in drylands will increase by an estimated 1.1 billion people; 47% of this increase will be contributed by Africa (0.52 billion) and 34% (0.36 billion) by Asia, with smaller increases projected for North America (0.09), South America (0.08), Europe (0.02) and Oceania (0.03). Predicted population increases show different patterns between urban and rural dryland areas. Populations in urban centers are projected to increase by the end of this century, with a predicted tripling over the African continent. This reflects the profound impact of urbanization on population growth. By contrast, the rural dryland population is expected to shrink over this time period in all regions excluding Africa.

6.1.4 *Economic Development in Drylands*

Gross domestic product (GDP) is a conventional proxy for economic development. According to global gridded GDP maps (Kummu et al. 2018), dryland countries contributed >25% of the total global GDP over 1990–2015. The average GDP in dryland countries is well below the global average and is distributed unevenly across continents (Fig. 6.5). Asia, Europe, and North America have shown the most rapid GDP growth over the last three decades, with Asia having the most spatially homogenous growth distribution between dryland and non-dryland areas during 1990–2015.

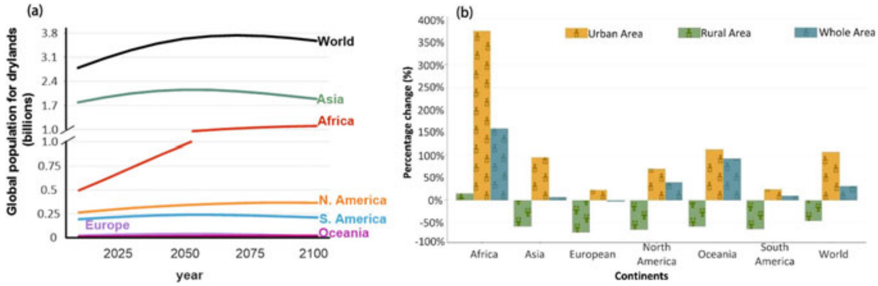

Fig. 6.4 **a** Projected population growth by region from 2010–2100 in current-day dryland areas. **b** Projected changes in population growth by region by 2100, as compared to 2010, for current-day dryland areas. Data were obtained from the Global 1-km Downscaled Population Base Year and Projection Grids provided by the Shared Socioeconomic Pathways database, version 1.01 (2000 – 2100)

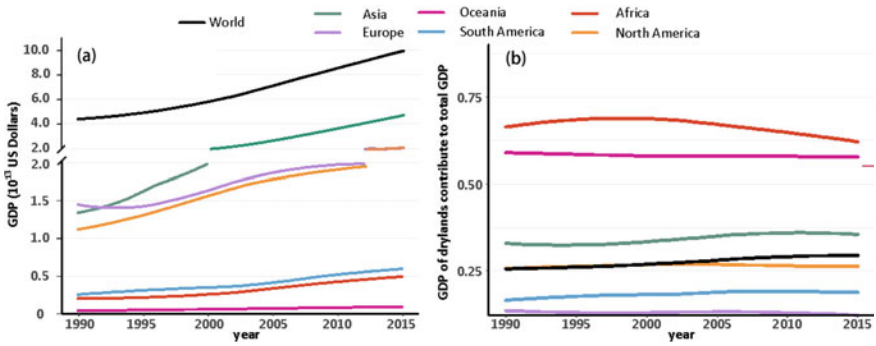

Fig. 6.5 **a** GDP and **b** the contribution of dryland regions to the total GDP for each continent during 1990–2015

In Africa, dryland countries showed less growth than humid ones. Among Asian and African countries, dryland regions typically contribute >50% of the total national GDP (Fig. 6.6). Dryland areas that are highly economically developed are mainly distributed in western Asia, Eastern Europe, East China, and the United States. When overlain with patterns in net migration, it is clear that dryland regions with higher GDP are more strongly associated with net migration gains.

Satellite-derived nighttime light intensity is a robust proxy for human activity (Fig. 6.7). When assessed over decadal time periods, nighttime light intensity illustrates the intensification of human industrial activity, changes in the mosaic of human settlements, and wildfire events. Combined use of population density and nighttime light intensity robustly reflects the relative level of economic development and human wellbeing in specific areas. Although persistent cloud cover can obscure urban centers in some areas like the Amazon and Congo, dryland urban centers tend to have more reliable light measures due to clear nighttime skies. Regions denoted in light grey in Fig. 6.7 represent older established urban centers, whereas those in cyan, yellow,

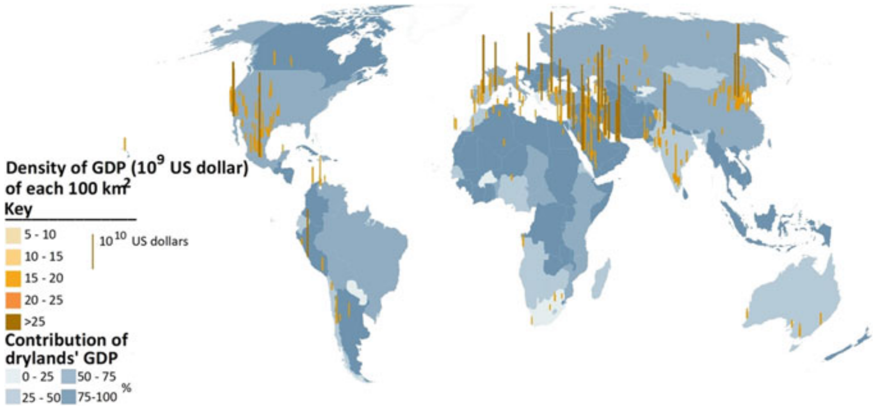

Fig. 6.6 GDP density in global drylands as of 2015. Cells (10 × 10 km) with a GDP <5 billion are not shown. Bar heights reflect the GDP for each cell. The colors indicate the relative contributions to GDP. Countries that do not have dryland regions are not shown

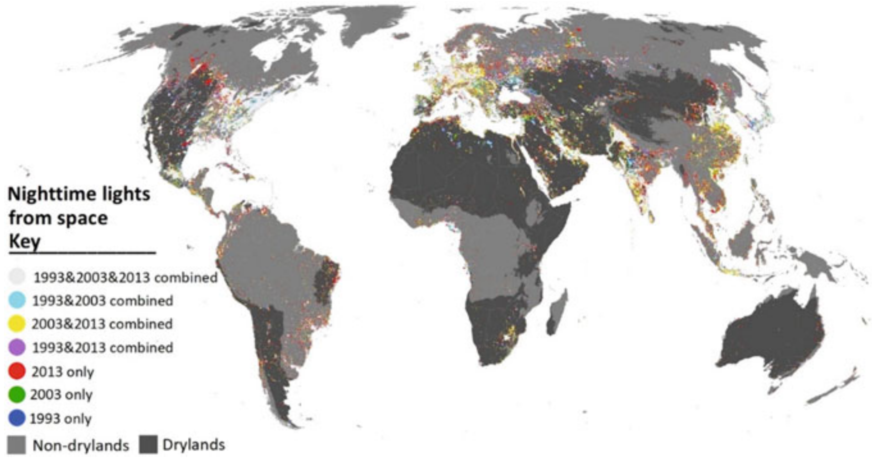

Fig. 6.7 Global patterns of nighttime light intensity (DMSP-OLS) as observed from space. Colors are representative of three annual cloud-free composites: 1993 (blue), 2003 (green), and 2013 (red). The remaining colors indicate changes in nighttime light intensity within this 10-year period

and magenta show urban growth that has occurred during the twenty-first century. Prior to 2003, there were far fewer large urban centers in drylands compared to non-drylands, particularly in Africa, Asia, and South America. More diffuse light over these dryland areas reflects their low levels of infrastructure. Differences in urban growth between dryland and non-dryland areas have been less pronounced in North America.

6.2 Changes in Dryland Ecosystems

6.2.1 Vegetation Greenness

Dryland ecosystems, primarily comprised of savannas, grasslands, and shrublands, are characterized by long-term water stress and high sensitivity to climate fluctuations. Remote sensing techniques provide valuable and continuous data on dryland vegetation at the global scale. However, satellite orbit drift and sensor degradation and replacement have led to uncertainties in the data time series (Jiang et al. 2017; Zhang et al. 2017). We used the most recent versions of multiple remote sensing datasets to systematically assess trends in global dryland vegetation for 1982–2019.

For the assessed period, all remotely sensed vegetation indices indicated significant vegetation growth in arid areas. Leaf area index (LAI) values provided by the Global Inventory Modeling and Mapping Studies (GIMMS) and GLOBMAP both suggested a significant increase in the annual mean LAI in global drylands (0.013 m^2m^{-2} $decade^{-1}$, $p < 0.01$, and 0.015 m^2m^{-2} $decade^{-1}$, $p < 0.01$, respectively; Fig. 6.8). This trend was further supported by the GIMMS normalized difference vegetation index (NDVI) 3 g, which indicated a significant increase in the greenness of global dryland vegetation (0.003 $decade^{-1}$, $p < 0.01$; Fig. 6.8). Although several studies reported a slowdown of the greening of dryland vegetation after 2000 (Gonsamo et al. 2021; Yuan et al. 2019), recent Moderate Resolution Imaging Spectroradiometer (MODIS) vegetation indices consistently suggest that this trend is continuing and evolving (Fig. 6.8). Notably, MODIS LAI suggests a larger increase in LAI over the period 2003–2019 (0.036 m^2m^{-2} $decade^{-1}$, $p < 0.01$; Fig. 6.8) than the LAI estimates provided by GLOBMAP and GIMMS.

Greening trends are not uniform across drylands, as shown in the spatial pattern of annual mean LAI in dryland vegetation over the past three decades (Fig. 6.9). The two long-term LAI datasets showed consistent greening and browning trends (Fig. 6.9), mainly in North/Central America, western India, Inner Mongolia, southern Sahara, South Africa, and eastern Australia. By contrast, browning trends were clearly shown in west Asia, central and southern South America, and northwestern Australia. Globally, greening trends increase along with the aridity index (Fig. 6.9). Based on GIMMS LAI3g data, the changes in annual mean LAI across hyper-arid (aridity index < 0.05), arid (0.05 $\leq$ aridity index < 0.2), semiarid (0. 2 $\leq$ aridity index < 0.5), and sub-humid arid (0.5 $\leq$ aridity index < 0.65) regions are 0.004 m^2m^{-2} $decade^{-1}$, 0.005 m^2m^{-2} $decade^{-1}$, 0.014 m^2m^{-2} $decade^{-1}$, and 0.024 m^2m^{-2} $decade^{-1}$, respectively. These trends were also consistent with GLOBMAP estimates (Fig. 6.9).

Driving mechanisms of dryland vegetation changes

Dryland greening, as observed by satellites, is an integrated response vegetation to environmental change. Understanding and quantifying the contribution of individual environmental factors to dryland vegetation growth is challenging yet critical research highlights (Lian et al. 2021; Piao et al. 2020; Zhu et al. 2016). Among many

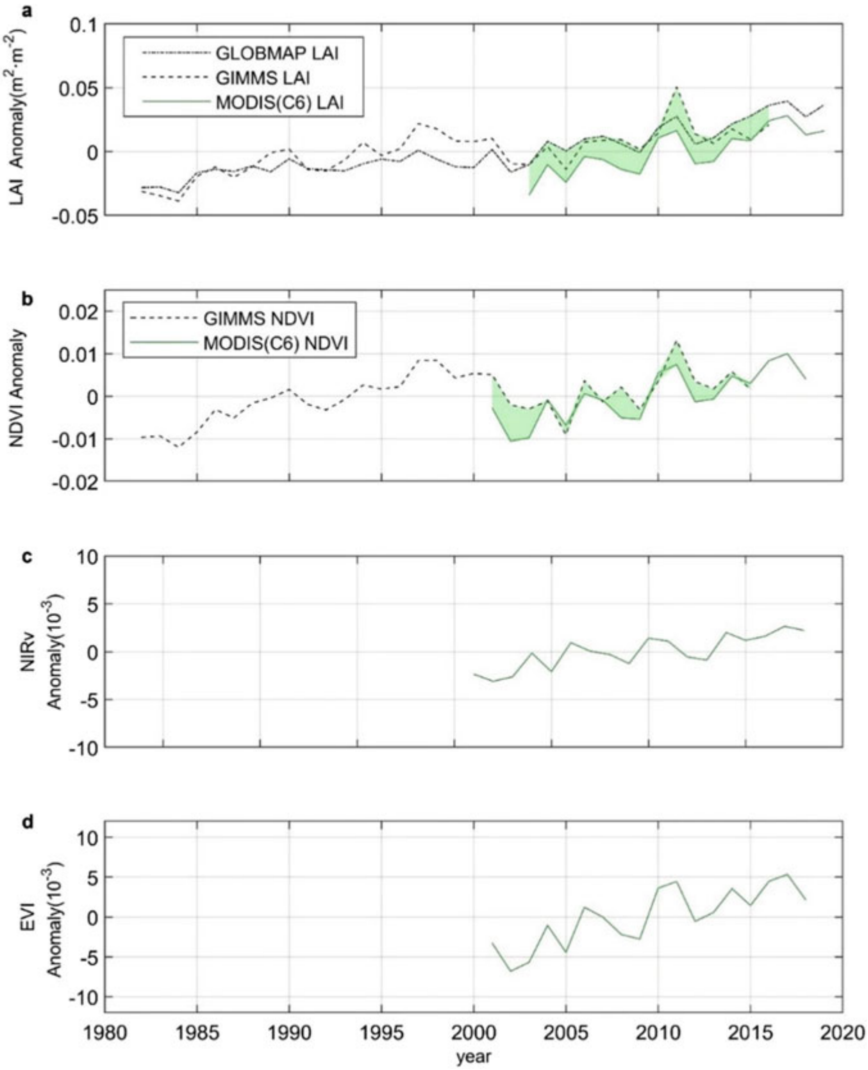

Fig. 6.8 Changes in satellite-derived vegetation indices and solar-induced fluorescence in global drylands. **a** Leaf area index (LAI) from three products: GIMMS LAI3g (Zhu et al. 2013), GLOBMAP LAI (Liu et al. 2012), and MODIS LAI (Myneni et al. 2002; Yan et al. 2016). **b** Normalized difference vegetation index (NDVI) from GIMMS NDVI3g (Pinzon and Tucker 2014) and MODIS C6 (Huete et al. 2002). **c** Near-infrared reflectance of terrestrial vegetation (NIRv) (Badgley et al. 2017). **d** Enhanced vegetation index (EVI) from MODIS C6 (Huete et al. 2002)

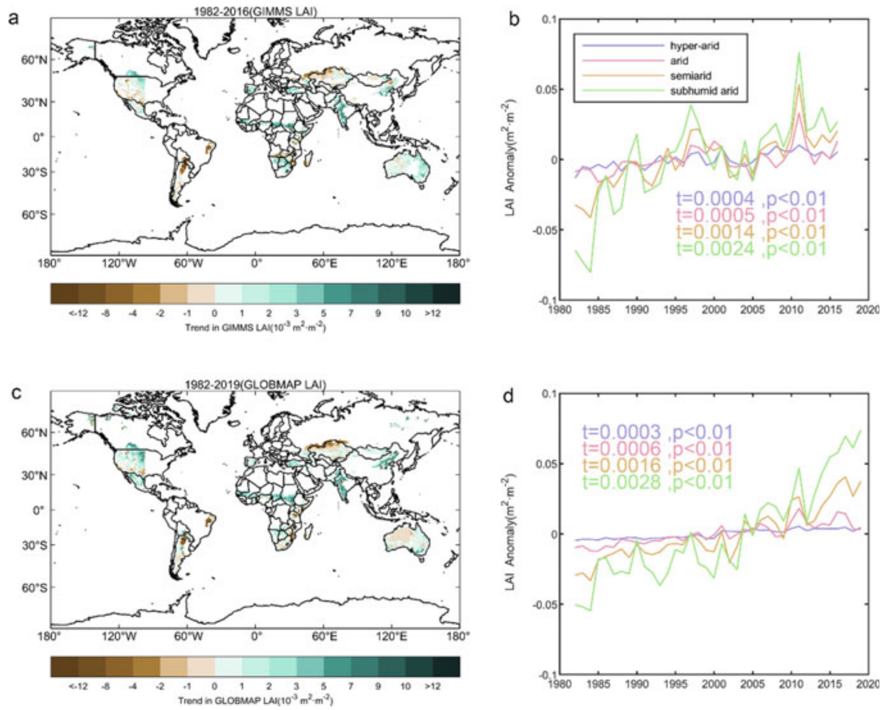

Fig. 6.9 Spatial pattern of changes in annual mean leaf area index (LAI) in global drylands during 1982–2019 from **a** GIMMS LAI3g (Zhu et al. 2013) (1982–2016), **b** GIMMS LAI3g over four dryland categories (hyper-arid, arid, semiarid, sub-humid arid), **c** GLOBMAP LAI (Liu et al. 2012), and **d** GLOBMAP LAI over four dryland categories

environmental changes, increasing atmospheric CO_2 concentration, climate change, nitrogen deposition, and land cover change have been widely assessed and identified as the major driving factors of dryland greening (Piao et al. 2020; Zhu et al. 2016).

We used an ensemble of eight state-of-the-art ecosystem models (CLM5, ISAM, ISBA-CTRIP, JULES-ES-1.0, LPJ-GUESS, ORCHIDEEv3, DLEM, and ORCHIDEE) to quantify the contributions of elevated atmospheric CO_2 concentration, climate change, and land cover change to the satellite-observed trends in the annual mean LAI of global dryland vegetation during 1982–2016 (Fig. 6.10). Factorial simulations included no forcing change (S0); varying CO_2 only (S1); varying CO_2 and climate (S2); and varying CO_2, climate, and land use (S3). Simulation S3 forced by all environmental factors well captured the interannual variation in annual mean LAI (Fig. 6.10). The correlation coefficients between the model-simulated LAI, GIMMS LAI, and GLOBMAP LAI time series were 0.83 ($p < 0.01$) and 0.85 ($p < 0.01$), respectively. GIMMS LAI3g and GLOBMAP LAI both suggested a significant increase in annual mean LAI ($0.013^{0.014, GLOBMAP}_{0.012, GIMMS}$ m^2m^{-2} $decade^{-1}$, $p < 0.01$) during 1982–2016. Ecosystem models to some extent reproduced the observed greening trends but with notable uncertainties ($0.018^{0.030, JULES-ES-1.0}_{-0.001, ISAM}$ m^2m^{-2}

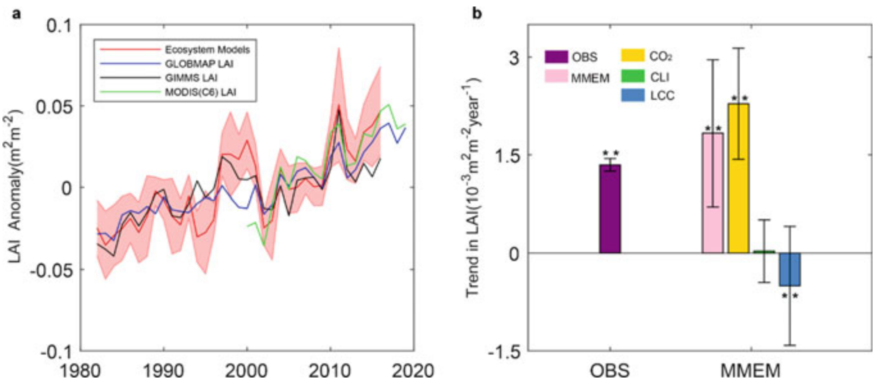

Fig. 6.10 Changes in the annual mean LAI of global dryland vegetation and its driving factors. **a** GIMMS LAI3g, GLOBMAP LAI, and MODIS LAI. **b** The relative contribution of increasing atmospheric CO_2 concentration (CO_2), climate change (CLI), and land use and land cover change (LCC) to the LAI trends simulated by the ensemble ecosystem models

decade^{-1}, $p < 0.01$). Overall, consistency between the modeled simulations and remotely sensed data provides confidence for using ecosystem models in attribution analyses.

Factorial simulations of ecosystem models provide an effective means to quantify the major drivers of global dryland vegetation change (Piao et al. 2020; Zhu et al. 2016). The contributions of atmospheric CO_2 concentration, climate change, and land cover change to changes in LAI were quantified using S1–S0, S2–1, and S3–S1, respectively.

Rising atmospheric CO_2 concentration. The fertilization effect of elevated atmospheric CO_2 has been quantified in open-top-chamber experiments (Drake et al. 1989; Leadley and Drake 1993) and free-air CO_2 enrichment (FACE) experiments (Norby et al. 2010; Norby and Zak 2011). Increased atmospheric CO_2 contributed $0.023^{0.033,CLM5.0}_{0.001,ISBA-CTRIP}$ m^{2}m^{-2} decade^{-1} ($p < 0.01$) to the global LAI (Fig. 6.10). Relative to other vegetation types, CO_2 fertilization effects are more prominent in drylands, where elevated CO_2 concentration alleviates water stress by reducing stomatal apertures and increasing the water use efficiency (WUE) of plants (Donohue et al. 2013; Lian et al. 2021). The simulations suggested that the effects of CO_2 fertilization on global vegetation growth have been uniformly positive over the past three decades (Fig. 6.11). However, this trend may decline as other environmental factors start to limit plant physiology (Hovenden et al. 2019; Norby et al. 2010; Reich et al. 2014; Terrer et al. 2016).

Climate change. Climate change contributed $0.0003^{0.008,ORCHIDEE}_{-0.007,CLM5.0}$ m^{2}m^{-2} decade^{-1} ($p = 0.92$) to the global dryland LAI (Fig. 6.10). In contrast to the uniform effect of CO_2 fertilization, the effect of climate change on global dryland vegetation is notably heterogeneous (Fig. 6.11). Positive effects dominated vegetation growth in >55% of the vegetated land in the northern and southern high latitudes (north of 50°N

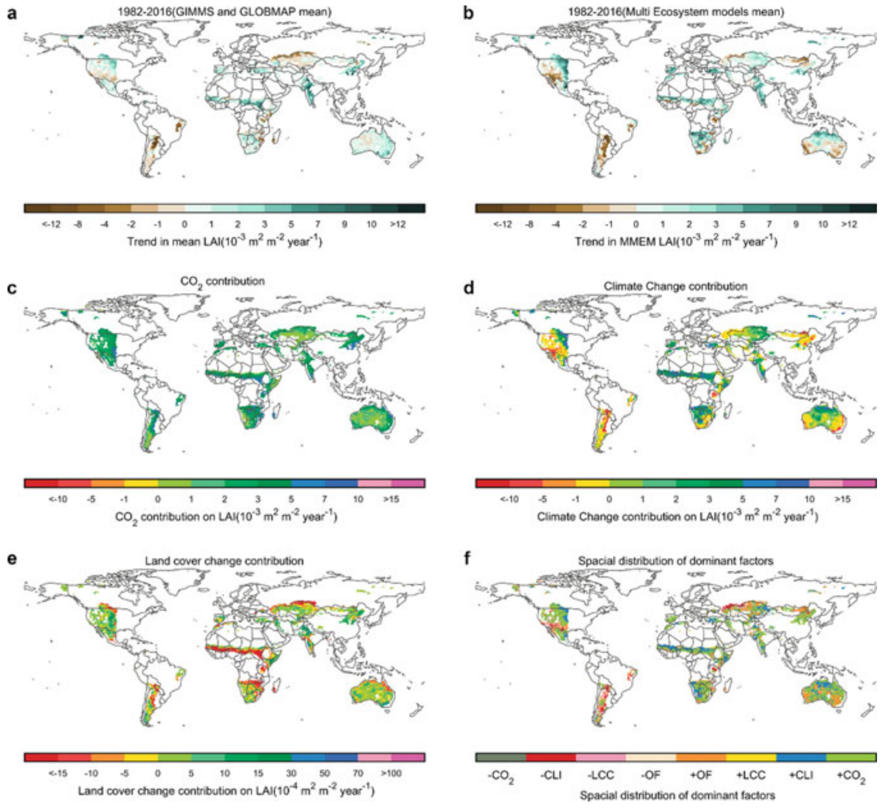

Fig. 6.11 Attribution analysis of trends in the annual mean LAI across global drylands. **a** Satellite-observed LAI trends. **b** Model-simulated LAI trends. **c** CO_2 fertilization effects. **d** Effects of land cover change. **e** Effects of climate change. **f** The dominant factors that drive annual mean LAI. Other factors (OF) are defined as the fraction of the observed LAI trends not accounted for by modeled factors. The prefix '+ ' indicates a positive effect of the corresponding driver on LAI trends, and the prefix '−' indicates a negative effect

and south of 50°S, respectively) and the Tibetan Plateau. This is due to increased air temperature extending the growing season and enhancing photosynthesis in these regions (Keenan and Riley 2018; Xu et al. 2013). Global-scale precipitation redistribution, including the amount, seasonality, and frequency, is also likely to be an essential driver of the heterogeneity of climate change effects (Ukkola et al. 2021; Zhu et al. 2017). Overall, there were low net effects due to climate change, caused by positive effects offsetting negative ones.

Land cover change. As human societies have highly developed, natural vegetation has been cleared for agriculture, but large swaths of croplands have since been abandoned, and natural vegetation has regrown in these areas (Foley et al. 2005; Song et al. 2018). Current ecosystem models partially represent these biogeographical processes, which strongly affect regional vegetation greenness

(Hansen et al. 2013; Piao et al. 2018; Zhu et al. 2016). The ensemble ecosystem models suggested that land cover change contributed $-0.005^{0.005, ORCHIDEEv3}_{-0.018, LPJ-GUESS}$ m^2m^{-2} $decade^{-1}$ ($p < 0.01$) to the global LAI. Positive effects of land cover change were most prominent in regions with extensive agricultural activity, whereas areas affected by negative change tended to be clustered in western Asia, southern Sahara, and South Africa. Notably, the conversion of agricultural land to forest (reforestation) and large plantation forest programs (afforestation) have also greatly contributed to the greening of vegetation in these regions (Chen et al. 2019; Song et al. 2018).

Other factors. The unexplained portion of the satellite-derived LAI trends was determined by subtracting the satellite-derived trends from the trends simulated by the TRENDY models considering all driving factors (Fig. 6.11). We consider this unexplained variation to have been driven by other factors (OF). OF effects are likely best summarized into three categories: uncertainties in satellite observations, misrepresentation of processes in the ecosystem models, and missed processes in the ecosystem models. Interestingly, OF effects are not widely represented in Fig. 6.11. This indicates that current ecosystem models can reasonably reproduce satellite-derived LAI trends. Nevertheless, ecosystem models still require improvement in terms of their ability to represent processes associated with agricultural activities, forest aging, other regionally important ecosystems such as wetlands and peatlands, and disturbances (Chazdon et al. 2016; Kantzas et al. 2015; Pan et al. 2011; Zhou et al. 2015).

6.2.2 Land Cover Change

Drylands occupy approximately 42% of the total land area globally, with predominant types including grasslands, shrublands, croplands, and barren lands (Fig. 6.12). In a MODIS-derived land cover product, grasslands comprise 40.5% of global drylands, followed by barren lands (33.2%), shrublands (15.5%), and croplands (10.1%). These natural and semi-natural lands provide invaluable ecosystem services for human populations. Globally, grasslands are mainly distributed in western America, western Asia, northern Mongolia, Inner Mongolia, southern Sahara, eastern and southern Africa, and northern and eastern Australia. Croplands in drylands are concentrated in the Northern hemisphere, i.e., northwestern America and western India. Shrublands are concentrated in the Southern hemisphere, including central Australia, southern Arica, and southern South America.

Dryland land cover types are vulnerable to environmental change and anthropogenic activity, and dryland land cover has changed significantly due to these factors (Song et al. 2018). We quantified land cover change in drylands during 2001–2019 using an annual MODIS-derived land cover product (Fig. 6.13). The most significant change over this period was the interconversion of grassland and shrubland. Approximately 9.7×10^5 km^2 of grassland was converted to shrubland, and 9.3×10^5 km^2 of shrubland was converted to grassland during this period. Approximately 2.7 ×

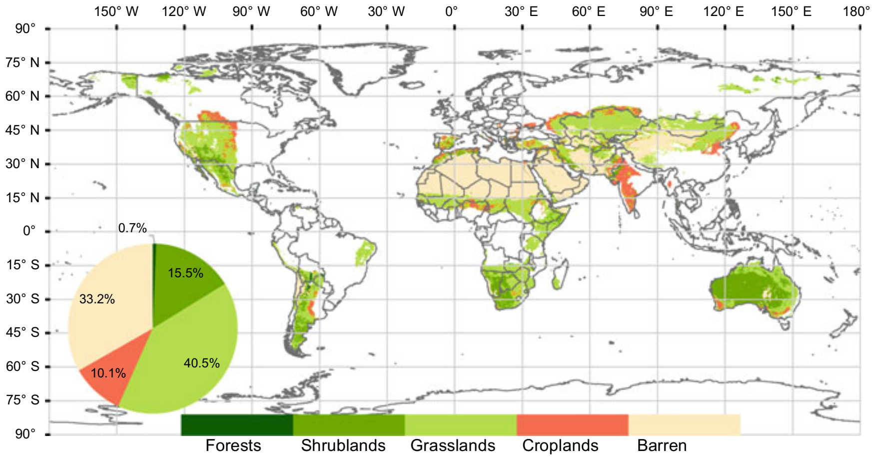

Fig. 6.12 Global pattern and proportion of land cover in drylands during 2019. The pie chart represents the proportion of each land cover type. The most common land cover types in arid areas are grasslands and barren lands, followed by shrublands and croplands

10^5 km^2 of vegetation dryland was converted to barren land, but 7.2×10^5 km^2 of barren land was converted to a vegetated land cover type, primarily grassland (4.2×10^5 km^2) and shrubland (2.9×10^5 km^2). Overall, global drylands now have more vegetated land cover than they were in 2000.

Spatial patterns of land cover change are strongly heterogeneous (Fig. 6.14). Conversion from other land cover types to cropland was concentrated in the northern hemisphere, mostly in northeastern America, western India, and Inner Mongolia. Transitions among natural vegetation types was more prevalent in drylands south of 15°S. A notable fraction of barren lands has been converted to grasslands in northwestern China. The same land cover transition was also observed in the southern belt of the Sahara. It appears that environmental changes, including increased CO_2 concentration and altered precipitation, are promoting vegetation encroachment in these regions (Li et al. 2018; Lian et al. 2021; Ukkola et al. 2021).

Land cover change in drylands at the global scale influences profoundly climate feedback systems and human societies (Lian et al. 2021). In the context of global environmental changes, quantifying land cover change in arid areas and elucidating the driving mechanisms are vital for both current understanding and future predictions of ecosystem services (Zelnik et al. 2013). Future efforts to formulate land management strategies and polices in arid areas could benefit from quantifying the interactions between natural environmental change and human activities under climate change (Burrell et al. 2020; Tian et al. 2019).

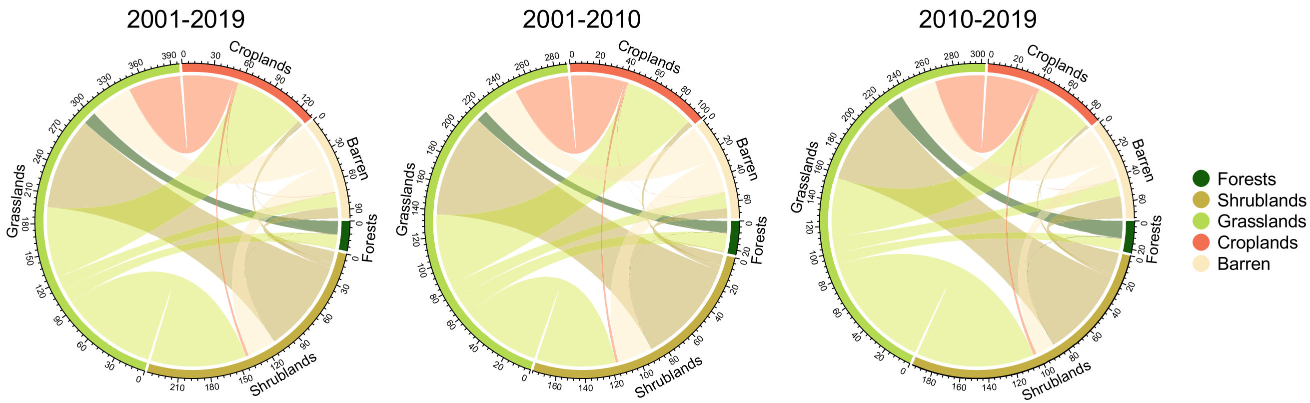

Fig. 6.13 Land cover changes over three periods; 2001–2019, 2001–2010, and 2010–2019. Numbers indicate the amount of converted lands (10^4 km^2). Colors indicate the total change in land cover type. The change in land cover type is represented by the position of the chord and ring; identical colors represent a loss of that type, and different colors represent a gain

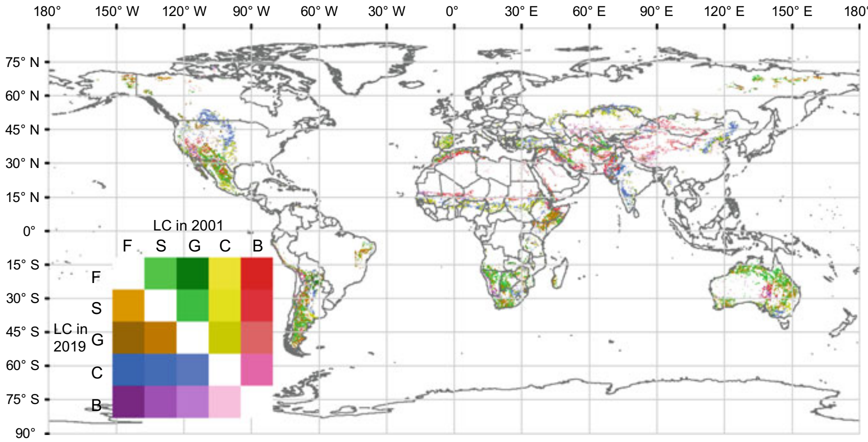

Fig. 6.14 Global pattern of land cover change during 2001–2019. F, S, G, C, and B indicate forests, shrublands, grasslands, croplands, and barren land, respectively

6.3 Changes in Ecosystem Functions in Drylands

6.3.1 Ecosystem Productivity

Gross primary productivity (GPP) is the amount of carbon fixed by vegetation through photosynthesis, which is a critical component of the terrestrial carbon cycle. Quantification of GPP from regional to global scales is important for understanding the feedbacks of terrestrial ecosystem to climate change (Le Quéré et al. 2009; Piao et al. 2009; Sitch et al. 2015). In drylands where long-term and continuous field observations are scarce, remote sensing is a primary approach for monitoring vegetation functional dynamics at broad scales (Smith et al. 2019). Another alternative to observations is dynamic global vegetation models (DGVMs) which simulate major biochemical processes in terrestrial ecosystems. Such models have been widely used to study spatiotemporal variability in global and regional carbon cycles and its driving processes.

We used DGVMs to estimate annual mean GPP in drylands during 1980–2019. An annual average GPP is about 29 $\pm$ 5.0 Pg C yr^{-1}, with larger values distributed in transition zones between dry and wet regions (Fig. 6.15). Dryland ecosystems in Australia, USA, and Brazil had the highest annual mean GPP values, which together comprised 27% of the global total GPP. Dryland ecosystems comprise 91% of the total land area in Australia. Consistently, satellite-observed solar-induced fluorescence (SIF), as a robust proxy for GPP, also showed a similar spatial pattern (Fig. 6.15) of enhanced productivity.

According to the DGVMs, dryland GPP increased significantly at an average rate of 0.10 Pg C yr^{-2} during 1980–2018 (Fig. 6.16), partly contributing to the overall growth in global GPP over the same period (Campbell et al. 2017). SIF showed

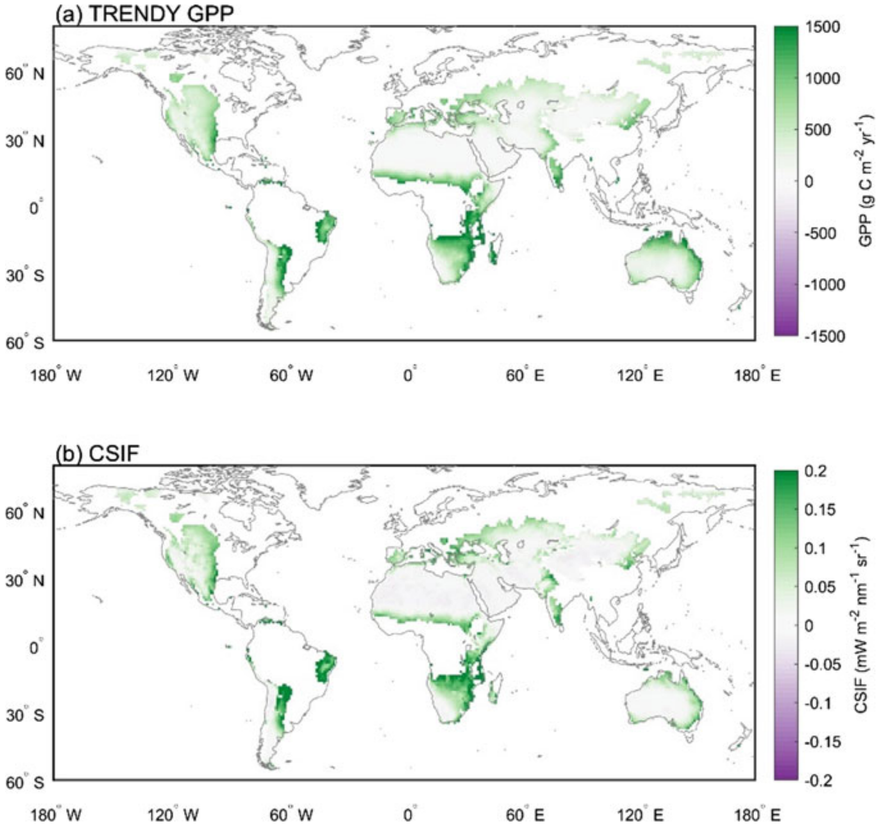

Fig. 6.15 Spatial distributions of gross primary productivity (GPP) and solar-induced fluorescence (SIF) across drylands during 1980–2018. **a** Spatial patterns of annual mean GPP estimated from multiple dynamic global vegetation models (DGVMs) provided by the TRENDY project. **b** Spatial patterns of annual mean SIF, obtained from a contiguous solar-induced chlorophyll fluorescence (CSIF) product (Zhang et al. 2018), during 2000–2018

similar interannual variations, with a significantly increasing trend in dryland ecosystems during 1980–2018 (Fig. 6.16). Increased CO_2 concentration, climate change, and land cover change are likely to drive this pattern. Increased atmospheric CO_2 directly stimulates photosynthesis, and indirectly by reducing stomatal conductance and elevating water use efficiency (Lian et al. 2021; Piao et al. 2020). Factorial simulations from multiple DGVMs suggest that CO_2 fertilization accounted for 91 $\pm$ 20% of the GPP trend in 1980–2018 (Fig. 6.16). The fertilization effect of CO_2 is particularly prevalent in drylands, where positive enhancement of GPP was seen over 94% of drylands (Fig. 6.17). This effect was greatest in Australia, followed by the United States and China, countries with the greatest area of drylands globally.

In contrast to the straightforward positive influence of CO_2 fertilization, climate change contributed both positively and negatively to dryland GPP trends from 1980 to

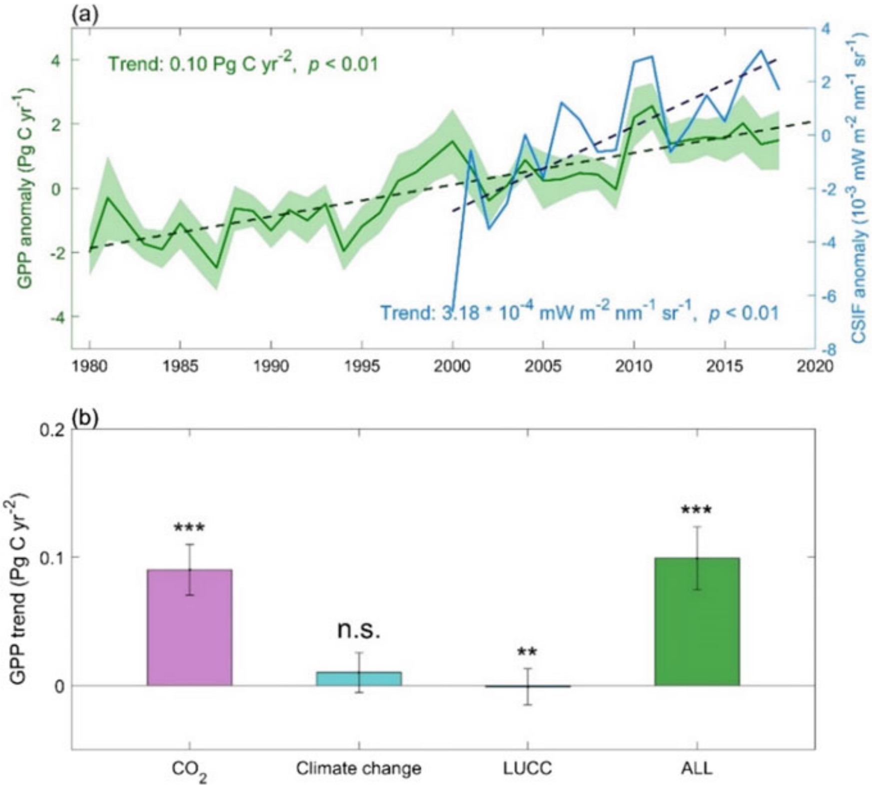

Fig. 6.16 Trends in GPP and SIF, and attribution analysis of GPP trends in drylands. **a** Temporal variation in annual mean GPP from multiple DGVMs provided by the TRENDY project (1980–2018), and the annual mean SIF obtained from a CSIF product (2000–2018). **b** Attribution analysis of the DGVM-estimated GPP trend (1980–2018): effects of CO_2 fertilization, climate change, and land cover and use change, and the overall GPP trend. Error bars represent standard deviations across multiple DGVMs. Stars represent significance levels for the estimated GPP trend. ***$p <$ 0.01; **$p < 0.05$; *$p < 0.1$; n.s., $p > 0.1$

2018 (Fig. 6.17). Climate change had a significant effect for 32% of the total dryland area globally, with larger positive effects on the GPP trend seen in the United States, Canada, and Sudan, and larger negative effects observed in Australia and Brazil. Climate change explained approximately 10 ± 16% of the total GPP trend, but this influence was not significant across global dryland ecosystems due to the balance of positive and negative effects (Fig. 6.16). Despite large model spread, land cover and use changes had a significant, albeit small, negative effect (−1.1 ± 14%), on the GPP trend (Fig. 6.16). Spatially, positive effects of land cover and use changes were found in Sudan, India, and China, whereas strong negative effects were found in Russia, Australia, and the United States (Fig. 6.17). The expansion of forest and cropland cover in China and India (Chen et al. 2019) has positively influenced GPP.

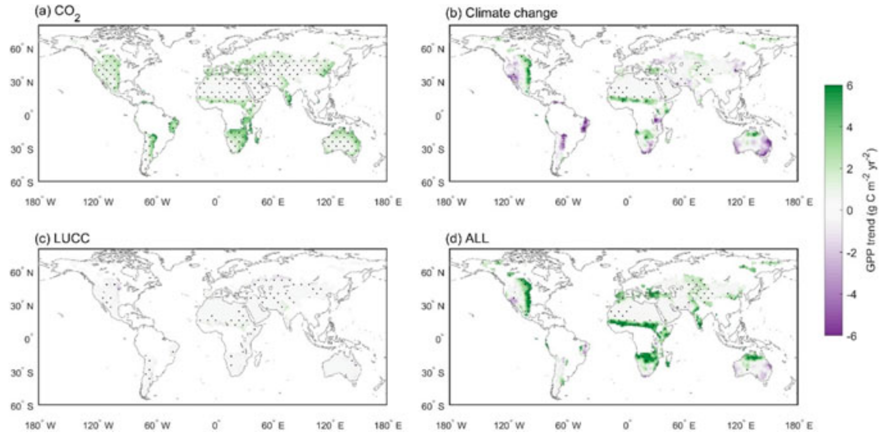

Fig. 6.17 Spatial patterns obtained by attribution analysis of dryland GPP trends during 1980–2018. **a** CO_2-induced dryland GPP trend estimated from multiple DVGMs. **b** Climate change-induced dryland GPP trend calculated from ensemble DGVMs. **c** Land cover and use change-induced dryland GPP trend estimated from multiple DGVMs. **d** Dryland GPP trend simulated from multiple DGVMs. Stippling indicates pixels with a significant GPP trend ($p < 0.05$)

Overall, the CO_2 fertilization effect was the main driver of the increasing trend in GPP in 63% of the dryland countries globally, including Australia, the United States, and China. Climate change was the main driver for 37% of dryland countries, including Sudan. Among all countries, the increasing trend in GPP was strongest in dryland areas of the United States ($8.32 \pm 3.83 \times 10^{-3}$ Pg C yr^{-2}), Sudan ($5.81 \pm 3.92 \times 10^{-3}$ Pg C yr^{-2}), and China ($5.31 \pm 4.58 \times 10^{-3}$ Pg C yr^{-2}). Significant increases in dryland GPP were clustered in the central United States, Sahel, southern Africa, India, and northern Australia. Small decreases in GPP in drylands were found in the southwestern United States, eastern Brazil, and eastern Australia (Fig. 6.17).

Whether the beneficial effect of CO_2 fertilization on photosynthesis vegetation will persist into a warmer and drier future is of critical concern. Simulations under RCP8.5, obtained from multiple earth system models (ESMs) provided by the Coupled Model Intercomparison Project Phase 5 (CMIP5), predict a linear enhancement of GPP in drylands over the twenty-first century (Fig. 6.18), in response to higher CO_2 and associated climate change. Large increases in GPP are predicted for drylands in southern South America, Sahel, southern Africa, India, and northern China (more than 600 g C m^{-2} yr^{-1} by the end of the century). GPP increases in dryland contribute substantially to global GPP increases, which could help slow down the warming and increase ecosystem service provisioning. However, there is substantial variation in GPP estimates derived from different DGVMs and ESMs, due to differences in how vegetation structure and function are simulated (Anav et al. 2013; Murray-Tortarolo et al. 2013; Wang et al. 2011). Therefore, greater efforts to improve model structures and benchmark model results are still needed. Expanding field observations in drylands, combining data from multiple sources, and developing

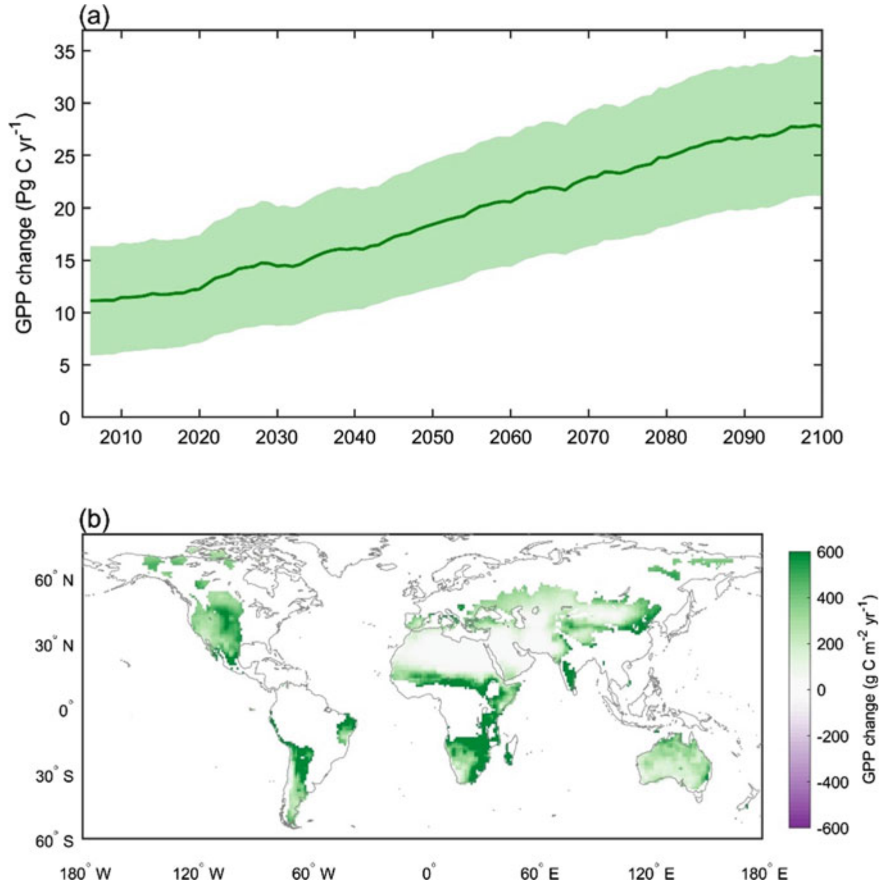

Fig. 6.18 Changes in annual mean GPP under RCP 8.5. **a** Temporal variation in GPP change during 2006–2100 estimated from multiple earth system models (ESMs) in CMIP5. Data have been smoothed with a 5-year running window. Shaded areas represent the standard error among different ESMs. **b** GPP change predicted using multiple ESMs in CMIP5 for 2081–2100 relative to 1986–2005

suitable algorithms for dryland ecosystems are especially important for monitoring and predicting variation in GPP in these areas (Smith et al. 2019).

Croplands are nonnegligible parts of the global dryland ecosystems, which provide the major food sources in humans' daily life. In contrast to the GPP used for the natural vegetation growth, crop yield is a more representative index of the productivity of the agricultural ecosystems. Wheat, rice, maize, and soybeans are the world's 'four' staple foods, which provide two-thirds of human caloric intake (Tilman et al. 2011). Thus, yield changes for these four major crops were assessed in the global dryland ecosystems.

The historical yield data used here were from a recently released global gridded dataset of historical yields for major crops (GDHY), which is a hybrid of agricultural census statistics from FAOSTAT and satellite remote sensing (Iizumi and Sakai 2020). According to the GDHY dataset, the crop yields in the dryland increased significantly at an average rate of 0.019–0.051 t ha^{-1} yr^{-1} during 1982 − 2016 (Fig. 6.19), that is to say an increase of 200 − 300% during the past 35 years. The average yield of maize (C4 crop) was higher than wheat, rice and soybean (C3 crop), but the relative change was smaller than the other three crops. Factorial simulations from global gridded crop model (GGCM; EPIC, GEPIC, pDSSAT, LPJ-GUESS, LPJmL, and PEGASUS) intercomparison, coordinated by the Agricultural Model Intercomparison and Improvement Project (AgMIP) (Rosenzweig et al. 2013) as part of the Inter-Sectoral Impact Model Intercomparison Project (ISI-MIP) (Warszawski et al. 2014) indicate that yield changed little due to the climate change from 1980 to 2005 (Trend = −0.001–0.002 ha^{-1} yr^{-1}). Therefore, the historical reported yield increase of 200 − 300% in the drylands might be attributed mainly to the technology advances and management improvements, such as modern cultivars applications, more harbors, fertilizers and irrigation inputs, and advanced pest and diseases controls.

The spatial patterns of yield trends (Fig. 6.20) also indicate that most of croplands in the drylands experienced the obvious yield increases, except some specific crops in hot regions (e.g., wheat and maize in East Africa, and wheat in North Australia). The north part of dryland regions, such as North America, Europe, and North China (yield trends were all above 0.05 t ha^{-1} yr^{-1}), dominate the boosting of global crop yield during the last three decades. The GGCMs simulated yield trends due to climate change were much smaller than the statistical trends across most of

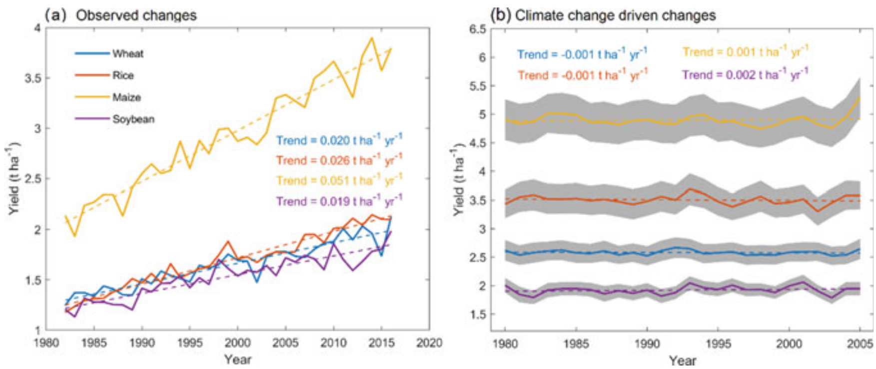

Fig. 6.19 Historical trends of crop yields from observations and climate change driven simulations in the drylands. **a** Temporal variation in annual mean yield of major crops provided by the GDHY (1982–2016), a reanalysis dataset based on the FAOSTAT database. **b** Global gridded crop models (GGCMs) simulated yield trend (1980–2005) from the climate change impact. The absolute yields from simulations were higher than those from the census statistics, due to the potential management conditions by simulations. The grey shades represent the standard errors from the ensembles of thirty members (five crop models coupled with five global climate models)

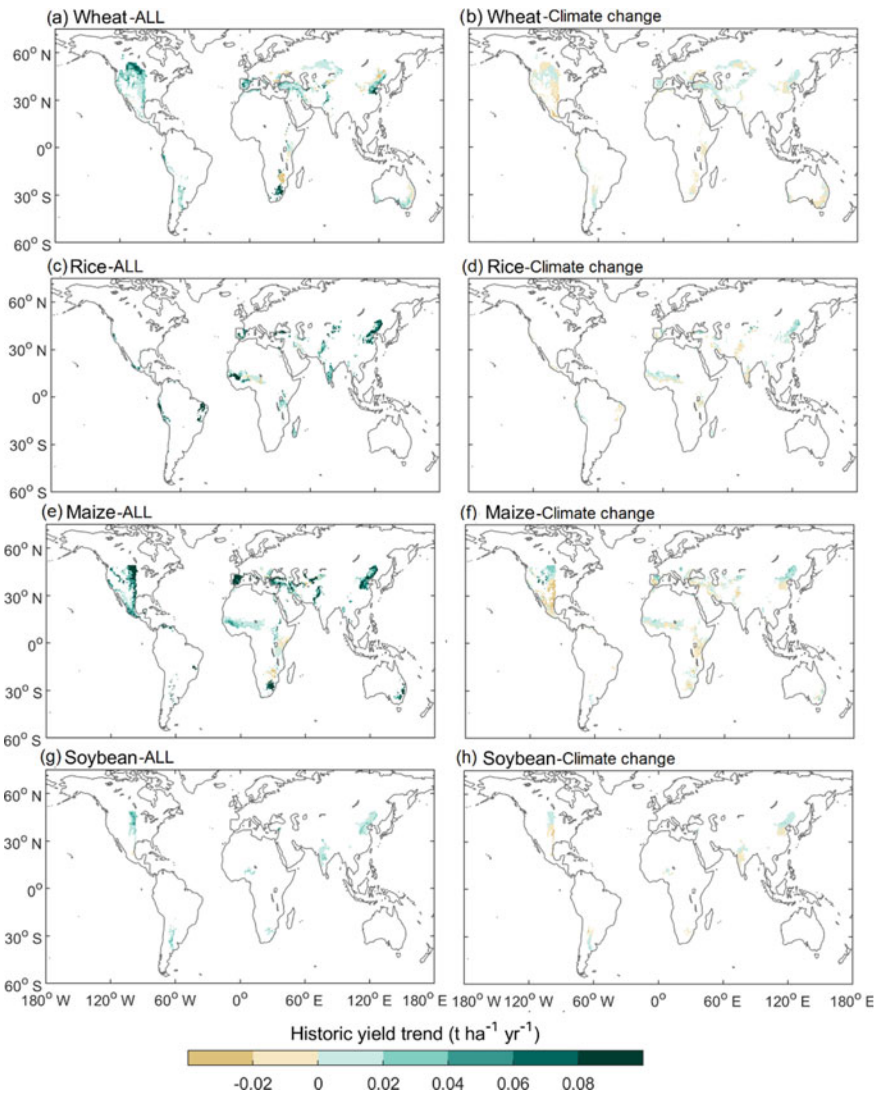

Fig. 6.20 Spatial patterns of crop yield trends in the drylands obtained from GDHY observations (1982–2016) and GGCMs simulations driven by climate change (1980 − 2005). **a**, **c**, **e**, **g** Observed yield trend for wheat, rice, maize and soybeans. **b**, **d**, **f**, **h** Simulated yield trend driven by climate change for wheat, rice, maize, and soybeans

regions. However, the historical climate change impact on yield shows considerable heterogeneity across the studied regions. The north parts of drylands benefited from the climate change, but the south parts suffered the yield loss. The contrast between impacts on the north and south of drylands might be related to the differences in the background climate, especially the temperature (Challinor et al. 2014; Zhao et al. 2016).

The future yield changes were assessed with the GGCMs ensembles coupled with 5 global climate models (GCMs). The patterns of ensemble medians (Fig. 6.21) show that under RCP 2.6 scenario, if the CO_2 effects were not considered, the crop yield changes until the end of century (2070–2099) will be positive or negative across regions, similar to the patterns of historic trend but with a larger magnitude. In general, wheat, rice, and soybeans in the north part of America, Europe, and Asia will generally have yield gains, up to 10% relative to the baseline period; Africa and some hot regions might have slight negative yield changes. For maize, large parts of the drylands might suffer yield loss, especially in Africa (up to 50% of yield loss) where hungers already happened. If the climate becomes warmer, from RCP 2.6 to RCP 4.5 scenario, more regions will see the yield loss and the regions with negative impact might become more vulnerable. Overall, to the end of this century, the climate scenario of RCP 2.6 (RCP 4.5) will change the average yield of wheat, rice, maize, and soybeans for the global drylands by 1.2% (−5.7%), −2.1% (−7.2%), −12.3% (−14.1%), and −9.2% (−17.9%), respectively.

6.3.2 Carbon Sink

Despite relatively low mean productivity, vegetation in drylands dominates trends and interannual variability in the terrestrial carbon sink (Ahlström et al. 2015; Poulter et al. 2014). Dryland ecosystems acted as carbon sinks during 1980–2018, as indicated by the annual mean net biome production (NBP) of 0.25 ± 0.19 Pg C yr^{-1} (Fig. 6.22a) estimated by an ensemble of 14 DGVMs from the TRENDY project. Three countries fixed more carbon than any others during this time period, including the United States (0.038 ± 0.019 Pg C yr^{-1}), Australia (0.030 ± 0.033 Pg C yr^{-1}), and Russia (0.022 ± 0.015 Pg C yr^{-1}). However, there was a large interannual variation in NBP during 1980–2018, resulting in a non-significant increasing trend of 0.0064 ± 0.0093 Pg C yr^{-2}.

Increasing atmospheric CO_2 concentration, climate change, and land cover and use changes are the three main drivers of variation in global carbon sinks (Le Quéré et al. 2009; Piao et al. 2009; Sitch et al. 2015). According to DGVM factorial simulations, the CO_2 fertilization effect (0.0084 ± 0.0036 Pg C yr^{-2}; $p < 0.01$) dominated the increase in dryland GPP (Fig. 6.16b), thus significantly contributing to the NBP trend (Fig. 6.22b). However, large increases in NBP were offset by significant reductions (-0.0049 ± 0.0039 Pg C yr^{-2}; $p < 0.01$) caused by land cover and use changes, which was a carbon source during 1980–2018. Although drylands are sensitive to climate, climate change explained a non-significant proportion of the total NBP (0.0029 ± 0.0077 Pg C yr^{-2}; $p > 0.1$). Overall, the trend in NBP of dryland ecosystems was not significant during 1980–2018, and the DGVMs showed marked cross-model variations in dryland carbon sinks (Fig. 6.22).

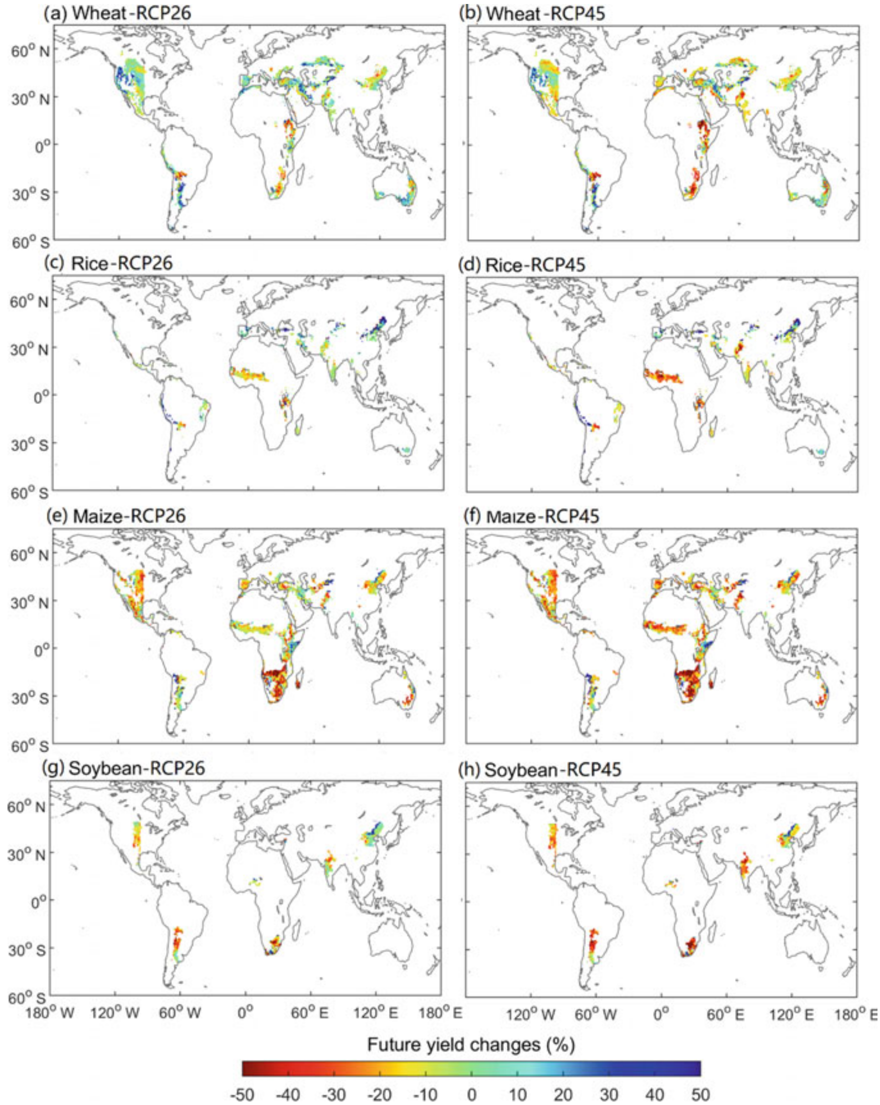

Fig. 6.21 Median crop yield changes (%) for RCP 2.6 and RCP 4.5 (2070 − 2099 in comparison to 1971 − 2005) without CO_2 effects over all five GCMs × six GGCMs simulations in the drylands. **a**, **c**, **e**, **g** Simulated yield changes for RCP2.6 for wheat, rice, maize, and soybean. **b**, **d**, **f**, **h** Simulated yield changes for RCP8.5 for wheat, rice, maize, and soybeans

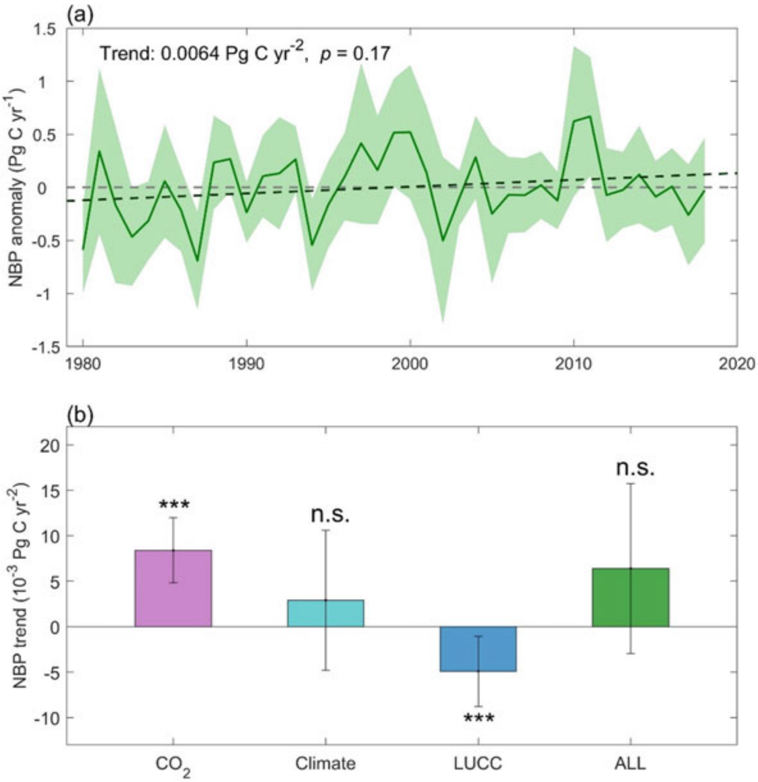

Fig. 6.22 Trends in net biome production (NBP) in drylands and attribution analysis of the NBP trend for 1980–2018. **a** Temporal variation in annual mean NBP from multiple DGVMs obtained from the TRENDY project, and the annual mean SIF from a CSIF product (2000–2018). **b** Attribution analysis of the DGVM-estimated NBP trend: effects of CO_2 fertilization, climate change, and land cover and use change, and the overall NBP trend. Error bars represent standard deviations across multiple DGVMs. Stars represent significance levels for the estimated NBP trend. ***$p < 0.01$; **$p < 0.05$; *$p < 0.1$; n.s., $p > 0.1$

The CO_2 fertilization effect enhanced NBP over almost all dryland regions in 1980–2018 (Fig. 6.23). Similar to the role of CO_2 fertilization in GPP trends (Fig. 6.17), greater increases in NBP were found in transitional areas between dry and wet regions. There was a large spatial variation in the effect of climate change on NBP (Fig. 6.23). Strong positive effects were found in the Sahel and other parts of Africa, whereas negative effects were found in eastern and western Australia, and southwestern Europe (Fig. 6.23). Land cover and use changes drove NBP reductions in western South America, Sahel, and eastern Africa, along with slight increases in China and Eastern Europe (Fig. 6.23). All effects combined, the overall NBP trends were spatially variable, with strong positive trends in the central United States and eastern and southern Africa (Fig. 6.23). Although the influence of climate change on NBP was not significant (Fig. 6.22), it did explain a large proportion (72%) of

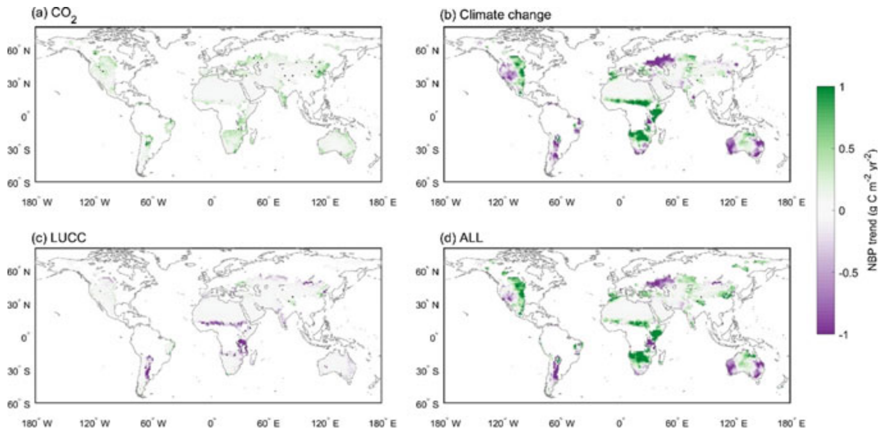

Fig. 6.23 Spatial patterns obtained by attribution analysis of NBP trends for 1980–2018. **a** CO_2-induced dryland NBP trends estimated from multiple DGVMs. **b** Climate change-induced NBP trends calculated from ensemble DGVMs. **c** Land cover and use change-induced NBP trends estimated from multiple DGVMs. **d** NBP trends simulated from multiple DGVMs. Stippling indicates pixels with a significant NBP trend ($p < 0.05$)

the variance in the spatial pattern of NBP, thus confirming the sensitivity of dryland ecosystems to climate change (Ahlström et al. 2015).

Increasing NBP trends were most pronounced in drylands in China ($9.75 \pm 7.63 \times 10^4$ Pg C yr^{-2}), followed by the United States; both were dominated by the effect of CO_2 fertilization. Other areas showing increases included Botswana, Sudan, and Zambia, where drylands account for nearly 80% of the total land area of these countries. This trend was driven by climate change. Overall, climate change dominated the NBP trends in roughly two-thirds of dryland countries, whereas CO_2 fertilization and land cover and use changes dominated the trends in approximately 18% and 15% of all countries, respectively.

The ESM simulations provided by CMIP5 predict an increase in NBP in 2006–2100 in dryland ecosystems (under the RCP8.5 scenario), relative to 1986–2005, although there is a large interannual variation in these predictions (Fig. 6.24). Dryland NBP is predicted to increase from the beginning to middle of this century, followed by a decline toward the end of the century. Thus, annual mean NBP in drylands is expected to increase by 0.42 ± 0.24 Pg C yr^{-1} during 2081–2100, with the largest increase projected for western China (Fig. 6.24). Annual mean NBP is also predicted to increase in the northern hemisphere during this period, with simultaneous declines in the southern hemisphere. Under RCP8.5, carbon sinks in dryland ecosystems are expected to increase across 80% of the total global dryland area during 2081–2100.

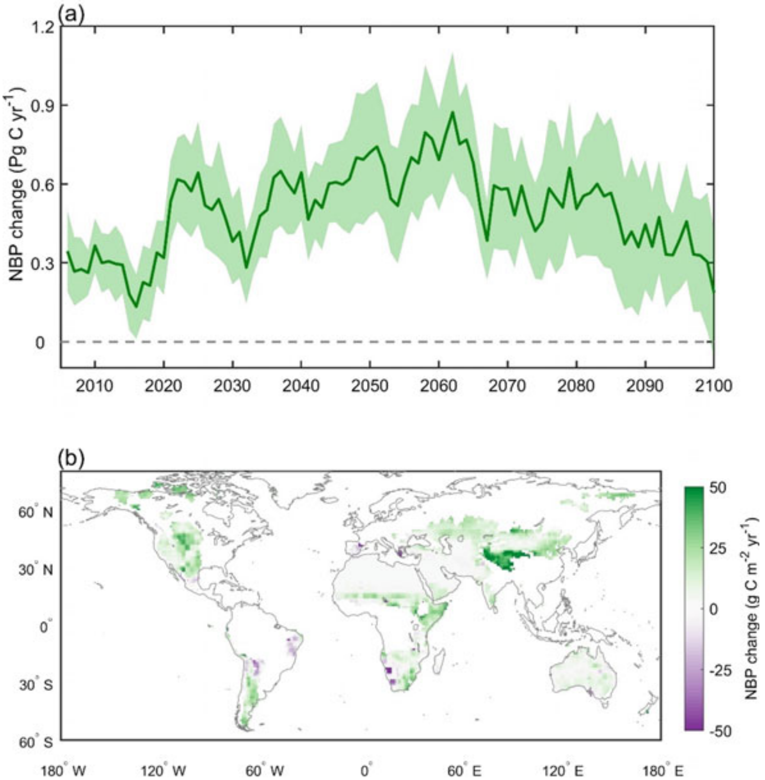

Fig. 6.24 Changes in annual mean NBP under RCP 8.5. **a** Temporal variation in NBP during 2006–2100 estimated using multiple ESMs in CMIP5. Data is smoothed using a 5-year running window. Shaded areas represent standard error among different ESMs. **b** NBP change predicted using multiple ESMs in CMIP5 during 2081–2100 relative to 1986–2005

6.3.3 Carbon–Water Coupling

The structure and function of vegetation in water-limited biomes are regulated by surface water availability and variations thereof. When plants open stomata to take up CO_2 for photosynthesis, they simultaneously lose water due to transpiration. This process links ecosystem carbon and water cycles. Plants can actively adjust the stomatal aperture to regulate the rates of photosynthesis and transpiration in response to changes in humidity, soil moisture, and atmospheric CO_2 concentration. This leads to changes in the WUE (carbon gained per unit of water lost) of vegetation.

Abundant evidence, provided by experimental and modeling studies, shows that plants can partially close stomata under higher atmospheric CO_2 concentrations, leading to higher leaf-level WUE (Cheng et al. 2017; Keenan et al. 2013; Knauer et al. 2017; Lee et al. 2011). For example, multi-year FACE experiments in semi-arid grasslands showed that elevated CO_2 concentration (>180 ppm above ambient CO_2

concentration) increased the rate of photosynthesis by 10% and decreased leaf stomatal conductance by 22%, translating to a 40% increase in leaf-level WUE (Lee et al. 2011). Greater WUE allows plants to reduce their water use while maintaining the same or increasing the rate of photosynthesis (Cheng et al. 2017). Climate change, particularly temperature-driven vapor-pressure deficits and precipitation-driven soil moisture changes, also affect WUE to some degree (Hatfield and Dold 2019). Recent evidence shows that under drought conditions, with low soil moisture and high vapor-pressure deficits, plants can maintain high WUE to alleviate extreme water stress (Peters et al. 2018). At the ecosystem level, WUE is often measured as the ratio of GPP to evapotranspiration (including transpiration, canopy interception, and bare soil evaporation) (Huang et al. 2015). In arid environments, those non-biological water fluxes (i.e., interception and soil evaporation) contribute substantially to evapotranspiration variations, responsive to changes in vegetation structure, such as an increase in leaf area. Therefore, leaf-level WUE variations may not scale to the ecosystem level.

According to the DGVMs, dryland ecosystem-level WUE increased significantly during 1980 − 2020, at an average rate of 0.039 g C m^{-2} mm^{-1} yr^{-1} ($P < 0.05$; Fig. 6.25). Increased atmospheric CO_2 concentration is the primary driver of this increase, with a descending gradient from less to more extreme aridity (Fig. 6.25). This CO_2-driven increase is slightly (~10%) offset by a decreasing trend caused by climate change. Unlike the positive effects of elevated CO_2, climate change has significant negative effects on WUE in many dryland areas, including Australia, the western United States, South Africa, parts of northern China, Eastern Europe, and western Asia. Anthropogenic land use and management contributed little to the change in WUE during this period, although a decreasing trend in WUE was observed in the northern fringe of the Eurasian dryland area, i.e., the Mongolian steppe (Fig. 6.26). This regional decrease is at least partially attributable to grassland deterioration caused by overgrazing.

The ability to self-adjust water use efficiency is beneficial to the growth of vegetation. Stomatal regulation of transpiration under elevated CO_2 concentrations conserves water and reduces water limitations for photosynthesis and growth in arid and semi-arid ecosystems, known as the water-saving effect (Leuzinger and KÖRner 2007). C4 species benefit more than C3 species from CO_2-induced water saving, as they show a stronger decline in stomatal conductance, allowing for greater increases in vegetation cover (Ainsworth and Rogers 2007; Morgan et al. 2011). Under global warming, the amount of conserved water is potentially sufficient for offsetting the higher water demands driven by a warmer atmosphere; this is likely the key mechanism underlying the enhanced carbon gains and increased biomass observed in dryland ecosystems (Lian et al. 2021). In drylands, the beneficial effects of increased CO_2 on vegetation growth through water saving is amplified by OF and processes including direct stimulation of photosynthesis (CO_2 fertilization effect) (Zhu et al. 2016), longer growing seasons (Hufkens et al. 2016), attenuated soil moisture stress in areas with increased precipitation (Al-Yaari et al. 2020), and human management activities (Chen et al. 2019).

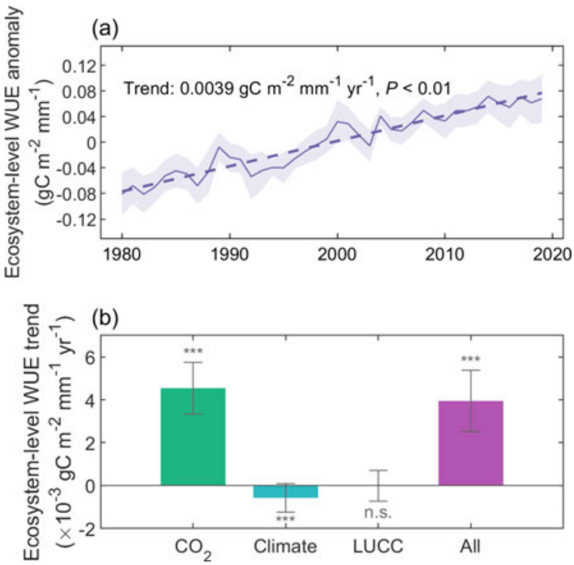

Fig. 6.25 Changes in ecosystem-level WUE in drylands during 1980–2018. **a** The solid line indicates the ensemble mean WUE of 14 DGVMs, with the shaded areas indicating standard deviations. Dashed lines indicate linear trends. **b** Trends in mean WUE attributed to elevated CO_2, climate change (Climate), and land cover change (LUC), based on DGVM factorial simulations. Error bars indicate standard deviations across models. ***$p < 0.01$; **$p < 0.05$; *$p < 0.1$; n.s., $p > 0.1$

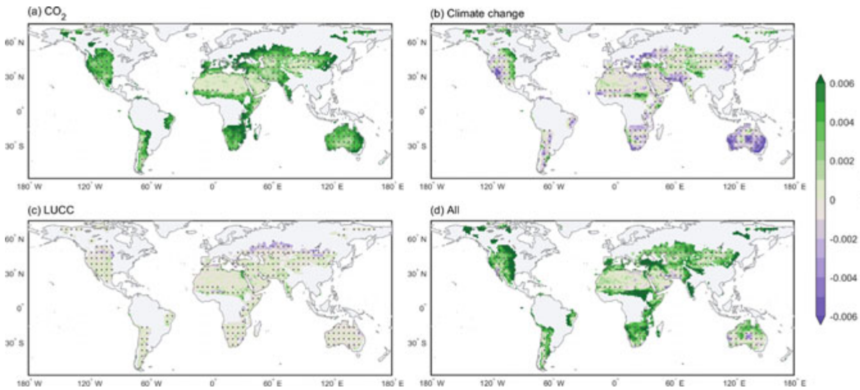

Fig. 6.26 Spatial patterns of the WUE trends for 1980–2019 over drylands, attributed to **a** elevated CO_2, **b** climate change, **c** land cover change (LUC), and **d** all factors combined. Changes were calculated using DGVM factorial simulations. Stippling indicates areas with statistically significant trends ($p < 0.05$)

6.4 Changes in Hydrological Regimes

In areas with scarce water resources, changes in water availability have profound impacts on ecosystem functions and services, including vegetation and agricultural productivity (Ciais et al. 2005; Gray et al. 2016), freshwater supplies (Greve et al. 2018), and ultimately the welfare of human societies. In recent years, the potential of exacerbating dryland water scarcity in drylands under climate change has gained widespread attention and led to extensive research (Feng and Fu 2013; Huang et al. 2016; Park et al. 2018; Zhang et al. 2020). A large number of studies use the aridity index which balances water received by the surface (precipitation) and that demanded by the atmosphere (potential evapotranspiration) as a proxy for assessing changes in land aridity (Feng and Fu 2013; Huang et al. 2016; Park et al. 2018). Under climate change, aridity index values tend to decrease because warming-induced increases in potential evapotranspiration outpace concurrent precipitation increases (Fu and Feng 2014; Huang et al. 2016; Sherwood and Fu 2014). More rapid increases in potential evapotranspiration are mainly attributed to an insufficient increase in actual water vapor relative to saturated water vapor which increases exponentially with temperature (i.e., higher vapor-pressure deficits). Therefore, assessments based on the aridity index are characterized by a broad trend of surface drying in global drylands, which translates to continuous expansion of the geographical extent of drylands. Climate model simulations under RCP 4.5 and RCP 8.5 indicate that climate change could lead to a 11–23% expansion in global dryland area by the end of the twenty-first century (Huang et al. 2017b, 2016).

A more comprehensive understanding of dryland aridity changes can be achieved by assessing hydrological metrics such as soil moisture, runoff, and terrestrial water storage (Lian et al. 2021; Roderick et al. 2015). Similar to atmospheric aridity metrics like the aridity index and vapor-pressure deficit, assessments based on soil moisture and runoff also indicate decreasing availability of freshwater resources over drylands. For example, station-recorded streamflow of the world's largest rivers flowing through drylands showed an overall decrease of 11.9% during 1948–2016 (Dai et al. 2009; Lian et al. 2021). Microwave satellite observations of near-surface soil moisture indicate that 38.4% of global drylands have shown a significant drying trend since 1979, whereas only 2.9% showed a wetting trend (Feng and Zhang 2015). For the period 2002–2016, gravimetric sensors onboard NASA's Gravity Recovery and Climate Experiment satellites indicated a robust decline in endorheic water storage, by 106.3 Gt yr^{-1}, mostly in global dryland areas (Wang et al. 2018).

Although surface water availability decreases following near-surface atmospheric drying, there is apparent divergence in long-term trends of water availability. A recent study evaluating water-stressed areas at the global scale used thresholds of various hydrological parameters and reported that the rate of dryland expansion inferred from soil moisture and runoff (i.e., the area under soil moisture and runoff deficits) was much smaller than that inferred from vapor-pressure deficits or the aridity index (Lian et al. 2021) (Fig. 6.27). This divergence indicates that although increased atmospheric evaporative demand would accelerate soil water depletion, this has not

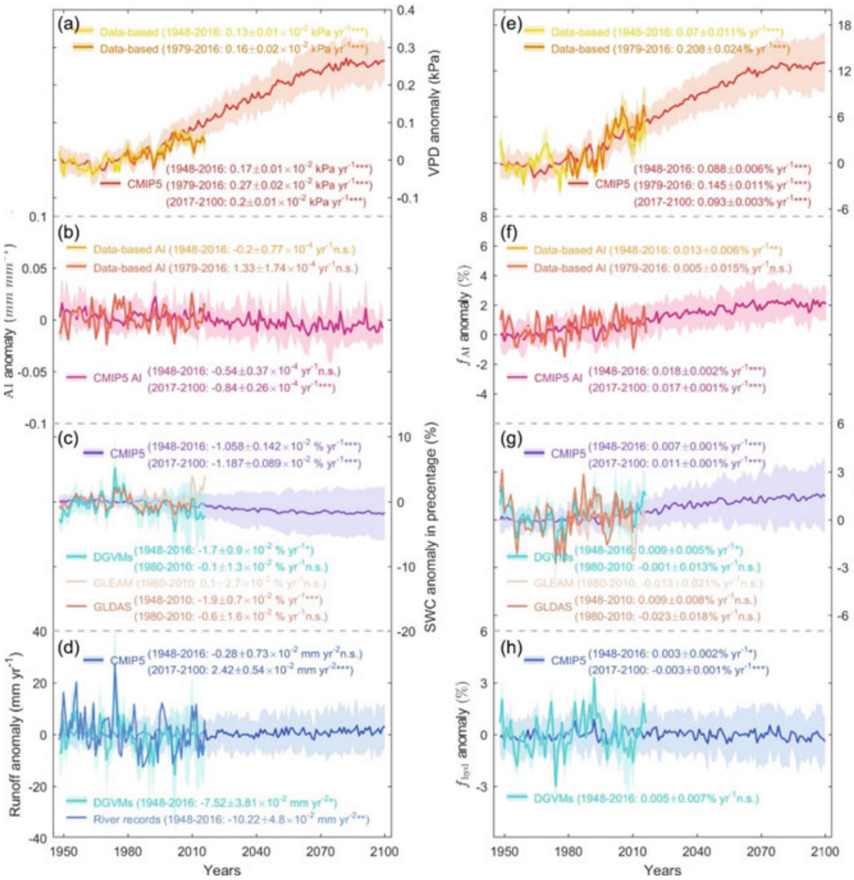

Fig. 6.27 Past and future dryland changes evaluated using different aridity measures including vapor-pressure deficit (VPD), aridity index (AI), soil water content (SWC), and runoff. This Fig. is based on the data used in Lian et al. (2021). **a–d** Various observational and model-derived anomalies in different aridity measures averaged over AI-defined baseline regions of drylands during 1961–1990. **e–h** Same as panels a–d, but showing anomalies (as %) in the fraction of water-stressed land areas (drylands) evaluated by these aridity measures. Shaded areas represent the 95% confidence intervals of multiple data sources (for VPD and AI) or model results (for DGVMs, or CMIP5 ESMs under 'historical' and 'RCP 4.5' scenarios)

fully translated into increased soil moisture and runoff deficits over drylands. Climate models also project persistence of the contemporary trend of soil drying to the end of the twenty-first century, but this is again less severe than that inferred from the projected atmospheric drying (Lian et al. 2021). However, projections under a warmer climate indicate that runoff increases may prevail in semi-arid and sub-humid dryland regions, although runoff decreases are projected for extremely arid regions (Lian et al. 2021).

The evolving ecosystem water consumption under climate change is thought to be a critical driver of the water available in soils and streams (Greve et al. 2014; Lian et al. 2021; Swann et al. 2016). In particular, plant physiological responses to higher CO_2 underlie the less severe surface drying trend than atmosphere-centered metrics. Stomatal regulation of transpiration under higher CO_2 favors the partitioning of precipitation towards runoff and soil moisture, thereby ameliorating the surface water losses driven by global warming (Greve et al. 2019; Scheff 2018; Swann et al. 2016). In future projections with, for example, quadrupled CO_2 from the pre-industrial level, modelling studies suggest that this CO_2 physiological forcing has a dominant effect on increased runoff production over CO_2 radiative forcing (Lian et al. 2021) (Fig. 6.28). Meanwhile, CO_2 physiological forcing substantially offsets radiative forcing-induced soil moisture decline (Lian et al. 2021; Swann et al. 2016). Although climate change dominates regional drying and wetting patterns, CO_2 physiological forcing acts to mitigate drying (or amplify wetting), particularly over some less arid dryland areas with sizeable vegetation cover (Fig. 6.29).

Although global patterns of aridification and dryland expansion remain debatable, there are clear regional hotspots of terrestrial water loss and increasing drought risks. For example, tree-ring reconstructions indicate an abrupt and unprecedented reduction in soil moisture during the twenty-first century in the western United States and interior East Asia, which overrides the range of natural variability of previous centuries (Williams et al. 2020; Zhang et al. 2020). This shift toward drier regimes is the result of a long-term precipitation deficit, and is further amplified and propagated through land–atmosphere coupling. Specifically, drier soils can suppress evaporative cooling and amplify increases in near-surface air temperature, which in turn amplifies soil moisture depletion (Seneviratne et al. 2010; Zhang et al. 2020). The

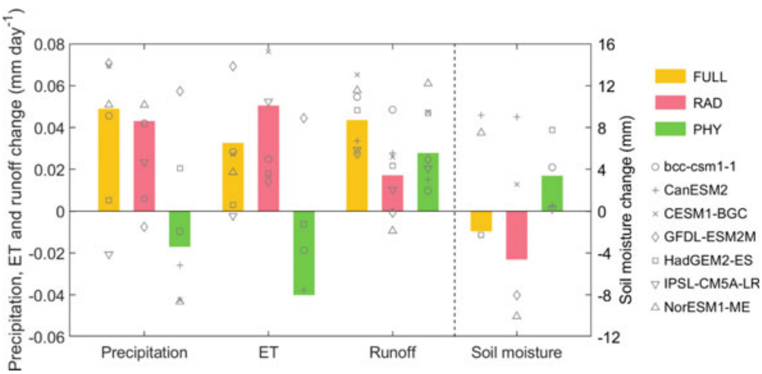

Fig. 6.28 Future changes in precipitation, evapotranspiration (ET), runoff, and soil moisture in global drylands. Bars (multi-model ensemble means) and symbols (individual models) show fractional changes in these variables caused by plant physiological responses to a quadrupling of atmospheric CO_2. Predictions are given for all CO_2-based forcings (FULL), and to physiological only forcings (PHY), and radiative only forcings (RAD). Changes were determined by subtracting the ensemble mean of the last 20 years from the first 20 years of seven CMIP5 models (bcc-csm1-1, CanESM2, CESM1-BGC, GFDL-ESM2M, HadGEM2-ES, IPSL-CM5A-LR, NorESM1-ME)

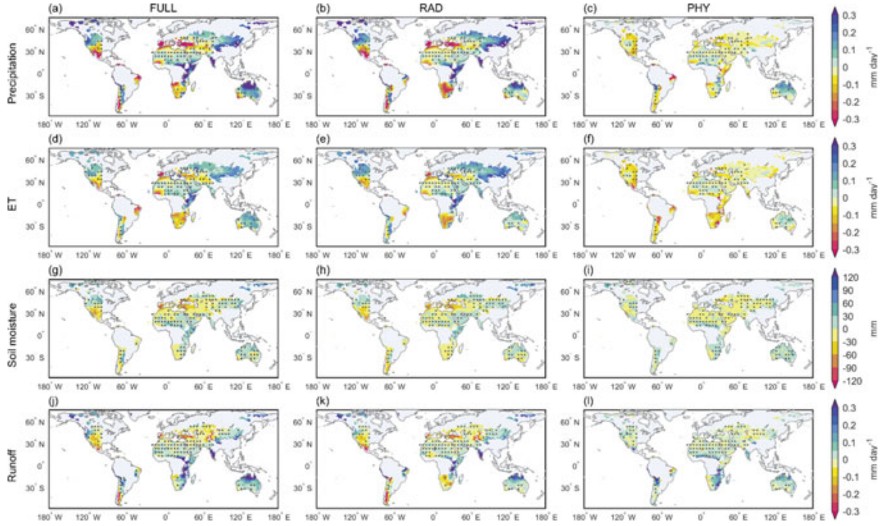

Fig. 6.29 Spatial patterns of future changes in **a–c** precipitation, **d–f** ET, **g–i** soil moisture, and **j–l** runoff from CMIP5 predictions. Columns represent predictions from all CO_2-based forcings (FULL), physiological only forcings (PHY), and radiative only forcings (RAD). Changes were determined by subtracting the ensemble mean of the last 20 years from the first 20 years of seven CMIP5 models (bcc-csm1-1, CanESM2, CESM1-BGC, GFDL-ESM2M, HadGEM2-ES, IPSL-CM5A-LR, NorESM1-ME). Dots indicate where at least five models agreed on the sign of the trend based on multi-model mean

unprecedented loss of water storage over some dryland regions is also the result of human activities such as afforestation, overgrazing, and agricultural expansion. For example, large-scale ecological restoration efforts in northern China have consumed an enormous amount of water and caused a decline in soil moisture (Feng et al. 2016; Zhao et al. 2020). These unintended hydrological consequences call for caution in additional reforestation programs in water-limited regions, where it is necessary to balance the benefits of forest ecosystem services with the costs of water resource consumption.

6.5 Vulnerability of Dryland Ecosystem and Human

6.5.1 Resistance and Resilience of Dryland Ecosystems

Drylands are among the most vulnerable ecosystems to anthropogenic climate change (IPCC 2019). Given the large predicted shifts in climate, especially warming (Huang et al. 2017a) and increased drought severity and frequency (Chiang et al. 2021), it is unclear how the stability of dryland ecosystem (i.e., the ability of ecosystem to retain their functions under climatic perturbations) will change in a hotter and drier future.

The stability of a given ecosystem in response to external disturbance is generally assessed by two indices: resistance and resilience. Resistance represents the ability of vegetation to withstand disturbance, and is typically quantified as the magnitude of the reduction in vegetation growth or production during a disturbance event (Isbell et al. 2015; Lloret et al. 2011). Resilience represents the rate at which ecosystem functions return to their pre-disturbance state following an event (Gazol et al. 2018; Isbell et al. 2015), and has been defined as "the amount of disturbance a system can withstand before it crosses a threshold and fundamentally changes" (Ciemer et al. 2019; Liu et al. 2019; Scheffer et al. 2009; Verbesselt et al. 2016).

The resistance of dryland ecosystems to climate change varies among regions (Fig. 6.30) (Gazol et al. 2018). Low resistance to climate variation has been identified in the biomes of the prairies of North America, grasslands in Asia, and the Caatinga in Brazil (Seddon et al. 2016). These biomes have relatively high production, and are sensitive to variations of precipitation. On the contrary, biomes in the Sahel, Australian outback, and Middle East may be able to sustain relatively low but stable productivity despite large variability in precipitation, suggesting strong resistance to changes in water availability (Seddon et al. 2016).

The resilience of dryland ecosystem also varies widely across regions and biomes. Numerous evidence suggested that grassland production could fully recover or even overshoot within one year after droughts (Isbell, et al. 2015). Such rapid recovery could benefit from the compensate regrowth during rewetting events after droughts (Chen et al. 2020). In addition, grassland ecosystem has fast turnover rate in species composition. Drought events could lead to increasing abundance of drought-tolerate species, which also contributes to the rapid recovery in grasslands (Hoover et al. 2014). On the contrary, reduction of tree growth in forests under droughts could last for 1 to 4 years, and such drought "legacy effect" is especially pervasive in dry forests (Anderegg et al. 2015; Kannenberg et al. 2019). Vegetation growth rates are slow under harsh conditions in dry compared to humid regions, and these water-limited systems, therefore, take a long time to recover (Schwalm et al. 2017).

Some evidence suggests broad changes in ecosystem resistance and resilience over the past several decades, along with divergent response patterns among species (Anderegg et al. 2020; Fang and Zhang 2019; Gazol et al. 2018; Li et al. 2020). For example, an investigation of changes in resistance under repeated drought events showed decreased resistance among gymnosperm-dominated forests, but increasing resistance among angiosperm-dominated forests. This pattern of divergent behavior between gymnosperm and angiosperm species suggests that angiosperms may show stronger acclimation responses, whereas gymnosperms suffer greater stress accumulation (Anderegg et al. 2020). Whether forests are becoming more vulnerable to droughts depends not only on the changes of resistance during droughts, but also on the post-drought recovery. A recent study based on global tree-ring datasets showed a temporal trade-off between resistance and recovery in gymnosperm forests over the past six decades, with trees becoming more sensitive (decreased resistance), but recovering faster (increased resilience) during the post-drought period

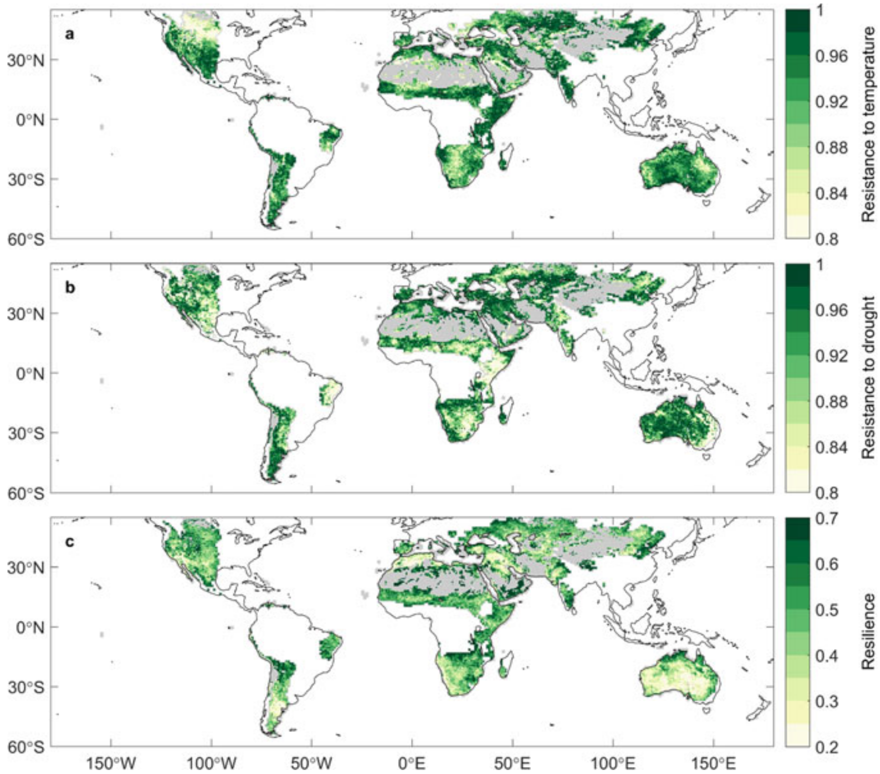

Fig. 6.30 Ecosystem resistance to **a** temperature and **b** drought, and **c** ecosystem resilience in drylands for the period 1982–2016. The resistance and resilience of ecosystem productivity (defined as the NDVI) were assessed based on the AR1 model ($NDVI_t = \alpha \times T + \beta \times SPEI + \gamma \times NDVI_{t-1} + \varepsilon$). Resistance was defined as 1 minus the absolute value of the model coefficient (1-$abs(\alpha)$ for temperature and 1-$abs(\beta)$ for drought). Resilience was defined as 1 minus the absolute value of the model coefficient of $NDVI_{t-1}$ (1-$abs(\gamma)$). High values (shown in green) indicate strong resistance or resilience to climate change. Bare land areas (NDVI < 0.1) in dryland regions (AI < 0.65) are shown in grey

(Li et al. 2020). Enhanced post-drought recovery could be linked to the fertilization effect of an increased atmospheric CO_2 concentration, which would suggest that the rise in CO_2 may at least partly alleviate drought stress in gymnosperms (Li et al. 2020; Liu et al. 2017).

The hotter, drier climate of the future presents a threat to dryland ecosystems. The fate of these systems, i.e., whether they will adapt to the future climate or undergo degradation and desertification, remains controversial. For example, global warming and drought may increase mortality rates in forests and woodlands in dryland areas (Brodribb et al. 2020; Choat et al. 2018). However, an increased CO_2 concentration could increase WUE and to some extent enhance resilience to drought, potentially compensating for the damage caused by droughts and overall warming (Sperry et al.

2019). Moreover, our understanding of the types of tree species that may survive or perish under severe drought conditions remains limited. There is a pressing need to quantify the climate stress thresholds at which tree mortality and forest dieback occur, to develop mitigation strategies for dryland ecosystems under climate change (Trumbore et al. 2015).

6.5.2 Water Scarcity of the Dryland Socio-economic System

In drylands, the limited available surface water is a challenge for dryland countries to maintain sustainable food production and economic development. Based on climate model projections, surface freshwater resources (mainly runoff) in drylands will see an overall insignificant increase through the 21st Century, suggesting that the water supply to the socio-economic system is almost unchanged. However, the water scarcity conditions are determined by a balance between changes of water demand and water supply. Human water demand is the potential amount of water use by agricultural (livestock/irrigation), industrial (energy/manufacturing), and domestic sectors, related directly to human population and the level of economic development. Approximately 1/3 of the population living in drylands depends on agriculture for food and livelihoods, often as their only source of income, thus agriculture is the sector of the greatest anthropogenic water use.

The rates of human population growth and economic development in drylands are faster than the global average, driving simultaneously faster growing demand for freshwater. Based on global hydrological model simulations forced by historical and future socio-economic factors, human water demand in dryland countries has almost doubled since the 1950s (Fig. 6.31). This dryland water demand is projected to further increase by 270% towards the end of the twenty-first century in the absence of effective mitigation efforts (Fig. 6.31). The increasing anthropogenic use of water is also spatially imbalanced, which is more intense over developing than developed countries (Fig. 6.31). The water demand by Africa countries is projected to increase continuously in the next few decades, primarily contributed by agricultural expansion to meet growing food demands (Fig. 6.31). However, there will be a decline in water demand by North America and Australia after 2030s (Fig. 6.31), with adoption of efficient water management measures and water conservation technologies.

In many dryland areas, the rapidly growing anthropogenic water demand is likely to be a major driver of future water deficits in drylands. Local residents may excessively exploit river and groundwater resources to meet the increasing water demand. This unsustainable water consumption would induce less water resources available for dryland ecosystems, and increase the vulnerability of these already fragile ecosystems to climate change, which, in a feedback, ultimately jeopardize human societies. To ensure food security and cope with the increasing water crisis, more practical strategies are needed to develop more water-efficient technologies and improve the overall efficiency of water management.

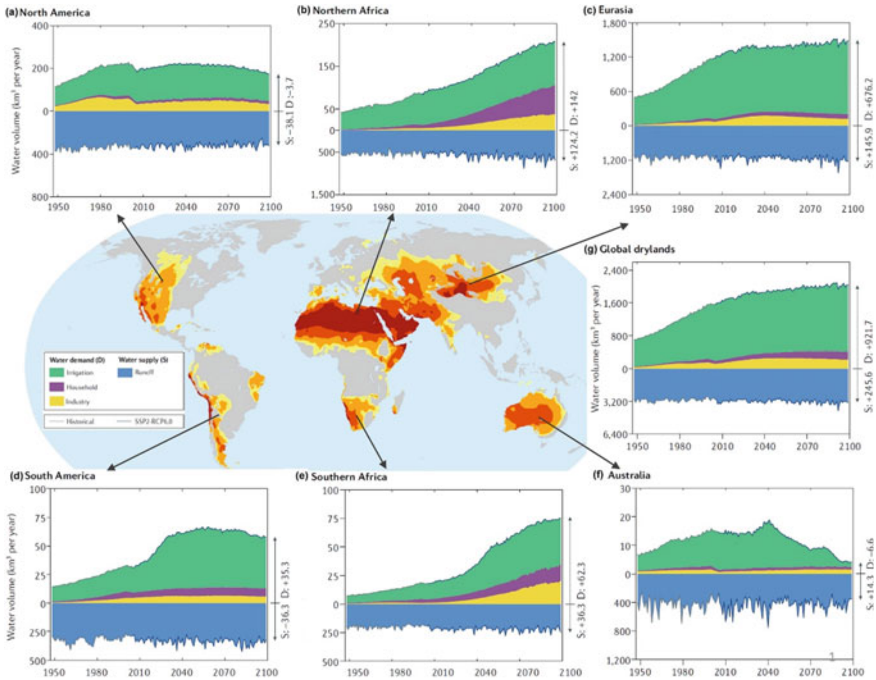

Fig. 6.31 Historical and future changes in total anthropogenic water use in global drylands. This figure is based on the data used in Lian et al. (2021). Water demand (D) is presented as the sum of agricultural, domestic, and industrial water withdrawal, and water supply (S) is primarily surface runoff. Note that the y-axis scale differs between D and S. The time series were derived from the ensemble mean of three global hydrological models (H08, MATSIRO and LPJmL) under the SSP2-RCP 6.0 scenario. Arrows indicate the amount of D and S during the 2090s, with the associated numbers showing relative changes from the 1961–1990 baseline

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Chapter 7
Dryland Social-Ecological Systems in Central Asia

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Abstract The countries of Central Asia are collectively known as Uzbekistan, Kyrgyzstan, Turkmenistan, Tajikistan and Kazakhstan. Central Asian countries have experienced significant warming in the last century as a result of global changes and human activities. Specifically, the five Central Asian countries' populations and economies have increased, with Turkmenistan showing the fastest growth rates in GDP and per capita GDP. Farmland change, forestry activities, and grazing are examples of land use/land cover change and land management in Central Asia. Land degradation was primarily caused by rangeland degradation, desertification, deforestation, and farmland abandonment. The raised temperature, accelerated melting of glaciers, and deteriorated water resource stability resulted in an increase in the frequency and severity of floods, droughts, and other disasters. The increase of precipitation cannot compensate for the aggravation of water shortage caused by temperature rise in Central Asia. The ecosystem net primary productivity was decreasing over the past years, and the organic carbon pool in the drylands of Central Asia was seriously threatened by climate change. Grassland contributed the most to the increase of ecosystem service values in recent years. Most ecosystem functions decreased between 1995 and 2015, while they are expected to increase in the future (except for water regulation and cultural service/tourism). Global climate change does pose a clear threat to the ecological diversity of Central Asia.

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Keywords Drylands · Land use/land cover change · Ecosystem services · Water resources · Climate change · Ecosystem networks · Aral Sea · Central Asia

7.1 Introduction

Central Asia is vulnerable to global climate change, which has a significant impact on the region's ecological environment and has long been a source of concern and research. Central Asian drylands have experienced significant warming in the last century as a result of global climate change and anthropogenic activities. The drylands also experienced more frequent extreme weather events, such as prolonged drought or flooding, which could exert serious consequences for the arid regions' ecosystem structure and functions.

This section will (1) explain the distribution of drylands in Central Asia; (2) illustrate the climate, soils, land use/land cover types, water resources, ecosystem structure and functions, and social and economic development of Central Asia; (3) examine the land use/land cover change, land degradation and desertification, dynamics of ecosystem structure and functions (including ecosystem productivity and carbon stock), ecosystem services, and human well-being; (4) investigate the driving forces of dryland changes from the perspective of climate change and anthropogenic activities as well as their combination; and (5) elaborate the ecosystem networks and Aral Sea crisis and discuss the conservation and effective practices of drylands in Central Asia. The main objective of this section is to improve the understanding of characteristics, ecosystem dynamics, and driving forces of drylands in Central Asia, an arid and semi-arid area that is extremely sensitive to climate change. Knowledge of dryland changes and driving forces in Central Asia in the context of global climate change and anthropogenic activities is important for environmental protection and improvement as well as sustainable social and economic development.

7.2 Major Characteristics of Dryland SESs in Central Asia

7.2.1 Distribution of Drylands in Central Asia

Central Asia refers to the central part of the Asian Continent. According to the United Nations Educational, Scientific and Cultural Organization (UNESCO), Central Asia includes the vast area between the Altay Mountains in the north, the Himalayas in the south, the Caspian Sea in the west, and the Da Hinggan Ling in the east. Following the UNESCO definition, Central Asia includes Afghanistan, Pakistan, Tajikistan, Turkmenistan, Uzbekistan, Kyrgyzstan, Kazakhstan, the northern part of Iran, the northwestern part of India, the northwestern part of China (Xinjiang Uygur Autonomous Region, Tibet Autonomous Region, Qinghai Province, Inner Mongolia

Autonomous Region, the Hexi Corridor in Gansu Province, and the northern part of Ningxia Hui Autonomous Region), and the southwestern part of the Mongolian People's Republic (Hu 2006).

Former Soviet scholars considered the term Central Asia to refer specifically to the regions where the five Central Asian republics are located (Kazakhstan, Uzbekistan, Kyrgyzstan, Tajikistan, and Turkmenistan). The Soviet Union's official definition of Central Asia was widely used internationally during the Soviet period (Editorial Board of the Silk Road Dictionary 2006). In this section, Central Asia primarily refers to the area of Kazakhstan, Tajikistan, Turkmenistan, Uzbekistan, and Kyrgyzstan.

The geographical location of Central Asia is bounded by longitudes 46°29′47″–87°18′55″E and latitudes 35°07′43″–55°26′28″N (Chen et al. 2014a, b) (Fig. 7.1). In the southeast, the altitude is higher, while in the northwest, it is lower. The Turfan Depression, located on the western side of Central Asia, is home to the Kara Kum Desert and the Kyzyl Kum (Chen and Zhou 2015). Terraces and hills can be found on the northern and northeastern sides of the Turgai region. The majority of the flat land is located between −28 m below sea level and 300 m above sea level.

However, some of the marshy basins in the Karagiye Depression are located at altitudes as low as −132 m below sea level. Aktau Mountain, at 922 m above sea level, is the highest point in the middle of Central Asia. The Tianshan Mountains Range and the Pamir-Alai, which are located on the southeastern side, reach a height of 7,495 m above sea level at their highest point. They are also known as the "water

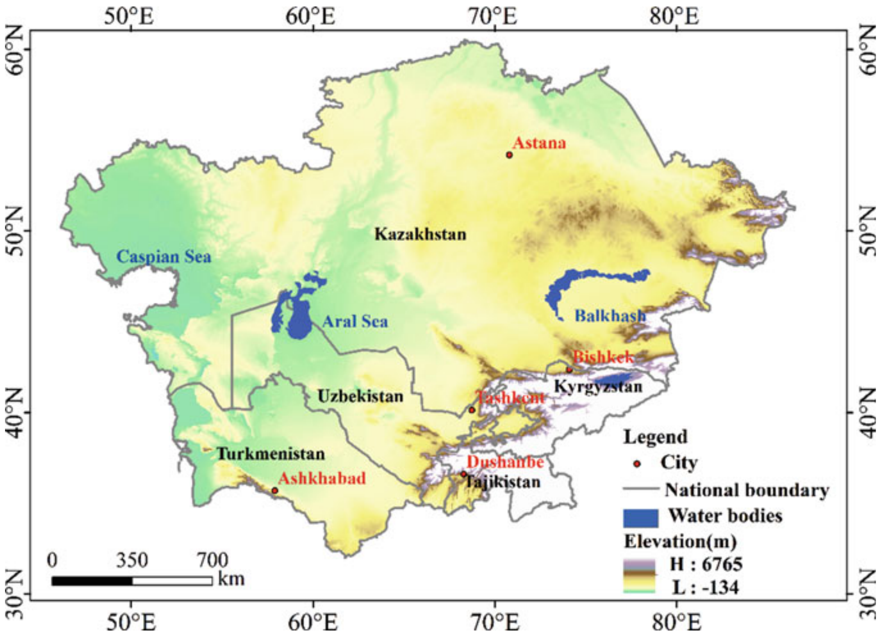

Fig. 7.1 Sketch map of Central Asia

tower" of Central Asia because they provide a vital source of water for rivers and lakes. The Kopet-Dag Range is located on the southwestern side.

With a total area of 4.0×10^8 hm^2 and a total population of around 65 million, Central Asia mainly consists of arid and semi-arid regions (Yang and Du 2013; Yu et al. 2019). Wet air from the Pacific and Indian Oceans is difficult to reach these regions due to the isolation by the Pamirs in Tajikistan, the Tibetan Plateau, and the Tianshan Mountains on the border between China and Kyrgyzstan.

Central Asia is mainly covered by drylands (arid, semi-arid, and semi-humid areas as defined by the United Nations Convention to Combat Desertification), with ecosystems that are extremely sensitive and vulnerable to climate change (UNDP 2005). Due to the continental location of Central Asia, climate and water resources are the key factors in Central Asia's social and economic development (Yu et al. 2019, 2020).

7.2.2 Climate, Soils, Land Use/Land Cover, and Water Resources in Central Asia

Climate

Because of its special geographical location and complex topography, Central Asia is dominated by an arid and semi-arid climate that is primarily controlled by the westerly winds (Chen and Zhou 2015; Lioubimtseva and Cole 2006; Ta et al. 2018). The Atlantic and Arctic Oceans provide the most moisture fluxes to the regions, while moisture fluxes from the Pacific and Indian Oceans are largely obstructed by the Tianshan Mountains and Pamirs (Schiemann et al. 2008). During El Nio, a portion of the moisture fluxes from the Indian Ocean is carried by the westerly wind, which increases precipitation over most area of Central Asia, particularly the middle southern region (Hu et al. 2017; Mariotti 2007).

Precipitation in Central Asia is mainly distributed in the Pamirs and Tianshan Mountains; it has a significant spatial distribution pattern: less precipitation in the western and eastern edge regions, and more precipitation in the central mountainous regions (Chen 2012; Chen et al. 2013). The windward slope of the Pamirs receives 2000 mm of precipitation per year, and the west windward slope of the Tianshan Mountains receives 1000 mm (Balashova et al. 1960; Hu 2004; Yang and Du 2013). Less precipitation is observed on the leeward slopes, valleys, basins, and valleys that are influenced by the mountains (Donat et al. 2016). Therefore, some valleys and basins (such as A Keqi Valley, Caracol) are famous for an arid climate. Precipitation of these basins is lower than 100 mm; in the Issyk-Kul Basin, annual precipitation ranges from 200 to 400 mm, with 399 mm in the northern region and 242 mm in the southern region. Precipitation in winter and spring is significantly higher than that in summer and autumn due to the Mediterranean climate that affects the five Central Asian countries from the southwest to the northeast.

Temperature in Central Asia has the opposite distribution with precipitation. Specifically, precipitation in the plain regions of Uzbekistan, Turkmenistan, and the southern region of Kazakhstan is less than 15 mm, especially in summer, despite average temperature of more than 24.0 °C in these regions. In winter, average temperature in Uzbekistan and Turkmenistan is higher than 0.0 °C, and it is nearly 0.0 °C in the southern region of Kazakhstan. Among the five Central Asian countries, Turkmenistan has the highest temperature (annual mean temperature >15.0 °C), followed by Uzbekistan. Annual mean temperature is lower than −3.0 °C in the western region of Tajikistan and lower than 0.0 °C in the southeastern part of Kyrgyzstan.

Soils

The most important factors affecting soils in Central Asia are rapid evaporation of soil water and lack of water resources. Desert covers two-thirds of the land area, with soils varying significantly from the north to the south and from the west to the east. The soil patterns follow climatic gradients of decreasing precipitation and increasing temperature as they move from the north to the south (Chen et al. 2014b). A comparatively high proportion of sodic soils and saline soils, which are common in alluvial plains, is a distinguishing feature of soil patterns in Central Asia (Chen et al. 2014b).

The landscape here is typical of temperate desert in the world. Soil profiles do not contain any obvious weathered materials, because the weathered products have been removed by erosion. Due to the low vegetation cover, soils have low humus and fulvic acid contents. Central Asia has a great diversity in soil types. According to the 1974 Food and Agriculture Organization (FAO) soil classification, 18 of the 26 soil orders are distributed in Central Asia, covering 244 soil associations. The most important ten soil types in Central Asia are as follows: Orthic Solonetz, Luvic Kastanozem, Haplic Kastanozem, Lithosol, Mollic Gleysol, Luvic Xerosol, Calcic Chernozem, Calcic Xerosol, Eutric Histosol, and Haplic Chernozem (Chen et al. 2014b).

Land Use/Land Cover

Based on the land classification system of the Chinese Academy of Sciences (CAS), land use/land cover in Central Asia can be classified into the following types: cropland, forestland, grassland, wetland, urban land, bare land, and water bodies. In 2015, grassland is the most important land use/land cover type in Central Asia, with a total area of 19,948.14 × 10^4 hm^2, followed by bare land (9,183.36 × 10^4 hm^2), cropland (8,814.59 × 10^4 hm^2), water bodies (1,049.48 × 10^4 hm^2), forestland (800.48 × 10^4 hm^2), wetland (125.25 × 10^4 hm^2), and urban land (89.19 × 10^4 hm^2) (Li et al. 2019).

Water Resources

The five Central Asian countries located in Eurasia's hinterland have various geographical conditions, numerous trans-border rivers, and significant differences in water resource formation and consumption. The problems and contradictions of water resource utilization are very visible, which is representative of the development and utilization of the water resources of trans-border rivers as well as the protection of ecological environment in the world. Most rivers in the five Central Asian countries have no ocean outlet, and the water is diverted for irrigation, lost to the desert, or injected into inland lakes (Deng et al. 2010; Yang et al. 2013).

There are more than 10,000 rivers (such as the Syr Darya River and Amu Darya River) and 10,000 natural lakes (such as the Balkhash Lake and the Aral Sea) in Central Asia (Yu et al. 2019). The Syr Darya River, with the length of 3,019 km, originates from the West Tianshan Mountains in Kyrgyzstan and crosses through Kyrgyzstan, Uzbekistan, Tajikistan, and Kazakhstan (Wang et al. 2021). The Amu Darya River, with the length of 2,540 km and an average annual runoff of 780×10^8 m^3, originates from the Pamirs, Hindu Kush, and Tianshan Mountains before crossing through Tajikistan, Afghanistan, Uzbekistan, and Turkmenistan (Chen et al. 2018). The total lake area in Central Asia is more than 0.50×10^6 km^2, about 1/5 of the Earth's total lake area. The Aral Sea, located between Kazakhstan and Uzbekistan, is the largest tail lake of two inland rivers (the Amu Darya River and Syr Darya River), ranking the fourth largest lake in the world (Deng et al. 2010).

Kyrgyzstan and Tajikistan are primarily located in the upper reaches of trans-border rivers, where there is an abundant supply of water from the mountains (Yu et al. 2019). Water users from Uzbekistan, Turkmenistan, and Kazakhstan frequently complain about a lack of river outflow into their countries in the lower reaches (Fig. 7.2) (Yu et al. 2019). Rivers in Central Asia have two major characteristics. For starters, the seasonal fluctuations in the hydrographs are frequently significant (Yu et al. 2015, 2019). Peak flows are typically observed in summer, while most rivers experience a glacier period in winter. Intermittent streams and channels with an unstable flow are quite common. Second, water volumes are usually low in the downstream with less precipitation, higher evaporation and infiltration losses, and great water consumption (Yu et al. 2019). Temporary flow in rivers has a special biological significance, enabling certain species to breed while eliminating others (Karthe 2018; Yu et al. 2019). The growth of most natural vegetation (also known as Tugai vegetation) in Central Asia's arid and semi-arid regions is highly dependent on groundwater conditions (Yu et al. 2018).

The exploitation and utilization rates of groundwater are relatively low when compared to surface water, and groundwater is primarily used for irrigation and domestic purposes (Liu et al. 2018). Tugai vegetation is highly resistant to dry and saline soils (Thevs et al. 2012). In farmlands, crops often have to grow under water-stressed conditions (Yu et al. 2019).

7.2.3 Ecosystem Structure and Functions

Desert, semi-desert, and steppe are the most common ecosystem types in Central Asia (Zhang et al. 2020). These ecosystems are found throughout the lower mountain slopes and foothills, as well as in some outlying ranges and major basins, covering nearly 75% of Central Asia (Zhang et al. 2020). The gravel desert

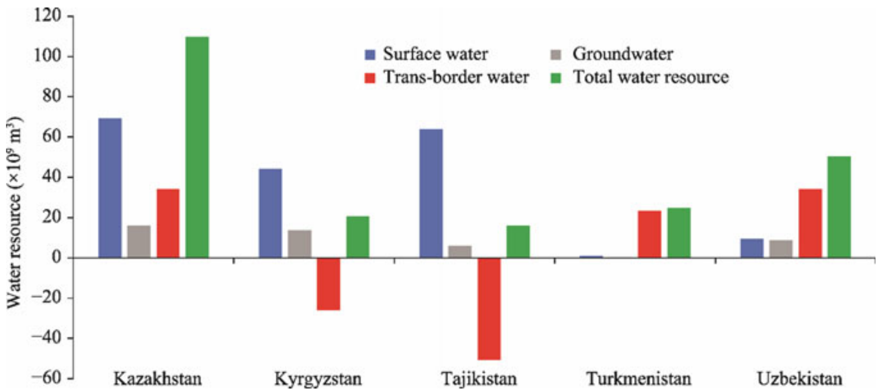

Fig. 7.2 Water resources in five Central Asian countries. The double-counting of the interface between surface water and groundwater was eliminated when calculating the total water resource. The negative value of trans-border water represents the net outflow out of the country. *Source* Yu et al. (2019)

flora comprises more than 400 species, including representatives of the genera *Seriphidium*, *Anabasis*, *Atraphaxis*, and *Caragana*, along with the species *Halolachne songaricum*, *Krascheninnikovia ceratoides*, *Iljinia regelii*, *Salsola gemmascens*, and *Artemisia pectinate* (Zhang et al. 2020). Central Asian deserts are centers of origin and differentiation of ephemeral plants and contain more than 400 such species.

Figure 7.3 depicts the major vegetation types of Central Asia (Zhang et al. 2020). Approximately 7,000 species of vascular plants can be found in Central Asia's mountains, accounting for more than 75% of the region's total plant diversity (Zhang et al. 2013). Dryland ecosystems predominate at lower elevations and in the foothills. Grasslands, shrubs, and forests are common at middle elevations on the mountain slopes. Higher elevations have meadows and tundra-like ecosystems. Spruce and birch forests are mostly found in the Tianshan Mountains, whereas old-growth juniper forests are more common in the Pamir-Alai Mountains. In Central Asia, many mountain and riverside forest ecosystems are legally protected (Zhang et al. 2020).

In Central Asia, various land use/land cover types can provide different ecosystem services as follows: provisioning (food production and raw material), regulating (gas regulation, climate regulation, and water regulation), supporting (soil formation and retention, waste treatment, and biodiversity), and culturing (recreation, culture, and tourism). Based on Li et al. (2019), for the seven land use/land cover types in Central Asia, wetland has the highest ecosystem service value (US$ 25,681/($hm^2 \cdot a$)), followed by water bodies (US$ 12,512/($hm^2 \cdot a$)), urban land (US$ 6,661/($hm^2 \cdot a$)), cropland (US$ 5,567/($hm^2 \cdot a$)), grassland (US$ 4,166/($hm^2 \cdot a$)), and forestland (US$ 3,137/($hm^2 \cdot a$)). It should be noted that the bare land has no ecosystem service value.

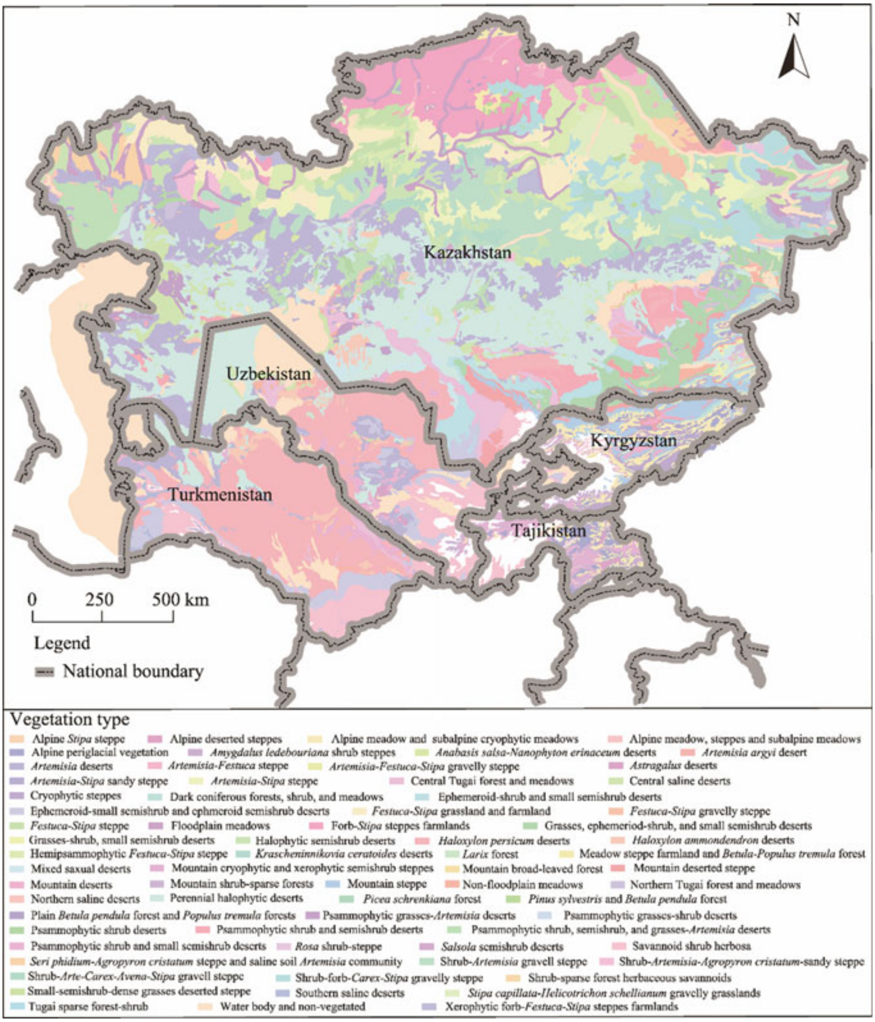

Fig. 7.3 Main vegetation types in Central Asia (Zhang et al. 2020)

7.2.4 Dryland SES Development in Central Asia

Central Asia mainly comprises the countries of Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan. It is a diverse area with a mix of upper-middle and low-income countries with major strategic importance due to their geographical locations and natural resource endowments (https://www.worldbank.org/en/region/eca/brief/central-asia).

The economy in Kazakhstan (as an upper-middle-income country) is primarily comprised of petroleum, natural gas, agriculture, and livestock. From 1991 to 2019, the population increased by 12.54%, rising from 16.45 million in 1991 to 18.51 million in 2019 (see Table 7.1). The rural population rate is decreased from 44% at 1991 to 42% at 2019. The Gross Domestic Product (GDP), increased from US\$ 248.81 $\times$ 10^9 in 1991 to US\$ 1,816.66 $\times$ 10^9 in 2019, with an increase rate of 630%.

The other four countries, i.e., Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan, are low-income countries. From 1991 to 2019, the population in these four countries increased by more than 44.00%, with Tajikistan having the highest increase rate (72.59%) (Table 7.1). Specifically, the population in Tajikistan increased from 5.40 million in 1991 to 9.32 million in 2019, and with the rural population rate 69% at 1991 and 73% at 2019. Uzbekistan had the second highest population growth rate among the four countries, increasing from 20.95 million in 1991 to 33.58 million in 2019 at a rate of 60.27%, but the rural population rate is decreased from 58% at 1991 to 50% at 2019. However, the rural population rate in Kyrgyzstan has tiny variations. For the variations of GDP, Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan had the GDP values of US\$ 25.69 $\times$ 10^9, 25.35 $\times$ 10^9, 32.08 $\times$ 10^9, and 136.78 $\times$ 10^9 in 1991, respectively. In 2019, the values were US\$ 84.55 $\times$ 10^9, 81.17 $\times$ 10^9, 452.31 $\times$ 10^9, and 579.21 $\times$ 10^9, respectively, for the four countries. The corresponding increase rates of GDP were 229.12%, 220.20%, 1,309.94%, and 323.46% from 1991 to 2019 for Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan, respectively (Table 7.1).

Table 7.1 Population, GDP, per capita GDP, and the increasing rate of Kazakhstan (KAZ), Kyrgyzstan (KGZ), Tajikistan (TJK), Turkmenistan (TKM), and Uzbekistan (UZB) in 1991 and 2019

Variable	Year	KAZ	KGZ	TJK	TKM	UZB
Population (million)	1991	16.45	4.46	5.40	3.79	20.95
	2019	18.51	6.46	9.32	5.94	33.58
Increasing rate (%)		12.54	44.66	72.59	56.82	60.27
GDP (US\$ 10^9)	1991	248.81	25.69	25.35	32.08	136.78
	2019	1,816.66	84.55	81.17	452.31	579.21
Increasing rate (%)		630.14	229.12	220.20	1,309.94	323.46
Per capita GDP (US\$)	1991	1512	575	469	846	652
	2019	9812	1309	870	7612	1724
Increasing rate (%)		548.94	127.65	85.50	799.76	164.42
Rural population (million)	1991	7.21	2.79	3.72	2.08	12.19
	2019	7.86	4.09	6.78	2.85	16.64
Rural population rate (%)	1991	44.00	63.00	69.00	55.00	58.00
	2019	42.00	63.00	73.00	48.00	50.00

Note The data are from the https://www.worldbank.org/en/region/eca/brief/central-asia

7.3 Changes of Drylands in Central Asia

7.3.1 Land Use/Land Cover Change

From the 1970s to 2015, the characteristics of land use/land cover change in Kazakhstan are as follows: cropland decreased and was mainly converted into grassland. Water bodies (including wetland) decreased, due to the shrinkage of the Aral Sea. Grassland increased significantly from the 1970s to 2015, which was primarily resulted from the abandonment of cropland and frequent conversion of cropland and grassland in northern Kazakhstan.

The following is the characteristics of land use/land cover change in Uzbekistan:

(1) Cropland expansion was primarily due to the conversion of grassland or sparse shrubs.
(2) The shrinkage of the Aral Sea was responsible for the decrease in water area.
(3) The majority of the water bodies were converted into bare land or sparse vegetation, swamp wetland, and grassland.
(4) The conversion of cropland has led to a significant increase in urban land.

In Turkmenistan, land use/land cover change showed a significant increase in cropland, mainly from the conversion of grassland or sparse vegetation. The increase in water bodies was mainly caused by bare or sparse vegetation and a small amount of grassland. A significant increase in urban land area was mainly due to the conversion of cropland and grassland.

Land use/land cover change in Kyrgyzstan was characterized by: (1) a reduction in cropland, mainly converted into grassland; (2) a significant decrease in water area because of the shrinkage of glaciers and the sharp reduction of permanent snow cover area; and (3) an increase in urban land area, primarily from the conversion of cropland and grassland. In Tajikistan, the increase of farmland was from the conversion of grassland. The significant reduction of water area was resulted from the shrinkage of glaciers and the sharp decrease of permanent snow cover area, while the increase in urban land area was mainly resulted from the conversion of cropland and grassland.

Cropland change, forestry activities, and grazing were the main land use/land cover changes and land management in Central Asia, each of which had different impacts on the ecosystem structures and services in Central Asia. The major land use/land cover change in Central Asia from the 1970s to 2015 can be summarized as follows: (1) continued desiccation of the Aral Sea; (2) continuous increase of urban land area; and (3) significant increase in grassland from the conversion of cropland in Kazakhstan and Kyrgyzstan and significant increase of cropland from the conversion of grassland in Uzbekistan, Turkmenistan, and Tajikistan. Rapid urbanization also caused the proportion of urban land to increase from 27.57×10^4 hm^2 in 1995 to 60.21×10^4 hm^2 in 2005 and then to 89.19×10^4 hm^2 in 2015, with an average growth rate of 10.64%/a (Fig. 7.4) (Li et al. 2019).

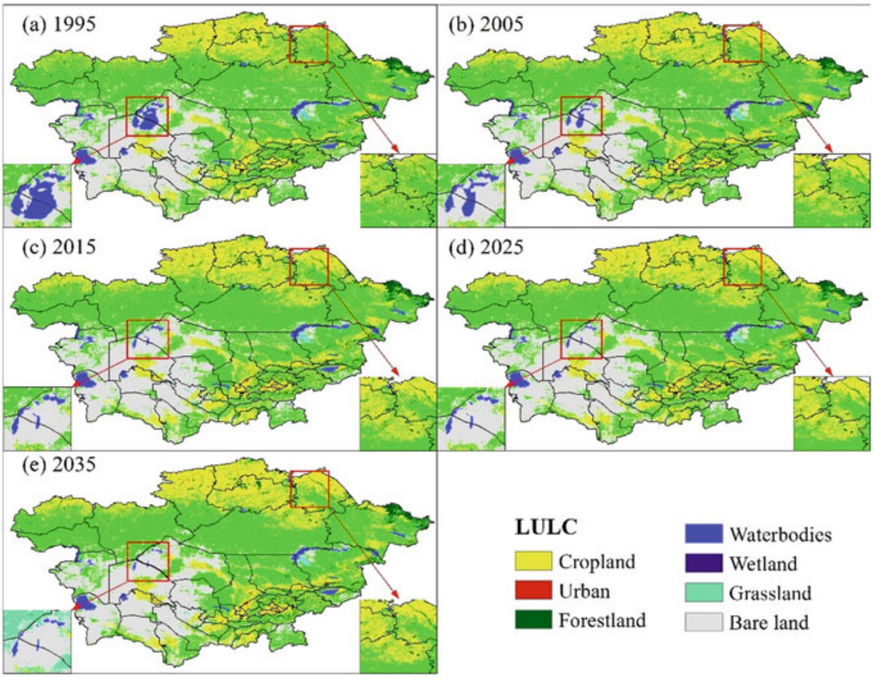

Fig. 7.4 Spatial distribution of land use/land cover (LULC) in Central Asia in **a** 1995, **b** 2005, **c** 2015, **d** 2025, and **e** 2035. *Source* Li et al. (2019); https://doi.org/10.7717/peerj.7665/fig-2

7.3.2 Land Degradation and Desertification

The main regions affected by land degradation were concentrated in the north of Kazakhstan, and extended across the eastern Kazakhstan of the southern part of Central Asia, covering Kyrgyzstan, the northwest of Tajikistan, and the southern parts of Uzbekistan and Turkmenistan (Mirzabaev et al. 2016). Mirzabaev et al. (2016) also estimated that the annual cost of land degradation in Central Asia due to land use/land cover change was about US$ 6.00 billion, most of which can be attributed to grassland degradation (US$ 4.60 billion), followed by desertification (US$ 0.80 billion), deforestation (US$ 0.30 billion), and the abandonment of cropland (US$ 0.10 billion). Further, the costs of action against land degradation were found to equal about US$ 53.00 billion over a 30-year period, whereas if nothing is done, the resulting losses may equal to almost US$ 288.00 billion during the same period (Mirzabaev et al. 2016).

From 2000 to 2014, the area of desertification land in Central Asia increased; the increasing magnitude of desertification land was 98,912.26 km^2 and the growth rate was 0.11% (Liu et al. 2017). Among them, 30,889.73 km^2 land was transferred from non-desertification to desertification, which was larger than the land transferred from desertification to non-desertification (266,497.67 km^2). In terms of spatial distribution, desertification land in Central Asia gradually decreased from severe desertification in the southwest to mild desertification in the northeast and continued to move toward the northern part of Kazakhstan (Liu et al. 2017).

Kazakhstan is a country with a large desert area and severe desertification. According to the Kazakhstan's "Second National Report on the Implementation of the United Nations Convention to Combat Desertification" issued in 2002, desertification affected two-thirds of the country's land to varying degrees.

From 1995 to 2001, the total area of desertification land in Kyrgyzstan increased by 0.41×10^5 km^2, an average annual increase of 590.00 km^2/a. The proportion of both severe and very severe desertification types in the country decreased in 2001 compared with 1995, while the land area of mild and moderate desertification increased.

Degradation of vegetation is the main cause of desertification in Turkmenistan. The desert vegetation covers most of the land in Turkmenistan and plays an important role in environmental protection. Desertification caused by vegetation degradation covered an area of 367,522.00 km^2. Under extreme desert conditions, trees can prevent soil erosion by winds and water, serve as fodders and fuels drain water ecologically, and help to protect settlements from dry winds and dust storms.

7.3.3 Dynamics of Ecosystem Structure and Functions

Changes in Ecosystem Productivity

Net primary productivity (NPP), as an important indicator of ecological health, has been widely used in studies of the effects of climate change on ecosystem functions (Zhang and Ren 2017). In the context of climate change, it is necessary to understand the temporal and spatial characteristics of NPP in Central Asia based on various climatic factors. Such knowledge is important for developing effective climate adaptation strategies in Central Asia.

The average annual NPP in Central Asia was 1125 (±129) Tg C (1 T = 10^{12}) or 218 (±25) g C/m^2. The annual NPP value was higher in the northern part of Kazakhstan 349 (±39) g C/m^2. In terms of vegetation types, the NPP value was the highest (556 (±82) g C/m^2) in the temperate coniferous forest, while the non-deep root shrub had the lowest NPP value (158 (±25) g C/m^2) (Zhang and Ren 2017). Climate change in Central Asia affected the temporal and spatial patterns of NPP, GPP (gross primary productivity), and RA (autotrophic respiration) (Figs. 7.5 and 7.6). From 1980 to 2014, the annual NPP in Central Asia showed a decreasing trend of 0.82 g C/(m^2·a), with high interannual variability (Zhang and Ren 2017). The changes in NPP were relatively stable during the period of 1980–1997 and more variable during the period of 1998–2008, with the lowest values found in 2001 2006, and 2008 when major La Niña events took place.

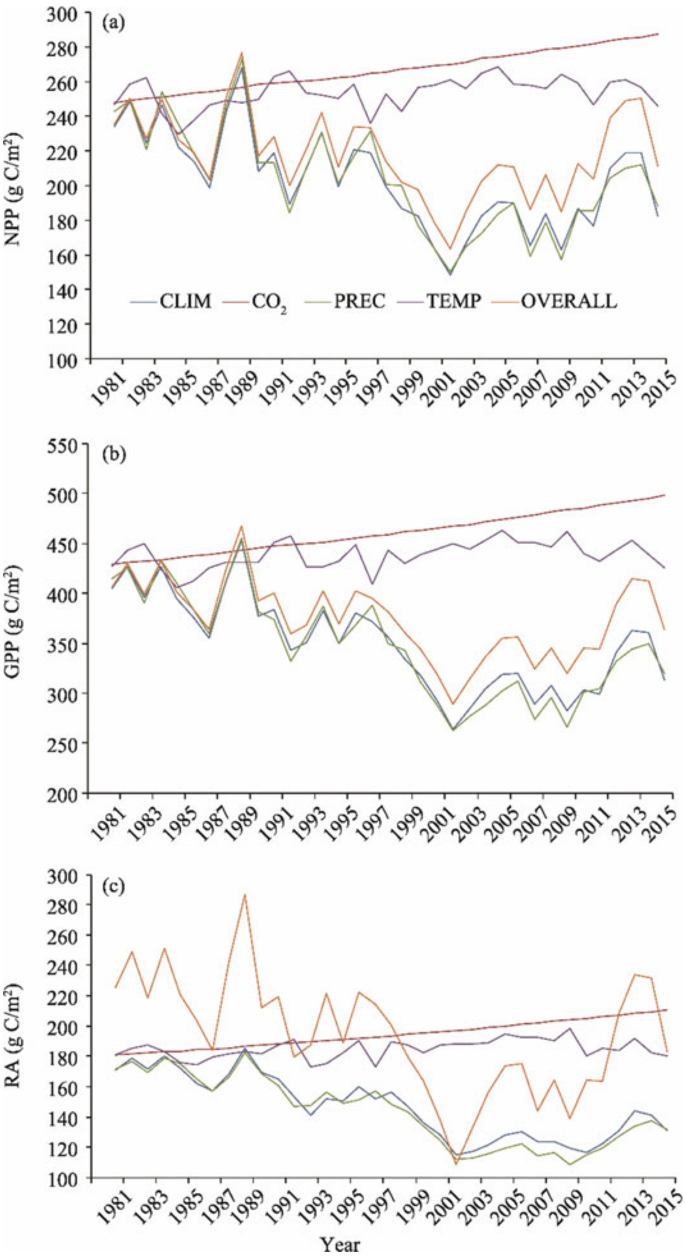

Fig. 7.5 Temporal patterns of **a** NPP, **b** GPP, and **c** RA from 1980 to 2014 in response to climate change factors. NPP, net primary productivity; GPP, gross primary productivity; RA, autotrophic respiration; CLIM, climate change effect; CO_2, CO_2 fertilization effect; PREC, precipitation change effect; TEMP, temperature change effect; OVERALL, combined effects of climate and CO_2 changes

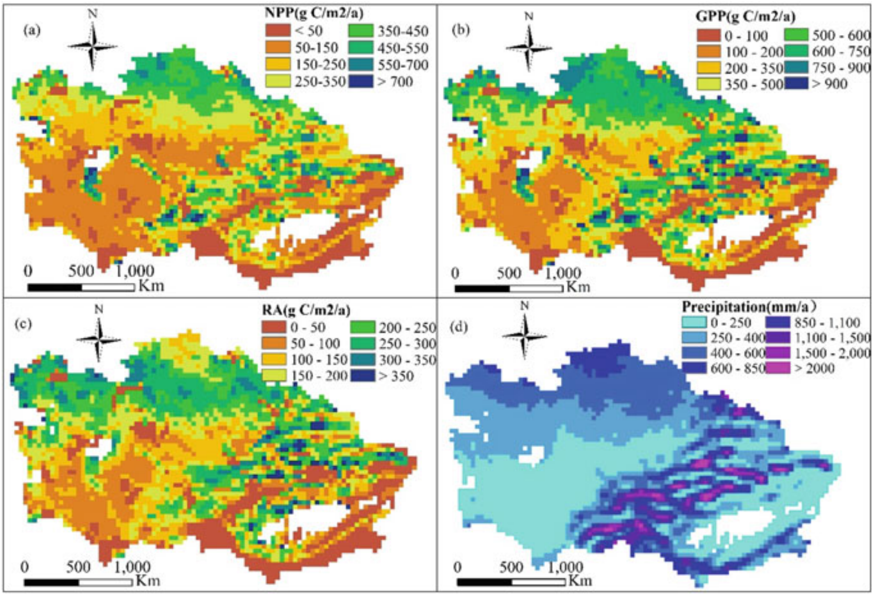

Fig. 7.6 Spatial patterns of **a** NPP, **b** GPP, **c** RA, and **d** precipitation from 1980 to 2014

Compared with the average annual NPP from 1980 to 1984, the total NPP in Central Asia decreased by 118 Tg (–10%) between 1985 and 2014 (Zhang and Ren 2017). Temperature was the main factor controlling NPP in 9% of Central Asia, which was primarily distributed in high-latitude alpine regions such as the Tianshan Mountains; precipitation was the main factor impacting NPP in 69% of Central Asia, which was mainly distributed in desert plains; CO_2 was the dominant factor influencing NPP in 20% of Central Asia, which was distributed in areas with good hydrothermal conditions, such as forest areas and low-altitude areas (Zhang and Ren 2017).

Changes of Carbon Stock

Central Asia is one of the largest drylands in the mid-latitude, containing over 80% of the world's temperate desert. Central Asia has become one of the most uncertain regions in global carbon cycle research (Lioubimtseva and Henebry 2009). The carbon stock of Central Asia was approximately 31.34–34.16 Pg, with 10.42–11.43 Pg stored in the deep soils (1–3 m) of its temperate desert, amounting to 24% of the global carbon stock in desert and dry shrubland (Li et al. 2015). The ecosystem carbon density (6.6–7.3 kg/m^2 in 1 m soil depth) in Central Asia was very close to that in Australia (7.1 (±1.4) kg/m^2). In Central Asia, soils stored approximately 90% of carbon stock, which was significantly higher than that in Australia (55%) and most other regions of the world. Kazakhstan had the largest carbon stock among the five Central Asian countries, accounting for more than 70% of the total carbon stock, followed by Kyrgyzstan and Tajikistan.

The mean vegetation carbon density in Central Asia was low, approximately 0.65 and 0.88 kg C/m^2 from the inventory method and arid ecosystem modeling, respectively (Li et al. 2015). This was mainly due to the low vegetation carbon density in grassland (0.40–0.50 kg C/m^2) and temperate desert (0.40–0.87 kg C/m^2), which together covered 82% of the land area in Central Asia. Vegetation carbon density was highest in the evergreen needle leaf forests, approximately 13.85 and 18.13 kg C/m^2 from the inventory method and arid ecosystem modeling, respectively (Li et al. 2015).

The mean soil organic carbon density in the top 1 m of soil was 5.81 and 6.59 kg C/m^2 from the arid ecosystem modeling and inventory method, respectively (Li et al. 2015). When the deep-soil carbon storage was considered, the regional mean soil organic carbon (SOC) density was 8.25 and 8.81 kg C/m^2 based on the arid ecosystem modeling and inventory method, respectively (Li et al. 2015). In this case, the lowest SOC was in grassland, approximately 5.52 and 6.84 kg C/m^2 based on the arid ecosystem modeling and inventory method, respectively (Li et al. 2015). The highest level of SOC was found in evergreen needle leaf forests, approximately 32.60 and 42.99 kg C/m^2 from the arid ecosystem modeling and inventory method, respectively (Li et al. 2015). The temperate desert and grassland together contributed to 51–60% of the regional vegetation carbon stock and 77–79% of the SOC stock (deep soil) because of their large coverage (82%) in Central Asia (Li et al. 2015).

Climate change posed a serious threat to the organic carbon pool in Central Asian drylands, which lost approximately 0.46 Pg C between 1979 and 2011 (Li et al. 2015). The long-term drought in northern Kazakhstan was the primary cause of the loss of regional carbon stock. The drought was closely related to the La Nia phenomenon, which has resulted in an 8% decrease in the vegetation carbon pool, mainly in northern Kazakhstan. Central Asia can be further divided into semi-arid grassland areas in northern Kazakhstan, arid desert in Central Asia, and Tianshan cold and wet forest-meadow area to analyze the carbon stock changes of ecosystems under different climatic zones (Fig. 7.7). Because of the persistent drought between 1998 and 2008, the semi-arid grassland of northern Kazakhstan has become the largest carbon source in Central Asia.

Climate change exhibited the greatest impact on the central arid desert shrub area. In contrast, the carbon pools in the Tianshan cold and wet forest-meadow area remained relatively stable in response to climate change, implying that forest ecosystems in Central Asian have a strong buffer capacity against climate change.

7.3.4 Changes in Ecosystem Service Values and Human Well-Being

Ecosystem services include regulating services (such as water regulation, climate regulation, and gas regulation), provisioning services (such as food and raw materials), supporting services (soil formation, waste treatment, and biodiversity), and

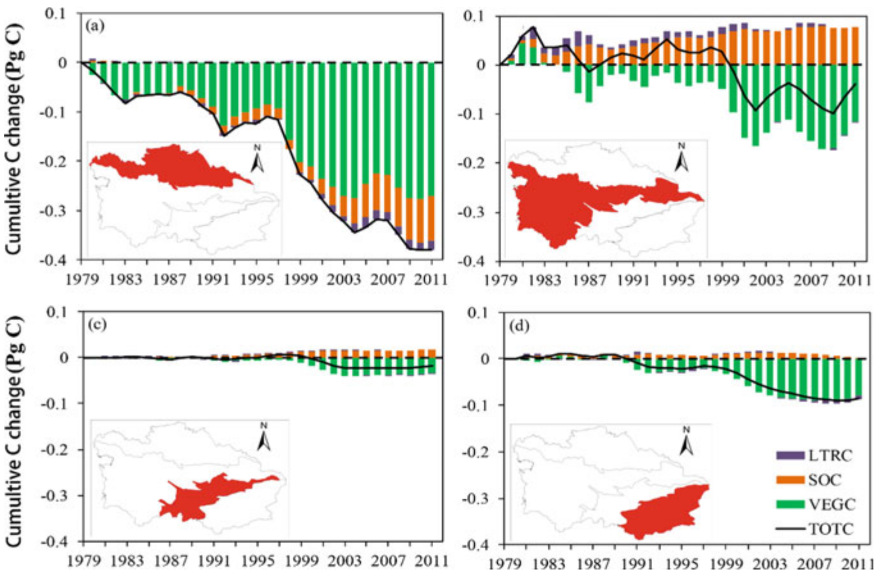

Fig. 7.7 Ecosystem carbon pool (TOTC), vegetation carbon pool (VEGC), soil organic carbon pool (SOC), and liter carbon pool (LTRC) changes in **a** northern Kazakhstan, **b** arid desert, **c** Tianshan cold and wet forest-meadow area, and **d** Xinjiang of China during the period of 1979–2011

culturing services (such as recreation, culture, and tourism) (Hassan et al. 2005; Li et al. 2019). Land use/land cover change can alter the ecosystem structures and functions and influence the ecosystem services (Hu et al. 2008; Li et al. 2019; Yirsaw et al. 2017).

Acute farmland expansion and rapid urbanization in Central Asia have accelerated land use/land cover change, which had substantial effects on ecosystem services. Evaluating changes in ESV in response to land use/land cover change and exploring the elasticity of the response of ecosystem service value to land use/land cover change could provide policy makers with important references for ecological environmental protection and the sustainable development of Central Asia.

The total ecosystem service value in Central Asia was approximately US$ 1505.31 billion in 1995 (Table 7.2) (Li et al. 2019). Grassland had the highest contribution the ecosystem service value (56.90%), followed by cropland and water bodies (28.09% and 11.15%, respectively) (Fig. 7.8) (Li et al. 2019). The total ecosystem service value increased by US$ 5.68 billion from 1995 to 2005, mainly due to the increased ecosystem service values of cropland and urban land. The total ecosystem service value increased by US$ 5.23 billion from 2005 to 2015 (Li et al. 2019). Overall, the total ecosystem service value in Central Asia increased by US$ 10.91 billion during the period of 1995–2015 (Li et al. 2019). It is noteworthy that the proportion of water bodies decreased sharply by 21.80% from 1995 to 2015, resulting in a loss of US$ 36.61 billion (Li et al. 2019). These trends were expected to continue to occur in 2025 and 2035 (Table 7.2) (Li et al. 2019). Land use/land cover change is

Table 7.2 Ecosystem service values in Central Asia from 1995 to 2035 (Li et al. 2019)

Land use/land cover	Ecosystem service value (US$ billion)					Change (%)		
	1995	2005	2015	2025	2035	1995–2015	2015–2035	1995–2035
Cropland	422.79	478.27	490.68	503.87	516.24	16.06	5.21	22.10
Forestland	25.12	25.13	25.11	25.08	25.05	−0.05	−0.26	−0.31
Grassland	856.54	825.31	831.02	836.40	841.66	−2.98	1.28	−1.74
Wetland	31.11	31.25	32.17	32.47	32.98	3.39	2.52	6.00
Urban land	1.84	4.01	5.94	6.45	7.76	223.41	30.59	322.33
Bare land	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Water bodies	167.92	147.02	131.31	117.51	103.54	−21.80	−21.14	−38.34
Total	1,505.31	1,510.99	1,516.23	1,521.78	1,527.22	0.73	0.73	

Note The Table is from Li et al. (2019); https://doi.org/10.7717/peerj.7665/table-3

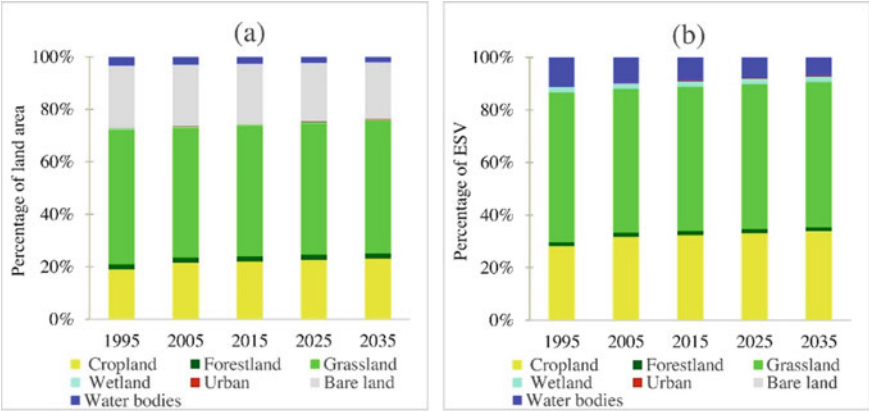

Fig. 7.8 Percentage of land use/land cover area (**a**) and percentage of ecosystem service value (ESV) of different land use/land cover types (**b**) from 1995 to 2035. *Source* Li et al. (2019); https://doi.org/10.7717/peerj.7665/fig-3

correlated with other global processes such as climate change and land degradation, which directly or indirectly affect local ecosystem services. Ecosystem service value in Karakalpakistan, Uzbekistan, decreased by more than 50% during 1995–2035, which was mainly caused by the shrinkage of the Aral Sea (Li et al. 2019).

Table 7.3 shows the changes in individual ecosystem functions (Li et al. 2019). The most important ecosystem functions in Central Asia were biodiversity, food production, and water regulation, contributing 40.44%, 28.30%, and 11.78% of the total ecosystem service value in 1995, respectively, 40.03%, 29.47%, and 10.21% of the total ecosystem service value in 2015, and 40.51%, 30.14% and 8.93% of the total ecosystem service value in 2035 (Li et al. 2019). Most of the ecosystem functions decreased during the period of 1995–2015 except for food production, raw materials, climate regulation, soil formation, and waste treatment, which increased by 4.87%, 7.92%, 12.11%, 12.01%, and 2.91%, respectively (Fig. 7.9) (Li et al. 2019). It is noteworthy that the ecosystem service value of water regulation declined more rapidly than other ecosystem services (−12.70%), followed by gas regulation (−3.00%), culture and tourism (−3.14%), and biodiversity (−0.29%). However, most of the ecosystem functions were expected to increase from 2015 to 2035 (Fig. 7.9) (Li et al. 2019). It should be noted that only the ecosystem service values of water regulation and culture/tourism were expected to decrease in the future (Li et al. 2019).

Table 7.3 Estimated values for different ecosystem functions in Central Asia from 1995 to 2035 (Li et al. 2019)

Service type	Sub-type	Ecosystem service value (US$ billion)				
		1995	2005	2015	2025	2035
Provisioning	Food production	426.08	440.12	446.82	453.74	460.31
	Raw materials	29.85	31.64	32.22	32.80	33.37
Regulating	Gas regulation	1.87	1.80	1.81	1.82	1.83
	Climate regulation	45.09	49.20	50.56	51.59	52.90
	Water regulation	177.36	165.34	154.83	145.58	136.33
Supporting	Soil formation and retention	63.98	70.02	71.66	73.23	74.84
	Waste treatment	62.49	64.37	64.31	64.33	64.38
	Biodiversity	609.01	601.26	607.24	613.05	618.73
Culturing	Recreation, culture, tourism	89.59	87.25	86.78	84.92	84.52
Total		1,505.32	1,510.99	1,516.23	1,521.78	1,527.23

Note The table was from Li et al. (2019); https://doi.org/10.7717/peerj.7665/table-4

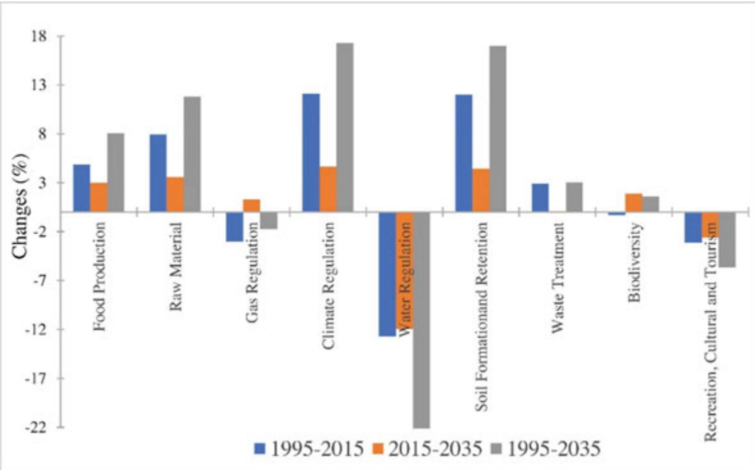

Fig. 7.9 Changes of ecosystem service functions in Central Asia from 1995 to 2035. *Source* Li et al. (2019); https://doi.org/10.7717/peerj.7665/fig-5

7.4 Driving Forces of Dryland Changes

7.4.1 Climate Change and Extreme Events

Based on the most recent CRU dataset, Central Asia has experienced a significant increase in temperature, at a rate of 0.38 °C/decade from 1991 to 2019 (Fig. 7.10a). It had a slight increase in annual precipitation, at a rate of 0.52 mm/a (Fig. 7.10b). With rising temperature and increased precipitation, potential evapotranspiration has increased significantly, reaching 0.85 mm/a (Fig. 7.10c).

For the seasonal variations, temperature showed significantly increasing trends in spring (March, April, and May) and summer (June, July, and August), with the increasing trends of 0.87 °C/decade and 0.49 °C/decade, respectively (Table 7.4). In autumn (September, October, and November), it exhibited a weak increasing trend. A weak negative trend of temperature was observed in winter (December, January, and February). For precipitation, positive trends were observed in spring and autumn, with the rates of 0.36 mm/a and 0.43 mm/a, respectively. While precipitation in summer and winter exerted negative trends, with the rates of −0.21 mm/a and −0.04 mm/a, respectively. The potential evapotranspiration in Central Asia showed significant positive trends in spring (0.37 mm/a) and summer (0.49 mm/a) during the period of 1991–2019 (Table 7.4). There was no discernible linear trend for the remaining two seasons (autumn and winter). The significant increases of temperature and precipitation in spring could result in spring flooding in Central Asia, especially in mountainous areas (Hu et al. 2014, 2016, 2017, 2019a, b).

Recent studies mainly focused on the extreme climate events in Central Asia using observed records and climate model results (CMIP 5: Coupled Model Intercomparison Project 5) (Liu et al. 2021; Peng et al. 2020; Yao et al. 2021). is the following findings were discovered (i) significant wetting and warming trends occurred in Central Asia during the period of 1881–2018, with 42.5%, 59.4%, and 79.2% of stations having change points for extreme precipitation, maximum temperature, and minimum temperature, respectively; (ii) the occurrences of extreme climate events showed spatial heterogeneity, with the highest risks of meteorological drought, flood, and frost events occurring in the southwest, southeast, and northeast of Central Asia, respectively; and (iii) global warming significantly affected the intensities and frequencies of extreme precipitation and temperature, and their univariate and multivariate risks were intensified in most regions of Central Asia (Liu et al. 2021; Peng et al. 2020; Yao et al. 2021).

7.4.2 Anthropogenic Activities

Variations in the cultivated land area (arable land and permanent cropland) always have significant effects on water consumption and withdrawal, particularly in arid regions. Another significant anthropogenic activity influencing dryland changes in

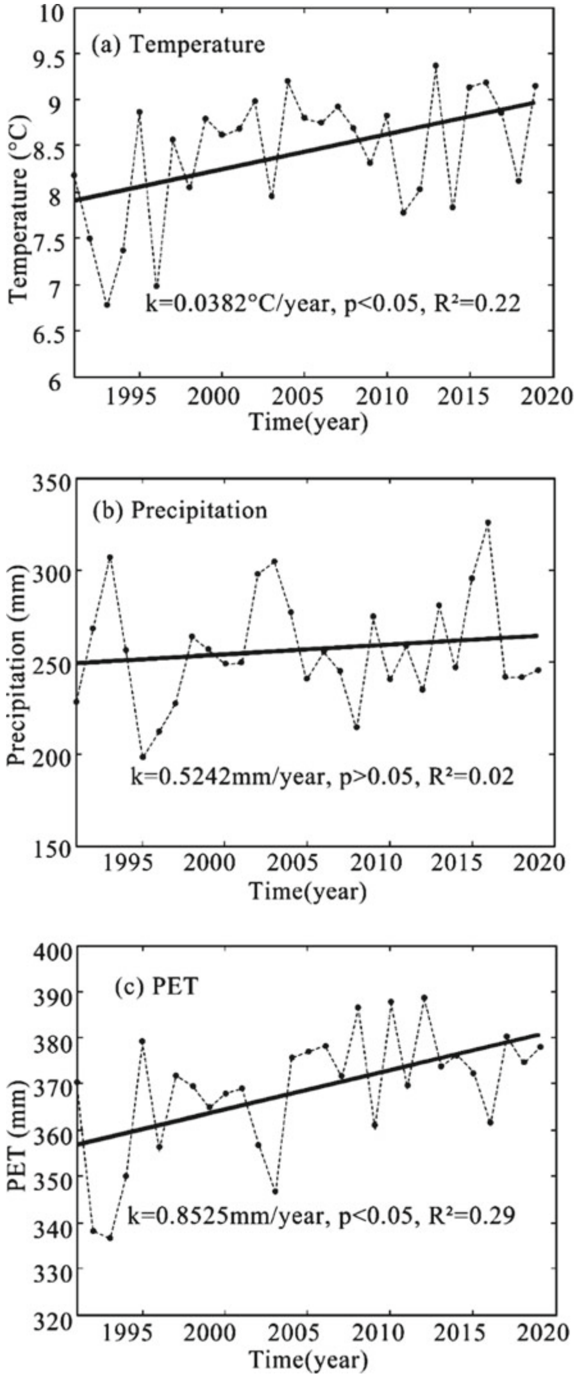

Fig. 7.10 Variations of **a** annual temperature, **b** annual precipitation, and **c** PET in Central Asia from 1991 to 2019. PET: potential evapotranspiration

Table 7.4 Seasonal linear trends of the temperature, precipitation, and PET in Central Asia from 1991 to 2019

Season	Temperature (°C/decade)	Precipitation (mm/a)	PET (mm/a)
Spring	0.87	0.36	0.37
Summer	0.49	−0.21	0.49
Autumn	0.12	0.43	0.01
Winter	−0.004	−0.04	−0.01

Note Spring: March, April, and May; Summer: June, July, and August; Autumn: September, October, and November; Winter: December, January, and February

the five Central Asian countries is livestock grazing. In this sub-section, anthropogenic activities in Central Asia were examined through the lens of two factors: cultivated land area variation and livestock grazing.

The cultivated land area of Kazakhstan increased from 2,847.00 × 10^4 hm^2 in 2002 to 2,953.00 × 10^4 hm^2 in 2014 with an increasing rate of 8.83 × 10^4 hm^2/a. For Uzbekistan, the cultivated land area decreased from 483.00 × 10^4 hm^2 in 2002 to 469.00 × 10^4 hm^2 in 2007 and then increased to 477.00 × 10^4 hm^2 in 2014. The cultivated land area of Kyrgyzstan decreased from 141.10 × 10^4 hm^2 in 2002 to 135.60 × 10^4 hm^2 in 2014. A weak decrease of the cultivated land area was observed in Tajikistan, from 88.10 × 10^4 hm^2 in 2002 to 87.00 × 10^4 hm^2 in 2014. For Turkmenistan, the cultivated land area decreased from 210.00 × 10^4 hm^2 in 2002 to 200.00 × 10^4 hm^2 in 2014 (http://www.fao.org/nr/water/aquastat/data/query/index.html?lang=en). The results demonstrated that Kazakhstan had the largest cultivated land area, followed by Uzbekistan and Turkmenistan.

Kazakhstan had a very small percentage of the cultivated land area in terms of irrigated land area, and the cultivated land area of the other four countries was nearly equal to the irrigated land area. For example, Uzbekistan exhibited the largest irrigated land area with the value of 430.00 × 10^4 hm^2 in 2008, followed by Kazakhstan (188.70 × 10^4 hm^2) and Turkmenistan (185.00 × 10^4 hm^2). Tajikistan had the smallest irrigated land area, with the value of 71.00 × 10^4 hm^2 in 2008 (Deng et al. 2010).

Grazing can result in rapid changes in ecosystem states that affect carbon stock (Grace 2004; Hobbs and Norton 1996). It can reduce the growth, survival, and fitness of most grazed plants, as well as the above-ground carbon stock (Tanentzap and Coomes 2012). The Gridded Livestock of the World (GLW) database of the Food and Agriculture Organization's Animal Production and Health Division (FAO-AGA) can be used to obtain the grazing intensity data (Wint and Robinson 2007).

The data for cattle, buffalo, sheep, and goats were created in the ESRI grid format with a spatial resolution of 3 min of arc (roughly 5 km at the equator), which were freely available for download from FAO's GeoNetwork data repository. Based on the Biome-BGC grazing model (Han et al. 2014), grazing resulted in a total carbon loss of 1,985 Tg C in grassland of Central Asia from 1979 to 2015. During the former Soviet Union period (1979–1991), 1,456 Tg C was emitted from grazing (an average annual emission intensity of 64 Tg C), showing an obvious strong carbon source process. Since the disintegration of the former Soviet Union (1992–2015), 529 Tg C has been emitted from grazing. The emission intensity was greatly reduced, with an average annual emission intensity of 22 Tg C. Therefore, grazing was converted into a process of weak carbon source.

7.4.3 Interactions Among Different Drivers

It has been widely observed that Central Asian oases have lower temperatures than the surrounding deserts, owing to evaporative cooling caused by plant transpiration and irrigation (Kai et al. 1997). The interactions among different drivers in Central Asia are demonstrated from the following aspects: (1) the influences of irrigation on temperature change and terrestrial water storage; and (2) the impact of urbanization on temperature change. To investigate the effects of irrigation on temperature change, the United Nations FAO global map of irrigated land was used to identify all meteorological stations that were located within 5 km of irrigated land. They were then assigned to the nearest non-irrigated land stations (Fig. 7.11). The result showed that there was no significant positive effect from the de-intensification of agriculture following the collapse of the former Soviet Union in the early 1990s on the observed temperature increase in Central Asia (Hu et al. 2014).

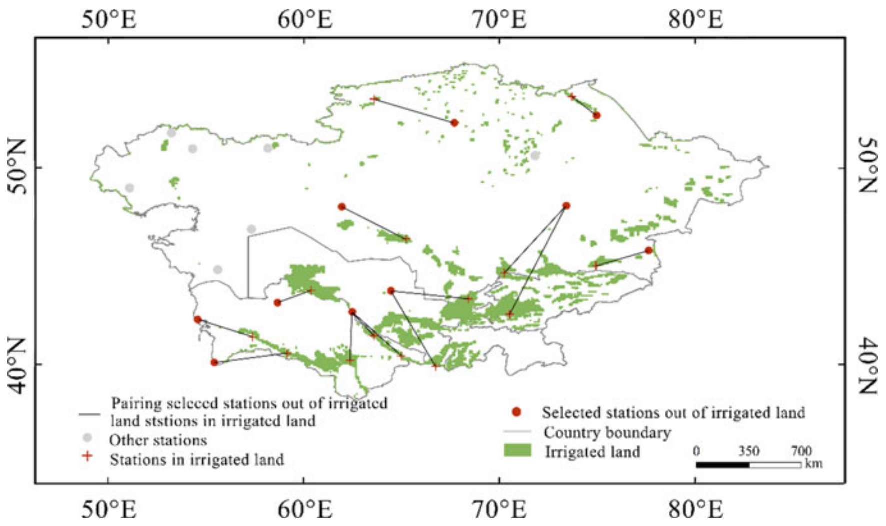

Fig. 7.11 Pairing of meteorological stations located in the irrigated land with the closest stations outside the irrigated land (Hu et al. 2014). © American Meteorological Society. Used with permission

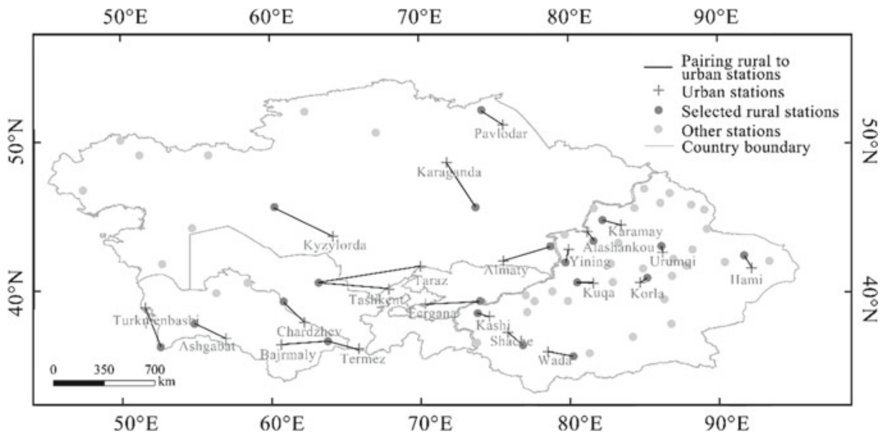

Fig. 7.12 Pairs of meteorological stations located in the urban areas and the closest rural stations (Hu et al. 2014). © American Meteorological Society. Used with permission

A similar approach was used to investigate whether urbanization in Central Asia may have affected the observed temperature change. Paired *t*-test showed no significant differences in temperature change rates between the urban and rural meteorological stations ($P > 0.05$). These test results suggested no significant effect from urbanization on the observed temperature change in Central Asia (Fig. 7.12) (Hu et al. 2014).

To investigate the impact of irrigation on terrestrial water storage, Central Asia was classified into irrigated regions and non-irrigated regions (Hu et al. 2019a). Figure 7.13a shows the irrigated regions in Central Asia and Northwest China. The data were extracted from the Global Map of Irrigation Areas (GMIA) V 5.0 of the FAO (http://www.fao.org/nr/water/aquastat/irrigationmap/index10.stm). Based on the Student's *t*-test at the 95% significance level, the annual terrestrial water storage anomaly of the irrigated grids was compared to the averaged terrestrial water storage anomaly of the non-irrigated grids surrounding them. The comparison excluded any irrigated grids that did not have any surrounding non-irrigated grids. Approximately 81% of the irrigated grids showed no significant difference when compared to the non-irrigated grids surrounding them (Fig. 7.13b) (Hu et al. 2021).

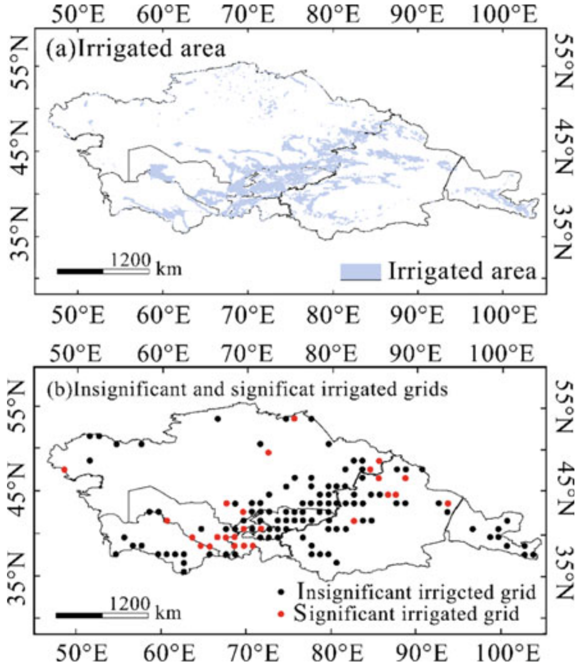

Fig. 7.13 Comparison between the irrigated grids and the matched non-irrigated grids based on the annual CSR dataset. **a** Irrigated area, and **b** insignificant irrigated grids and significant irrigated grid. Results of Student's *t*-test was used to test annual CSR between irrigated grids and their surrounding non-irrigated grids. The red dots represent irrigated grids with significantly different terrestrial water storage anomaly values from their surrounding non-irrigated grids, and the black dots represent that their differences are insignificant. *Source* Hu et al. (2021), with permission from Copyright Clearance Center's RightsLink®, Order Number: 5216920963705)

7.5 Ecosystem Management of Central Asia

7.5.1 Ecosystem Networks in Central Asia

The CAS Research Center for Ecology and Environment of Central Asia (RCEECA) was launched in 2013 by the Developing Countries Science and Education Cooperation Programme of the CAS. The RCEECA is based on the Xinjiang Institute of Ecology and Geography (CAS), and it is co-founded by the Institute of Tibetan Plateau Research (CAS), Institute of Earth Environment (CAS), Northwest Institute of Eco-Environment and Resources (CAS), Institute of Geographic Sciences and Natural Resources Research (CAS), Nanjing Institute of Geography and Limnology (CAS), Institute of Remote Sensing and Digital Earth (CAS), Shenzhen Institute of Advanced Technology (CAS), and the University of CAS.

Faced with the strategic demand of the Shanghai Cooperation Organization and the construction of the Belt and Road Initiative for scientific and technological cooperation, the RCEECA established research centers and overseas sub-centers to carry out joint scientific and technological research on sustainable development of natural resources and environmental protection between China and Central Asian countries.

The research center focuses on mutually beneficial cooperation research in climate and environmental changes, mineral resources, and water and soil resources, modern agricultural and biological resources, geoeconomic and regional development, ecological restoration and environmental governance, transportation, and information technology. In addition, it seeks to develop professional research teams both home and abroad for long-term scientific and technological cooperation in Central Asia. The establishment of an ecological and environmental research platform with field observation, indoor basic experimental analysis, satellite remote sensing monitoring, and technical demonstration will provide scientific and technological support as well as the basic platform for the implementation of resources and environment development strategy of the Shanghai Cooperation Organization and the Belt and Road Initiative.

Currently, the RCEECA operates three overseas branches (Almaty, Bishkek, and Dushanbe), several information centers and analytical testing laboratories, and 19 field observation and research stations that cover the entire Central Asian area. The 19 field observation and research stations covering the landforms of glacier, mountain, forest, desert, oasis, farmland, wetland, grassland, and other ecosystems were established in Kazakhstan, Kyrgyzstan, Tajikistan, Uzbekistan Mongolia, and Iran based on the typical landscape characteristics of different regions in Central Asia (Table 7.5; Fig. 7.14). The RCEECA has spent more than 3,000 million CNY on more than 60 sets of experimental analysis equipment and positioning observation instruments.

Kazakhstan has a vast territory, containing almost all ecosystem types in Central Asia. Six field observation and research stations have been established in Kazakhstan, focusing on desert, oasis, farmland, grassland, and woodland ecosystems. Tajikistan has constructed one mountain research station, one grassland/farmland research station, and one plateau research station in Pamirs. Furthermore, Kyrgyzstan plans to construct a mountain research station with three monitoring sites based on the altitude gradient, and Uzbekistan also intends to construct one oasis farmland research station and two desert research stations.

7.5.2 Aral Sea Crisis

Significant changes in water resources have occurred in the past three decades in Central Asia, including large inland rivers and the Aral Sea. Water scarcity affected the majority of Central Asian regions.

Water resources have been transferred from the west to the east and from the lower reaches to the upper and middle reaches. This water transfer was primarily

Table 7.5 Details of field observation and research stations in Central Asia

No.	Station name	Altitude (m)	Country	Location
1	Akdala Oasis Ecosystem Field Observation and Research Station	414	KAZ	Lower reaches of the Ili River
2	Atyrau Desert Ecosystem Field Observation and Research Station	−20	KAZ	Lower reaches of the Ural River
3	Karadzerin Wetland Ecosystem Field Observation and Research Station	55	KAZ	Northern black calcareous steppe belt
4	Karabarek Grassland Ecosystem Field Observation and Research Station	195	KAZ	Akmola region
5	Huchinsk Woodland Ecosystem Field Observation and Research Station	400	KAZ	Aral-Syr Darya Delta
6	Ural Mountain Ecosystem Field Observation and Research Station	204	KAZ	Ural mountainous region
7	Gondola Mountain Ecosystem Field Observation and Research Station	1411	TJK	Near the city of Dushanbe
8	Girondi Plateau Ecosystem Field Observation and Research Station	3600	TJK	Western region of Pamirs
9	Dangara Grassland Ecosystem Field Observation and Research Station	700	TJK	Central Dangara region
10	Kizilsu Mountain Ecosystem Field Observation and Research Station	2540	KGZ	Tesk Alatao Mountain
11	Jalalabad Farmland Ecosystem Field Observation and Research Station	727	KGZ	Fergana Basin
12	Isek Lake Ecosystem Field Observation and Research Station	1620	KGZ	Southeast of Lake Issyk-Kul
13	Zangiota Agroecosystem Field Observation and Research Station	370	UZB	Near the city of Tashkent
14	Bukhara Desert Ecosystem Field Observation and Research Station	273	UZB	Aral-Amu Darya Delta
15	Muynak Aral Sea Ecosystem Field Observation and Research Station	52	UZB	Kizilkum Desert
16	Khovd Desert Grassland Ecosystem Field Observation and Research Station	1437	Mongolia	Near the city of Khovd
17	Rasht Caspian Sea Ecosystem Field Observation and Research Station	28	Iran	Rasht University Campus
18	Isfahan Desert Ecosystem Field Observation and Research Station	1626	Iran	University of Isfahan
19	Shiraz Farmland Ecosystem Field Observation and Research Station	1794	Iran	Faculty of Agriculture, Shiraz University

Fig. 7.14 Field observation and research stations in Central Asia

accomplished through upstream river closure (reservoirs) and diversion (irrigation). The transferred water was used for regional agricultural and industrial development, causing water shortages downstream. Furthermore, it is reasonable to assert that the downstream ecological crisis was caused by the irrational exploitation of water resources in the upstream, which warns again the unsustainable development of Central Asian countries. The water storage in the northeast of Kazakhstan showed an increasing trend, while the water storage in the Aral Sea area exhibited an obvious decreasing trend.

During the period of 1910–1960, the level, area, and volume of the Aral Sea had been increasing, with the peaks in 1960: the highest lake level of 53.40 m, the largest lake area of 69,000.00 km^2, and the largest lake volume of $10{,}830.0 \times 10^9$ m^3. After 1960, the water from the upstreams and deltas of the Amu Darya and Syr Darya has been cut off by hydroelectric stations, reservoirs, and agricultural irrigation, so the water volume flowing into the Aral Sea has been decreasing. The Aral Sea was completely divided into two parts in 1986: the Large Aral Sea (south) and the Small Aral Sea (north). Following this division, the Amu Darya River supplies water to the Large Aral Sea, while the Syr Darya River supplies water to the Small Aral Sea. During the period of 1960–1986, the water level of the Aral Sea declined from 53.40 to 41.94 m, the lake area has decreased from 69,000.00 to 43,000.00 km^2, and the lake volume reduced from $10{,}830.0 \times 10^9$ to $4{,}446.0 \times 10^9$ m^3, corresponding to a 60% decrease.

During the period of 1986–2006, the level, area, and volume of the Small Aral Sea were remained unchanged, while those of the Large Aral Sea was decreased rapidly: the lake level decreasing from 41.02 to 30.40 m, the lake area from 38,000.00 to 13,000.00 km^2, and the lake volume from $3{,}806.3 \times 10^9$ to 814.0×10^9 m^3. In total, the lake area of the Aral Sea had decreased from 43,000.0 to 16,000.0 km^2, and the lake volume decreased from $4{,}446.0 \times 10^9$ to $1{,}054.1 \times 10^9$ m^3 from 1986 to 2006. The lake volume in 2006 was approximately 10% of that in 1960.

In 2006, the Large Aral Sea was divided into the west Aral Sea and east Aral Sea. The lake level of the Small Aral Sea was increased by 0.4 m between 2007 and 2014. The lake area of the Small Aral Sea showed an "increase–decrease" pattern during the period of 2007–2014, with the peak value in 2011 and a small change range. The level, area, and volume of the east Aral Sea and west Aral Sea exhibited a continuous reduction trend, with the eastern part decreasing faster than the western part. From

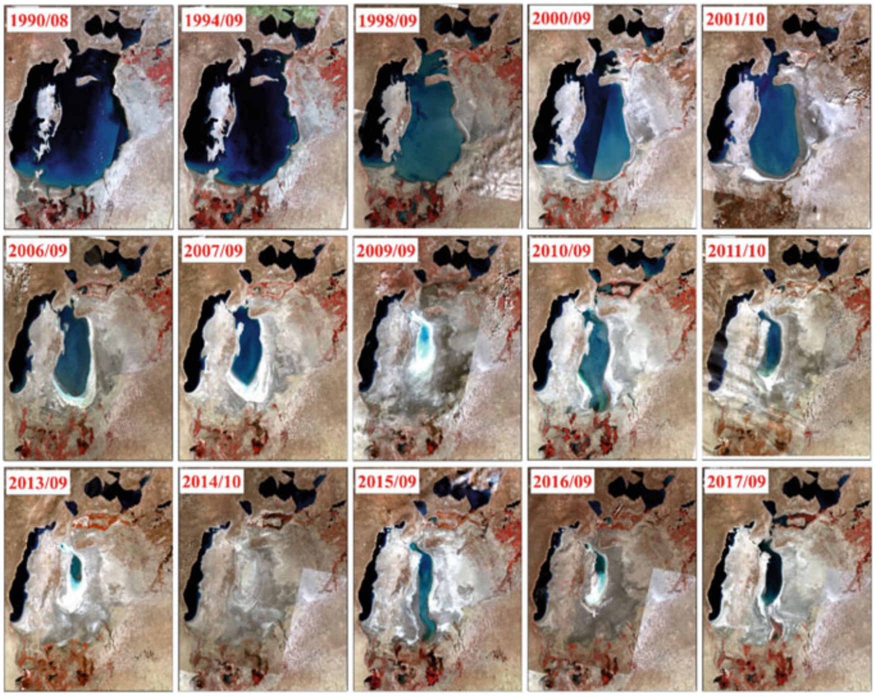

Fig. 7.15 Changes of lake surface area of the Aral Sea

2009 to 2013, the lake area of the Aral Sea changed to some extent, but showed a downward trend on the whole (Figs. 7.15 and 7.16).

The shrinkage of the Aral Sea cannot be solely attributed to water abstraction over the past 30 years (Yu et al. 2019). Varis (2014) has proposed that irrigation-intensive industries in the former Soviet Union have drained water bodies in the Central Asian countries. Massive amounts of water consumption have indeed resulted in a reduction in the lake area of the Aral Sea since the 1960s (Saiko and Zonn 2000), but the situation has not substantially changed in the last three decades (Yu et al. 2019). Water inflows have been much lower than water losses since the disintegration of the former Soviet Union, and the Aral Sea has gradually dried up due to excessive irrigation water use in the upper tributaries (Yu et al. 2019). Today, the gradual draining of the Aral Sea is not only a regional issue, but also becomes a world-renowned transnational ecological disaster (Cai et al. 2003; Kitamura et al. 2006; Yu et al. 2019).

Fig. 7.16 Current status of the Aral Sea

7.5.3 Response Measures to Aral Sea Crisis

The Muynak Aral Sea Ecosystem Field Observation and Research Station (see Fig. 7.17) is located in the Nukus region of Karakalpakstan Autonomous Republic of Uzbekistan, about 100 km away from the capital Nukus City. The station has an elevation of 52 m above sea level, with the geographical coordinates of 43°41′57.03″N and 59°02′10.04″E. The station is dedicated to studying the Aral Sea crisis.

On April 16 2019, the station was completed and began normal operation, with the assistance of the Institute of Botany, Uzbekistan Academy of Sciences. Total solar radiation, scatter radiation, ultraviolet radiation, net radiation, sunshine duration, soil temperature, and soil heat flux are the six meteorological elements measured

Fig. 7.17 Photography of Muynak Aral Sea Ecosystem Field Observation and Research Station

by the station. In addition, there are CO_2/H_2O flux observations and corresponding biometeorological observation systems in the station. The observation indicators include soil temperature, humidity, heat flux, etc.

Central Asia is located in an arid area with a fragile ecological environment, and is sensitive to climate change. Due to the limitations of the natural environment, historical changes, and socio-economic development conditions, there is a lack of long-term and reliable monitoring data on climate change and environmental factors in Central Asia. The scientific community generally has a limited understanding of environmental impact and green sustainable development model of climate change and human activities in this area. The construction of the ecosystem observation and research network in Central Asia will significantly enhance the long-term monitoring capacity of the area, provide a more comprehensive understanding of the basic information of the natural environment, and deepen the understanding of the relationship between climate change and social development.

The Aral Sea has shrunk dramatically since the second half of the twentieth century. The Aral Sea was divided into the Large Aral Sea (south) and the Small Aral Sea (north) in 1986, and the Large Aral Sea was further divided into the east Aral Sea and the west Aral Sea, due to the continuous decline of water level. The surrounding climate has changed as the shrinkage of the Aral Sea. The establishment of long-term observation sites in this region will be extremely useful for the study of the Aral Sea issues.

The Muynak Aral Sea Ecosystem Field Observation and Research Station is based on the frontier of the Aral Sea research, taking the Aral Sea ecosystem in the arid area as the objective to carry out long-term positioning monitoring and research on the Aral Sea ecosystem and environmental factors in the arid area. It helps to study the change law, evolution trend, driving mechanism, and environmental effects of the Aral Sea and its environmental factors, as well as the ecological engineering models and ecological technologies for the restoration, reconstruction, protection and rational utilization of the Aral Sea, which can provide theoretical basis and long-term data support for solving some fundamental problems in the study of the Aral Sea and its regional environment.

7.5.4 Conservation and Effective Practices of Drylands in Central Asia

To address the dryland changes, the five countries employed many policies, actions, and projects in multiple sectors, especially in saving water resources and increasing water use efficiency. These policies, actions, and projects are mostly supported by the United Nations Economic Commission for Europe (UNECE) and World Bank.

For the water and soil conservation measures, modernized irrigation system was established in the five countries of Central Asia. For example, water saving irrigation technologies and efficient irrigation technologies are used in the agriculture irrigation. When drip irrigation and spray irrigation are employed, water consumption per ton of produce is low and crop yield is high, which can improve the irrigation water efficiency (Fig. 7.18) (Rau 2016). Water conservancy projects are largely built across the whole Central Asia.

The irrigation water efficiency of irrigation systems in Kazakhstan is determined by a variety of factors, including crop structure, land-use intensity, and technologies. One method for increasing irrigation water efficiency is to build technically advanced irrigation systems that allow the use of water-saving irrigation technology such as drip irrigation, which can save 20–30% of irrigation water while increasing productivity by 2.0–2.7 times. The Second Irrigation and Drainage Project (IDIP-2) contributes to addressing a key pillar of the Kazakhstan Green Economy Concept: effective water resource management. The seven-year project aims to improve irrigation and drainage service delivery as well as the participation of water users in developing and managing the modernized systems in the four most densely populated regions in the south of Kazakhstan: Almaty, Kyzylorda, South Kazakhstan, and Zhambyl oblasts (The World Bank 2014).

For Uzbekistan, the total annual water withdrawal increased steadily from 45.50 km^3 in 1975 to 62.80 km^3 in 1985, mainly due to the expansion of irrigated land. The total annual water withdrawal was 62.50 km^3 from 1990, a declined trend because of agricultural water-saving methods and the recession in the industrial sector. In 2001, the total annual water withdrawal was estimated as 60.60 m^3, of which 3.90 km^3 was groundwater; in 2005, this was an estimation of 56.00 km^3, of which 5.00 km^3 was groundwater. Water allocations were regularly reduced to promote savings, satisfy

Fig. 7.18 Drip irrigation in Central Asia

demand from new users, and increase water flow to the Aral Sea. The total annual irrigation water withdrawal declined from 58.80 km^3 in 1990 to 50.40 km^3 in 2005 (FAO 2012a). Seventy kilometers of the Bustan irrigation channel will be modernized to significantly reduce water losses (50%) and decrease water withdrawn from the Amu Darya River. Irrigation supply will become more reliable, and farmers will be able to cultivate higher-value crops such as fruits and vegetables, which require less water and can generate five times the income of cotton and wheat.

The best water saving technologies in Kyrgyzstan were proposed by a team of national and international experts, focusing on the most promising of existing global practices. Currently, land is irrigated using surface water by distributing water to the surface of agricultural land through a system of ditches, but the associated water loss is about 50.0%. Comparing the advantages and disadvantages of two irrigation methods: sprinkler irrigation and drip irrigation, both require significant upfront investments but aid in the conservation of water resources and increase productivity (UNECE 2015; https://unece.org/press/unece-helps-kyrgyzstan-identify-more-efficient-irrigation-technologies).

For Tajikistan, the irrigation potential area was estimated as 1,580,000.00 hm^2, which is about 11.0% of the country's total area. Surface irrigation is the main irrigation technique used in Tajikistan. Drip, sprinkler, and micro-sprinkler irrigation technologies were applied in a small area only at the experimental level. In 2009, the surface water irrigation area was about 696,476.00 hm^2 (or 93.9% of the total full control irrigation area), the groundwater irrigation area was about 32,500.00 hm^2

(4.4%), and the mixed surface water and groundwater irrigation area was approximately 13,075.00 hm^2 (1.8%). Monitoring of direct use of agricultural drainage water and treated wastewater is difficult. The irrigated area pumped water from rivers is 298,500 hm^2. A new test shows that cotton water use can be saved effectively by a new irrigation technology (FAO 2012b).

For Turkmenistan, the irrigation area is 7,013,000.00 hm^2, which is equal to the cultivated land area. However, the area irrigable by water resources is estimated to be 2,353,000.00 hm^2. In 2006, the area equipped for irrigation was estimated as 1,990,800.00 hm^2. The whole area is the irrigation area, which is larger than the cultivated land area, because the irrigation area includes irrigated permanent pasture, which is not included in the cultivated land area. In 1994 and 1975, the area equipped for irrigation was 1,744,100.00 hm^2 and 857,000.00 hm^2, respectively. Cotton, wheat, vegetables, beetroot, melons and watermelons, lucerne, and corn are being planted in the field for the first time in many years. It is shown that the combination of modern water-saving irrigation technology and high-tech agricultural crop cultivation method has achieved good results. A 2.5–3.0 times reduction in irrigation water compared to conventional irrigation in 2018 could produce about 60 centers/hm^2 of cotton. At the same time, the amount of harmful salts in the soil has been significantly reduced (UNDP 2019).

7.6 Summary and Perspectives

With a total area of 4.0×10^8 hm^2 and a total population of around 65 million, Central Asia mainly consists of drylands, which are very sensitive to global climate change. In recent years, the five Central Asian countries' populations and economies have increased, with Turkmenistan showing the fastest growth rates in GDP and per capita GDP.

Desert, semi-desert, and steppe are the most common ecosystem types in Central Asia; and vegetation types in Central Asia are diverse, rich, and unique. Farmland change, forestry activities, and grazing are examples of mainland use/land cover changes and land management in Central Asia, each of which has a unique impact on the ecosystem structures and functions. Land degradation in Central Asia was primarily caused by rangeland degradation, desertification, deforestation, and farmland abandonment. The temperature in Central Asia continues to rise, glacier melting accelerates, water resource stability deteriorates, and uncertainty grows, resulting in an increase in the frequency and severity of floods, droughts, and other disasters.

The ecosystem and environment of the Aral Sea have become the key issues to be solved urgently for the sustainable development of Central Asia. The ecosystem NPP was decreasing over the past years, and the organic carbon pool in the drylands of Central Asia was seriously threatened by climate change, losing approximately 0.46 Pg C from 1979 to 2011. Grazing was an obvious strong carbon source process during the former Soviet Union period (1979–1991), but since the disintegration of the former Soviet Union (1992–2015), this activity has been converted into a weak carbon source process.

From 1995 to 2015, the value of ecosystem services in Central Asia increased overall, with grassland contributing the most. Except for food production, raw materials, climate regulation, soil formation, and waste treatment, most ecosystem functions decreased between 1995 and 2015; however, most ecosystem functions are expected to increase between 2015 and 2035 (except for water regulation and cultural service/tourism).

Global climate change poses a clear threat to the ecological diversity of Central Asia. Drylands in Central Asia are threatened by both natural and anthropogenic disturbances. The increase of precipitation cannot compensate for the aggravation of water shortage caused by temperature rise in Central Asia. The following suggestions are proposed for the long-term management of Central Asia's hydrology, socioeconomics, and ecosystems:

(1) Initiating an international joint research plan on water-social economy-ecosystem in the Aral Sea Basin.
(2) Implementing a scientific research plan on water and ecosystem in the context of climate change.
(3) Conducting joint monitoring research on the sources and diffusion paths of salt dust in the Aral Sea.
(4) Researching salt-tolerant and drought-tolerant vegetation cultivation and ecological restoration of the arid lakebed.
(5) Increasing international cooperation in biodiversity conservation and ecological security among the Central Asian countries and supporting the implementation of international joint protection actions are now of great importance.

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Chapter 8
Dryland Dynamics in the Mediterranean Region

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Abstract Mediterranean drylands are rich in biodiversity and play an important role in global ecosystem sustainable management. This study summarizes the characteristics, dynamic change, and change drivers of Mediterranean drylands. The drylands showed strong spatial heterogeneity, hyperarid and arid regions were dominant in North Africa and West Asia, and semiarid and dry subhumid regions were widely distributed in European countries. Mediterranean dryland is experiencing a warming trend that would become stronger under representative concentration pathways (RCP) 4.5 and 8.5, which would increase the risk of land degradation and desertification. Arid North Africa and West Asia faced rapid population growth that put considerable pressure on food supply and water consumption. The conflicts among land, water, food, and the ecosystem intensified under the warming trend. The significant expansion of cropland and urbanization was widely observed in arid areas, such as Egypt, while the rotation of land reclamation, degradation, abandonment, and reclamation was observed in arid areas and caused large-scale cross-border migration. The Mediterranean region had low food self-sufficiency due to a booming population, and the crop structure of cash crops was dominant. The expansion of cropland also significantly increased the water consumption in the arid area of the Mediterranean region, and water consumption increased by 684.54×10^6 m^3 from 2000 to 2020 in Egypt. More robust models and fine spatial resolution data should be developed for the sustainable development of Mediterranean drylands.

Keywords Warming trend · Land degradation · Shrub encroachment · Biocrusts · Food security · Water consumption

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8.1 Background

The Mediterranean region is a bridge connecting Asia, Africa, and Europe, consisting of 21 countries around the Mediterranean Sea. The Mediterranean region is the birthplace of ancient Egyptian, Babylonian, Roman, and Greek civilizations. It is a global biodiversity hotspot (Médail and Quezél 1999), with up to 25,000 plant species (Cuttelod et al. 2009), making it a typical and representative area for studying dryland ecosystems. The aridity index (AI) is the ratio of potential evapotranspiration (PET) and annual precipitation (Budyko 1974). A region with an AI of less than 0.65 belongs to a drylands region (Hulme 1996). The Mediterranean region is classified as hyperarid ($AI < 0.05$), arid ($0.05 < AI < 0.2$), semiarid ($0.2 < AI < 0.5$), dry subhumid ($0.5 < AI < 0.65$), humid ($0.65 < AI < 0.75$) and hyperhumid ($AI > 0.75$) areas based on the AI. Drylands occupy 85.98% of the Mediterranean region, of which hyperarid, arid, semiarid, and dry-subhumid drylands account for 48.76%, 13.44%, 18.75%, and 5.03%, respectively. Spatially, hyperarid and arid regions occupy the largest part of the drylands of North African and West Asian countries, while semiarid and subhumid arid regions are mainly in Turkey, Greece, Italy, Spain, and Portugal, as well as in the coastal areas of Morocco, Algeria and West Asia (Fig. 8.1).

The Mediterranean region faces challenges in achieving Sustainable Development Goals under a significant warming trend and complicated anthropogenic factors. The first challenge is water shortages, as 85.98% of the Mediterranean region is classified as drylands (Zeng et al. 2021). The second challenge is the pressure to feed a rapidly growing population, especially in the Middle East and North Africa,

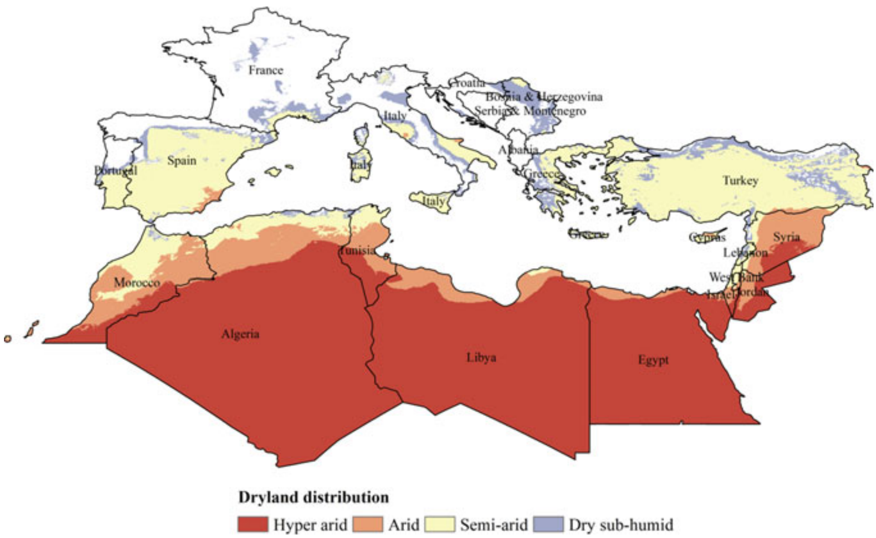

Fig. 8.1 Distribution of Mediterranean drylands based on the Global Drought Index Climate Database v2 (Trabucco et al. 2019)

where the population will double from 2015 to 2080 (Waha et al. 2017). The third challenge is a significant trend of warming (Lionello and Scarascia 2018), which tends to increase arid conditions and lead to land degradation and biodiversity loss in the ecosystem. Different challenges interact with each other. Population growth will lead to the expansion of cultivated land, which in turn will increase the consumption of water resources, while climate warming will reduce the amount of water resources. The conflict between the shortage of water resources and the expansion of cultivated land will lead to the abandonment of cultivated land and land degradation. How to balance the conflict between water, food and ecological protection is an urgent issue that requires close attention in the sustainable management of Mediterranean dryland ecosystems.

8.2 Major Characteristics of Drylands in the Region

8.2.1 Climate and Distribution of Drylands

The climate characteristics of the drylands show significant spatial heterogeneity, with arid, desert, and hot climates in the south; arid, steppe, and cold climates in Turkey and Spain; and temperate, dry, and hot summer climates in other regions. According to updated Köppen–Geiger climate data (Beck et al. 2018), there are 16 climate types in the drylands of the Mediterranean region (Fig. 8.2). BWh (arid, desert, hot), Csa (temperate, dry summer, hot summer), BSk (arid, steppe, cold), and BWk (arid, desert, cold) are the dominant climate types, accounting for 64.65%, 11.88%, 9.67% and 3.17% of the area of drylands, respectively. The Csa climate occupies the central part of coastal areas of the Mediterranean region, which favors the growth of heat-tolerant crops such as olives, grapes, figs, and citrus, as well as the accumulation of sugar in fruit crops.

The climate type, geography, and topography govern the precipitation and temperature in drylands. The annual precipitation and annual mean temperature of dryland areas are shown in Fig. 8.3 according to the WorldClim 2 dataset from 1970 to 2000 (Fick and Hijmans 2017). The annual precipitation (Fig. 8.3a) of 51.48% of the drylands is less than 50 mm and is mainly distributed in the Nile delta and the desert areas of North Africa. A total of 15.02% of the drylands receive between 50 and 200 mm of annual precipitation, mainly in the desert periphery. The annual precipitation of 8.24% of the drylands ranges between 200 and 400 mm per year, mainly in Morocco and Algeria. A total of 23.55% of the drylands have an annual precipitation amount between 400 and 800 mm per year, mainly in Turkey, parts of Europe, and the coastal areas of North Africa. Temperatures in the drylands also show significant spatial heterogeneity (Fig. 8.3b), gradually decreasing from south to north. In particular, the main parts of North Africa and West Asia are dominated by hot weather, with

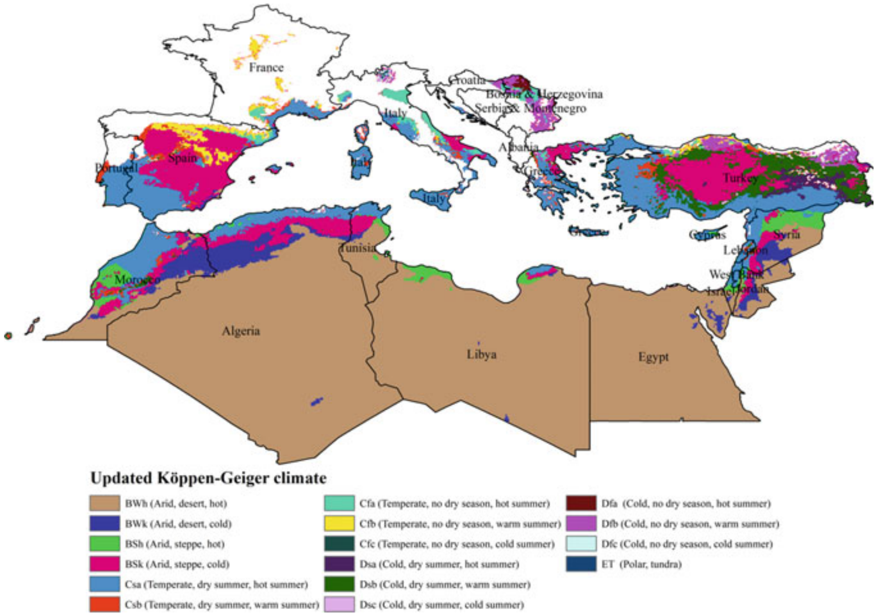

Fig. 8.2 Distribution of climate patterns based on updated Köppen–Geiger climate

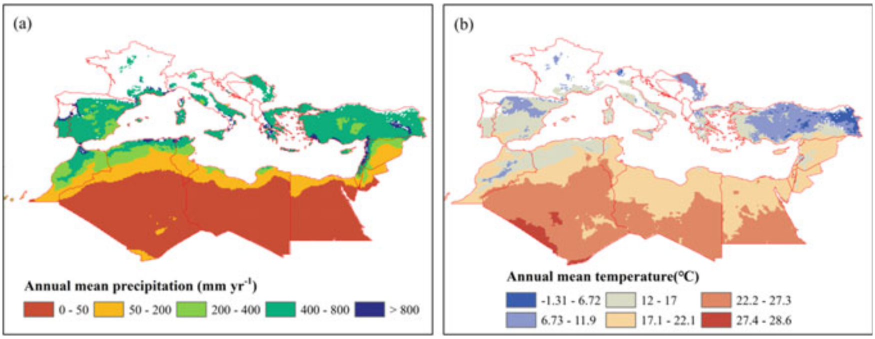

Fig. 8.3 **a** Annual mean precipitation; **b** temperature

annual mean temperatures varying between 17 and 29 °C. In contrast, the European part and the coastal areas of Morocco and Algeria are characterized by temperate and cold temperatures.

8.2.2 Land Cover and Land Use

Land cover types in the drylands of the Mediterranean region are significantly affected by dry and hot climates, which determine the predominance of low-productivity land cover in the drylands of the Mediterranean region. Moreover, water availability, soil characteristics, wildfires, and land abandonment have profound impacts on vegetation cover and patterns in the Mediterranean region (Fenu et al. 2013; Gouveia et al. 2017; Satir et al. 2016). For instance, drought strongly affects dry and desert vegetation (Gouveia et al. 2017), and dune plants along the western Mediterranean coast are deeply affected by soil properties (Fenu et al. 2013). According to the 2019 land cover and land use data from Copernicus Global Land Cover (Buchhorn et al. 2020), the land cover types and land use patterns in the drylands of the Mediterranean region show strong spatial heterogeneity (Fig. 8.4). Low-productivity bare/sparse vegetation accounts for 63.11% of the dryland area in the Mediterranean region, with most of these areas distributed in North Africa and West Asia. Cultivated agriculture accounts for 13.14% of the region's dryland area and is distributed in European countries, Turkey, and the coastal areas of North Africa and West Asia. Herbaceous vegetation, sparse forest, dense forest and shrubs account for 7.98%, 5.39%, 4.38%, and 4.46%, respectively. The highly productive land cover types are mainly distributed in the European part and Turkey, which have relatively high precipitation and mild temperatures. Although the desert climate dominates the Nile delta, it benefits from the rich water resources of the Nile River and is the central part of the cropland, vegetated, and built-up areas of Egypt.

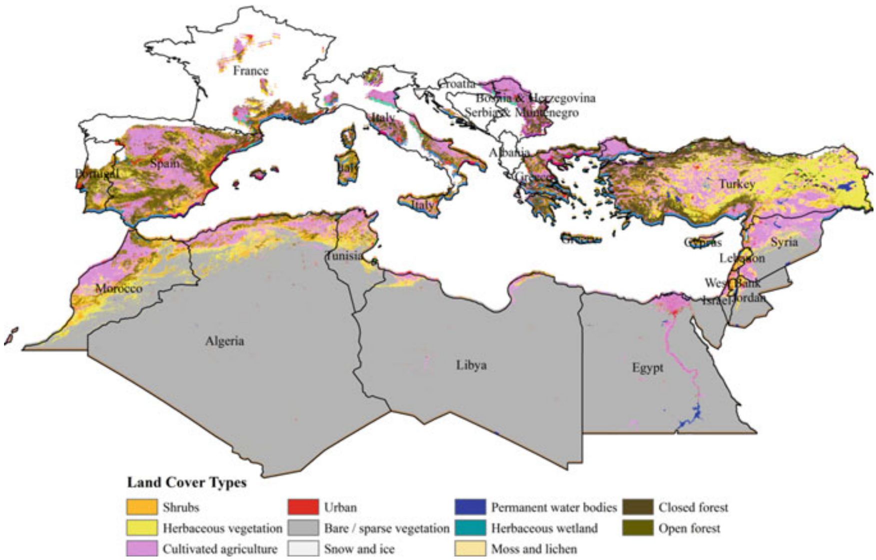

Fig. 8.4 Land cover type distribution in the dryland areas

8.2.3 Land Degradation and Its Signal

Land degradation or desertification indicates the transition from a productive vegetative state to unproductive bare land (Zelnik et al. 2013). The drylands of the Mediterranean region with high temperature and low annual precipitation are prone to land degradation and desertification. In particular, areas around deserts and barren land have a higher risk of desertification (Huang et al. 2020). Land degradation and desertification were widely observed in the southern part of the Mediterranean region (Safriel 2009) and even in the European part (Ferreira et al. 2022; Jucker Riva et al. 2017) due to the increasing trend of unsustainable land use. The results of previous studies suggest that drylands in the Mediterranean region may degrade in three ways under climate change. The first is soil erosion due to increased drought, intense rainfall, and other climatic extremes. The second is soil salinization due to increased drought, irrigation, and increased sea level. The third is the depletion of soil carbon stocks due to increased temperature and drought (Lagacherie et al. 2018). Land degradation and transboundary migration have occurred in Mediterranean drylands due to severe conflicts between population, water scarcity, and land (Mohamed and Squires 2018). This snowball effect, characterized by the reclamation, degradation, abandonment, and reclamation of arable land, has evolved land degradation in the Mediterranean region from an environmental biophysical phenomenon to a social security issue (Mohamed and Squires 2018). Land degradation drew the attention of the European Commission, which launched the Mediterranean Desertification and Land Use (MEDALUS) program to monitor the sensitivity of land to degradation and desertification during the period 1991–1999 (Kosmas et al. 1999). Different kinds of tools and methods are proposed for the assessment of soil erosion and desertification. The change in vegetation pattern (Zurlini et al. 2014) can be used to capture the early sign of land degradation. For example, spotted vegetation patterns can be considered a key signal of vegetation degradation from vegetation to bare land. Earth observations play an important role in land degradation at a large scale, while the signal of sparse vegetation in satellite images is susceptible to contamination by bare land due to low vegetation cover. The weakness of satellite data can be overcome by unmanned aerial vehicles (UAVs) that carry different sensors with a very fine spatial resolution and can capture overgrazing, aridity, and vegetation pattern changes in an easy manner (Kyriacos 2017). The abrupt change in vegetation time series can be used to identify land degradation (Smith et al. 2019). Time series segmented residual trends (TSS-RESTREND) for vegetation time series analysis (Burrell et al. 2017) is proposed to capture land degradation. With the development of remote sensing cloud computing platforms and the occurrence of mountain data, some cloud computing tools have been developed to estimate soil erosion and land degradation. Recently, Elnashar et al. developed the RUSLE-GEE for soil erosion assessment (Elnashar et al. 2021b) and MEDALUS-GEE for desertification (Elnashar et al. 2022). Both use public data to drive models to predict soil erosion and desertification, serving as tools to assess the risk of soil erosion and desertification in a developing country.

8.2.4 Shrub Encroachment

Shrub encroachment is generally considered an important cause of dryland grassland degradation (Cao et al. 2019) and has been widely reported in the Mediterranean region. It may threaten livestock and pastoralists' livelihoods (Belayneh and Tessema 2017; Nunes et al. 2019). However, whether shrub encroachment has a positive or negative impact on Mediterranean dryland ecosystems is controversial, depending on the function and traits of shrub species (Valencia et al. 2015). Several studies have shown that shrub encroachment negatively affects dryland biodiversity by altering soil bacterial communities (Stanton et al. 2018; Ubach et al. 2020; Xiang et al. 2018). Some studies have shown that shrub encroachment benefits ecosystem biodiversity. For example, Aleppo pine encroachment reduced the nesting success of Sardinian warblers and increased the activity of Eurasian jays (Ben-David et al. 2019), and soil soluble carbon and nitrogen mineralization in Mediterranean oak woodlands was higher in shrub-encroached areas than in nonencroached areas (Gómez-Rey et al. 2013). Shrub cover in agropastoral systems in southern Portugal increased above-ground biomass and net primary productivity (Castro and Freitas 2009). Shrub encroachment in degraded grasslands in Spain increased vascular plant abundance and the biomass of fungi, actinomycetes, and other bacteria (Maestre et al. 2009). The relationship between climate and encroachment is complex and controversial. Some studies have suggested that shrub encroachment might amplify the effects of climate, thus increasing the exposure of Mediterranean woody grasslands to drought (Rolo and Moreno 2019). Other studies have demonstrated that shrub encroachment has a positive effect on reversing desertification processes and improving ecosystem function (Maestre et al. 2009). However, the lack of fine-resolution shrub encroachment products hinders the ability to determine the impact of shrub encroachment on dryland ecosystems. The prediction and monitoring of shrub encroachment is essential to study its effects on dryland ecosystems. Related studies have revealed that topography and soil conditions are better predictors of shrub encroachment than climate (Nunes et al. 2019). Remote sensing has great potential for monitoring shrub encroachment, and recent studies have shown that unmanned aircraft systems and light detection and ranging (LiDAR) can be used to identify shrub encroachment (Madsen et al. 2020).

8.2.5 Loss of Biological Soil Crust

In the Mediterranean region, biological soil crusts (hereafter referred to as biocrusts) are mainly distributed in the southern and eastern arid regions, the Iberian Peninsula and Turkey (Fig. 8.5). The composition and distribution of cyanobacterial diversity in Mediterranean ecosystems are mainly governed by temperature and precipitation (Muñoz-Martín et al. 2019).

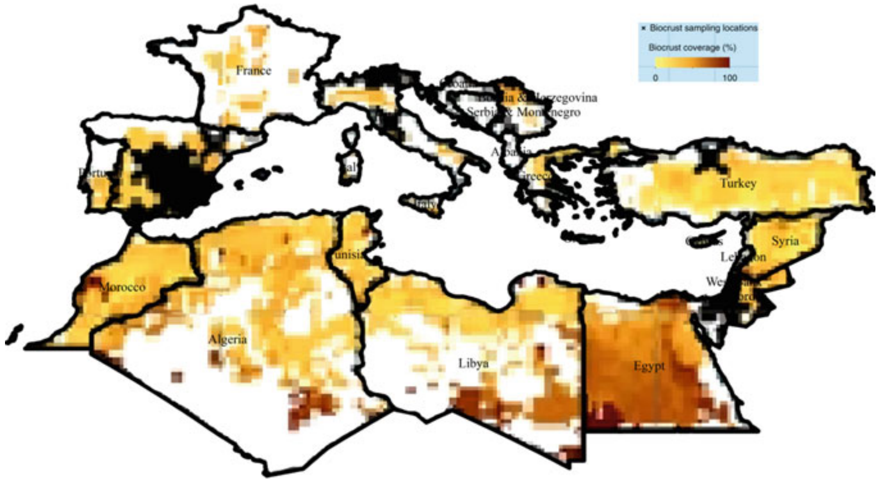

Fig. 8.5 Distribution of biocrusts in the Mediterranean region (redrawn based on Rodriguez-Caballero's study)

Biocrusts play a crucial role in maintaining the function of Mediterranean dryland ecosystems (Morillas et al. 2017). Biocrusts have profoundly impacted erosion control and the regulation of soil moisture and air quality (Morillas et al. 2017; Rodríguez-Caballero et al. 2018). Biocrusts are an excellent indicator that can be used to trace soil processes by modifying or improving soil chemistry (Miralles et al. 2020). They positively influenced seed germination and grass growth conditions of Mediterranean perennial grasses in Spain by improving soil chemistry and leaf nutrient uptake (Ghiloufi et al. 2017) and effectively increased water infiltration and soil moisture, reduced soil evaporation, and ultimately increased plant water (Chamizo et al. 2016).

The dryland ecosystem in the Mediterranean region is sensitive to climate warming. The intrinsic link between climate change and biocrusts is complex. The development of biocrusts can buffer the effect of climate warming (Delgado-Baquerizo et al. 2016; Lafuente et al. 2020) and mitigate the negative impacts of increasing aridity on the multifunctionality of dryland ecosystems (Delgado-Baquerizo et al. 2016). In turn, climate warming reduces soil water availability, leading to the loss of cover, abundance and diversity of biocrusts (Benvenutto-Vargas and Ochoa-Hueso 2020; Rodriguez-Caballero et al. 2018). In addition, biocrusts are very sensitive to atmospheric nitrogen (N) deposition and animal activity. A decrease in soil water availability and an increase in animal activity can reduce the coverage, abundance and richness of biocrusts (Ladron de Guevara et al. 2018). The warming of drylands in the Mediterranean region is already evident under representative concentration pathways (RCPs) 4.5 and 8.5. The intensity of arable cultivation and grazing in the drylands of the Mediterranean region has increased significantly with the rapid population growth in the arid zone. Under the influences of climate warming and increased human activities, the biocrust cover and abundance in the Mediterranean region will continue to decrease (Ladron de Guevara et al. 2018; Maestre et al. 2015).

Studying the changes in biocrust distribution, cover and abundance is critical in assessing the situation of dryland ecosystems. The identification of biocrusts at a large scale is challenging. Remote sensing is regarded as an effective way to map the distribution of biocrusts. Regional or global biocrust products with a fine spatial resolution are missing due to the spectral similarity between biocrusts and bare ground and signal interference from shrubland. To date, only Rodriguez-Caballero et al. (2018) have produced a global map of biocrusts with a coarse spatial resolution. Recently, satellites have developed toward high spatial resolution and frequency and provide a good opportunity for biocrust mapping. For example, Sentinel-2 multispectral data were used to trace biocrust changes in the Negev Desert (Israel) (Panigada et al. 2019). Hyperspectral airborne data were found to perform better than multispectral data in biocrust mapping (Rodríguez-Caballero et al. 2014, 2017).

8.2.6 Social and Economic Development

The populations of Mediterranean countries have shown diverse spatial changes. Populations in the southern and eastern Mediterranean region continue to grow rapidly, while European countries suffer population aging (Doignon 2020). Population aging in European countries accelerates land abandonment and shrub encroachment due to a lack of labor, which leads to a decrease in long-term soil erosion rates (Cerdà et al. 2018). The population boom in arid regions has significantly increased the pressure on the regional food supply. It has exacerbated the overexploitation of land and the overpumping of water resources in the arid region. Water limitation is prone to cause land abandonment and accelerate land degradation and desertification (Mohamed and Squires 2018). There is a very large gap in the economy between the European region and the North African and West Asian regions. Almost all North African and West Asian countries are in the lower-middle-income category, well below the world average, while most countries in Europe are in the high-income category (Fig. 8.6).

To pursue a better livelihood and higher income, large-scale population migration has been observed in the Mediterranean region. First, a large population migrated from rural areas into towns and cities to pursue better livelihoods along the Mediterranean coast (Wolff et al. 2020). Second, cross-border migration from the eastern and southern Mediterranean regions to European countries has been widely observed in the Mediterranean region (Crawley et al. 2016). In 2015, a Syrian refugee crisis occurred that led to more than 1 million people crossing the Mediterranean Sea to Europe along the eastern Mediterranean (Fig. 8.7). Cross-border migration has become a significant social issue affecting the sustainable development of Mediterranean drylands (Perkowski 2016; van Reekum 2016).

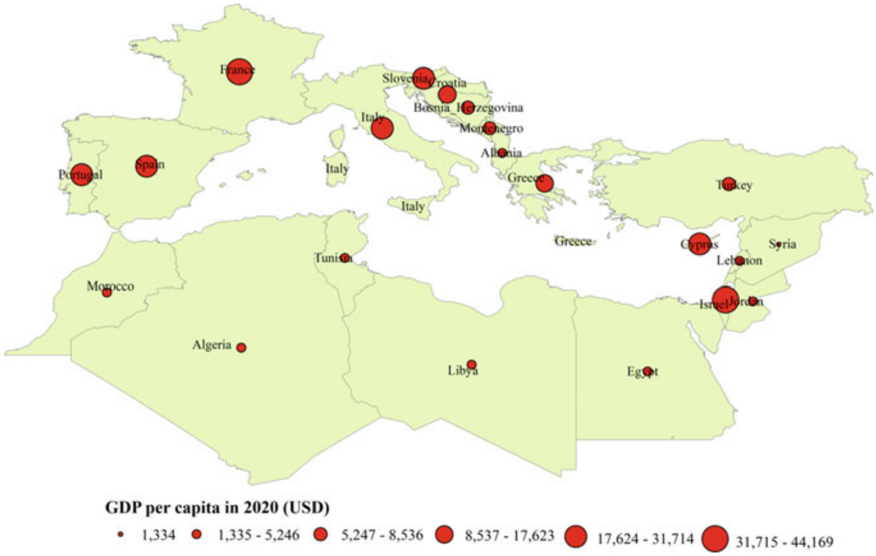

Fig. 8.6 GDP per capita (current US$) of the Mediterranean countries in 2020, Data source World Bank, https://ourworldindata.org/grapher/population-density-vs-prosperity?time=2020

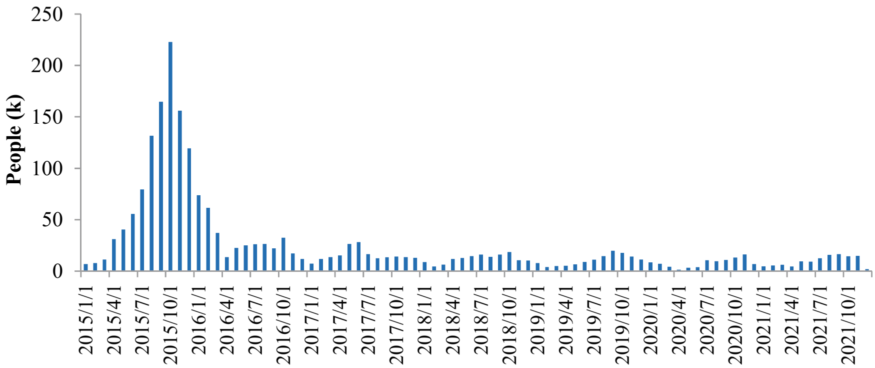

Fig. 8.7 Monthly demography of sea and land arrivals between 2015 and 2021. *Data source* UNHCR, https://data2.unhcr.org/en/situations/mediterranean

8.3 Change in Drylands in the Region

8.3.1 Climate Change

Dryland areas in the Mediterranean region have experienced a significant warming trend (Fig. 8.8), and it has been well documented that the Mediterranean region has experienced a significant decrease in precipitation and warming in recent decades

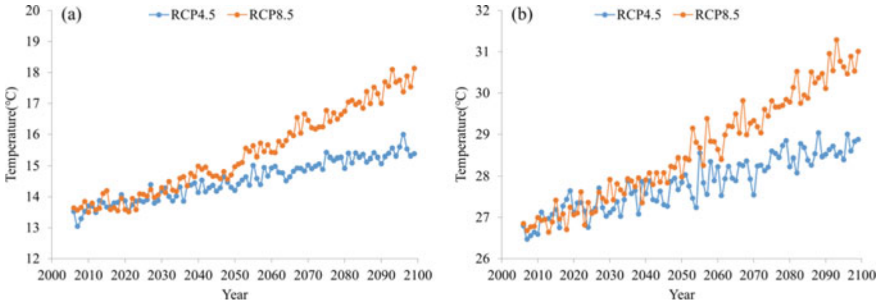

Fig. 8.8 **a** T_{min} and **b** T_{max} change trends under RCP4.5 and RCP8.5

(Prăvălie 2016). The warming trend in the Mediterranean region is more prominent than the global average, and the temperature in the Mediterranean region will be 20% higher than the global average in the twenty-first century (Lionello and Scarascia 2018). According to the CMIP5 climate change dataset (Thrasher et al. 2012), the annual growth rates of the minimum temperature (T_{min}) from 2020 to 2099 are 0.014 °C to 0.037 °C (Fig. 8.9a) and 0.034 °C to 0.075 °C (Fig. 8.10a) for RCPs 4.5 and 8.5, respectively; the annual growth rates of the maximum temperature (T_{max}) are 0.012 °C to 0.038 °C (Fig. 8.9b) and 0.033 °C to 0.079 °C (Fig. 8.10b), respectively, for RCPs 4.5 and 8.5. The most significant increase in T_{min} will occur in Turkey under RCP4.5 and RCP8.5, while the most significant increase in T_{max} will occur in Turkey and Morocco under RCP4.5 and RCP8.5.

Warming has had a significant negative impact on the dryland ecosystem in the Mediterranean region. Adverse effects have been widely found in regional ecology and sustainable development. For example, warming has led to the northward expansion of semiarid areas in the Mediterranean region (Feng and Fu 2013) and a decline in productivity, mediating the relationship between biodiversity and dryland ecosystem stability (García-Palacios et al. 2018). The warming trend has increased the frequency of droughts and heavy rainfall, aggravated soil erosion and salinization, and led to the depletion of soil carbon stocks (Lagacherie et al. 2018). It has reduced the coverage

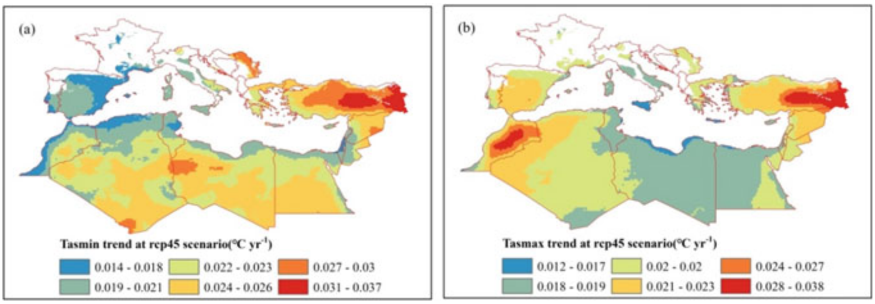

Fig. 8.9 **a** T_{min} and **b** T_{max} change trends from 2020 to 2010 at RCP4.5

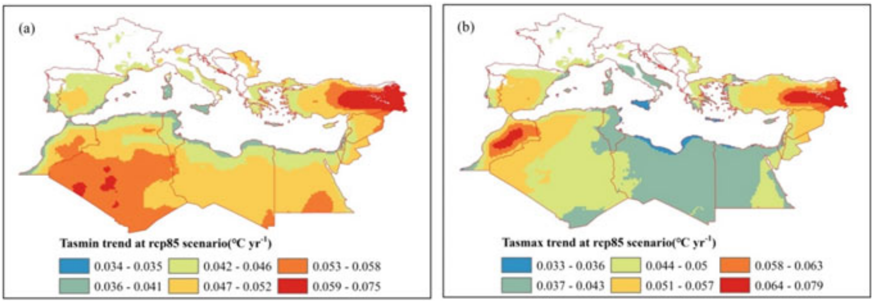

Fig. 8.10 **a** T_{min} and **b** T_{max} change trends from 2020 to 2010 at RCP8.5

of biocrusts (Rodriguez-Caballero et al. 2018), accelerated the loss of biodiversity (Verdura et al. 2019), and decreased species richness (Newbold et al. 2020). It has increased the risks of fire (Turco et al. 2018), land degradation and desertification (Yao et al. 2020) and exacerbated environmental problems. It has also increased the risks to water, ecosystems, food and health (Cramer et al. 2018). Warming and drying trends in the Mediterranean region have severely affected crop yields, leading to decreases in barley (Cammarano et al. 2019), olive in Western Europe (Fraga et al. 2020), and sunflower and wheat in the Mediterranean region (Abd-Elmabod et al. 2020). A recent study indicated that the wheat yield would decrease and the wheat price would increase in Egypt under the 2 °C warming scenario (Zhang et al. 2022). There are also some negative impacts on livestock, with shifts and reductions in livestock production in the Mediterranean region due to frequent and intensified droughts resulting from warming (Daliakopoulos et al. 2017).

8.3.2 NPP Change Trends

Vegetation indices based on remote sensing can reflect the dynamics, greenness, and biomass of vegetation, so they are widely used to assess changes in dryland ecosystems. The net primary productivity (NPP) of vegetation plays a vital role in the carbon cycle by indicating the amount of plant carbon fixed in the atmosphere minus the carbon released by respiration (Ji et al. 2020). According to the global 500-m Terra NPP gap-filling annual data from 2000 to 2020 (Running and Zhao 2021), the annual NPP in the Mediterranean region varied between 0 and 2.1 kg*c/m^2. The spatial distribution of NPP (Fig. 8.11a) is closely related to the intensity of aridity and water availability. The areas with a higher NPP are mainly located in the relatively humid European region and the coastal area of North Africa, as well as in the Nile delta where irrigated agriculture is well developed. In contrast, areas with a low NPP are mainly located in the dry Sahara Desert and the central and eastern Anatolian Plateau. The NPP in the Mediterranean drylands showed significant spatial variation. From 2000 to 2020, almost all dryland regions, such as Turkey, Greece,

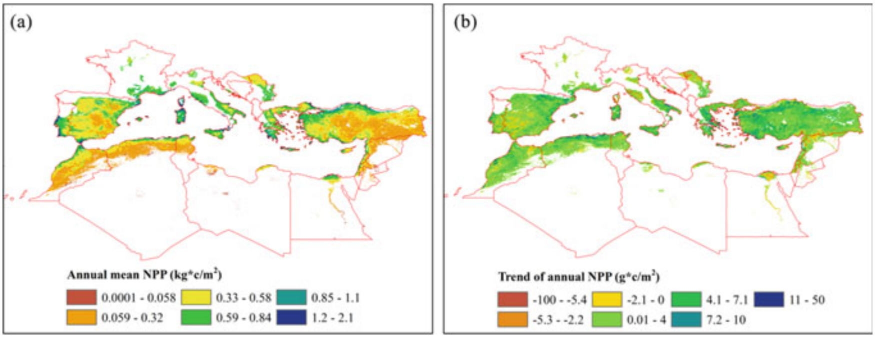

Fig. 8.11 Annual mean NPP and its trend in the dryland areas in the Mediterranean region

Italy, and northeastern Spain, showed a significant increase in the annual NPP, while the Nile delta had a decreasing trend in the annual NPP (Fig. 8.11b).

8.3.3 Land Cover and Vegetation Changes

We used a chord diagram to describe the land cover and land use conversion from 2000 to 2020 in the drylands of the Mediterranean region (Fig. 8.12). Agricultural land and bare land are the major sources of settlement and indicate the rapid urbanization process in the Mediterranean region. Bareland is converted into agricultural land, which reflects the process of cropland expansion. Agricultural land change is complex, bare land is the primary source of agricultural land, and agricultural land and forest are converted from each other. Moreover, the vegetation significantly changed in the Mediterranean region in recent decades. Between 1999 and 2012, the vegetation cover in the Middle East and North Africa decreased significantly, except in sporadic areas of Algeria and Egypt. Forest extent, structure, and composition in the northern Mediterranean have experienced dramatic changes and have become fragmented (Doblas-Miranda et al. 2017). Vegetation cover and the size and spatial pattern of vegetation patches have a direct impact on the health of dryland ecosystems (Meloni et al. 2020; Meron 2016). Dryland vegetation in the Mediterranean has a unique spatial pattern, ranging from alternating regular bands of vegetation and bare ground to regular gaps of bare ground within a continuous vegetation cover and scattered vegetated spots (Mander et al. 2017). Declining vegetation cover in small and overdispersed patches can lead to a rapid and significant loss of ground arthropod diversity (Meloni et al. 2020).

Egypt is the country with the largest population in the Mediterranean region. At the current stage, the population in Egypt has surpassed 100 million and is experiencing very large pressure on the food supply. Under the pressure of the rapidly growing population, the agricultural area and settlement area of Egypt significantly expanded from 2000 to 2020. The conversion matrix of land cover in Egypt from 2000 to 2020

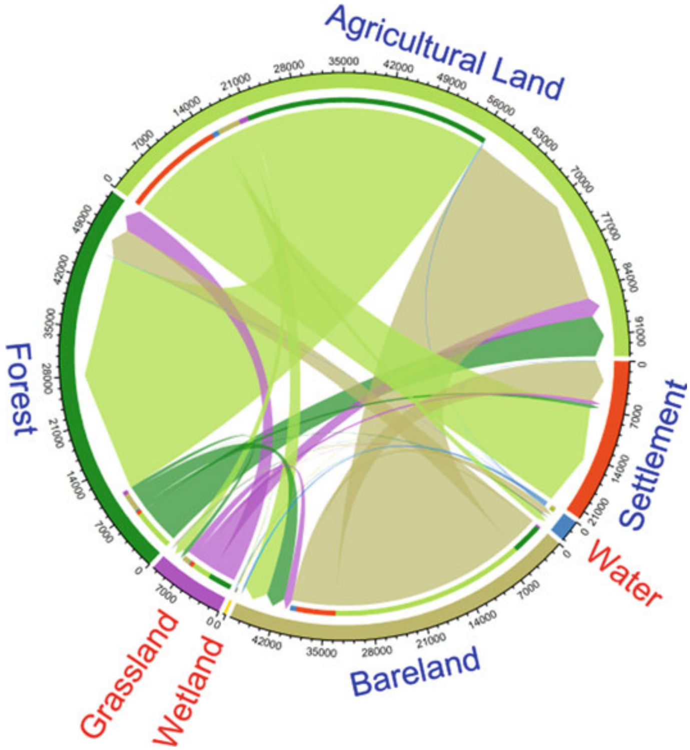

Fig. 8.12 Chord diagram of land use/land cover change (LULCC) changes from 2000 to 2020 in the drylands of the Mediterranean

is shown in Table 8.1. From 2000 to 2020, the agricultural land and settlement land expanded rapidly; in return, the area of bare land decreased by 3941.6 km^2. To feed the increasing population, Egypt tried to expand the area of agriculture to improve food production. The agricultural area increased from 54207.4 km^2 in 2000 to 55476.2 km^2 in 2020, an increase of 2.3%. A total of 3100.9 km^2 of bare land was converted into agricultural land. This conversion matrix also reflected the rapid urbanization process in Egypt. From 2000 to 2020, the area of settlement increased from 1506.4 km^2 to 3955.8 km^2, an increase of 162.6%. Agricultural land and bare land were the primary sources of increased settlement, and 1729.4 km^2 of agricultural land and 645.0 km^2 of bare land were converted into settlement. Water was the major limiting factor to agricultural development, and a water deficit leads to cropland abandonment. This phenomenon has occurred in Egypt. From 2000 to 2020, a total of 75.9 km^2 of agricultural land was converted into bare land in Egypt.

Table 8.1 Land cover conversion from 2000 to 2020 in Egypt

LULC	Agriculture		Forest		Grassland		Wetland		Settlement		Bareland and other		Water		Total 2000 (km^2)
	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	
Agriculture	52333.5	96.5	0.0	0.0	0.0	0.0	2.2	0.0	1729.4	3.2	75.9	0.1	66.3	0.1	54207.4
Forest	4.2	37.3	4.0	35.2	0.0	0.0	0.0	0.0	2.9	26.0	0.0	0.0	0.2	1.7	11.3
Grassland	0.0	0.0	0.0	0.0	6.6	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.6
Wetland	9.4	0.1	0.0	0.0	0.0	0.0	11929.2	99.1	55.5	0.5	0.4	0.0	46.1	0.4	12040.5
Settlement	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1506.4	100.0	0.0	0.0	0.0	0.0	1506.4
Bareland and other	3100.9	0.3	0.0	0.0	0.0	0.0	0.5	0.0	645.0	0.1	1039876.9	99.6	425.4	0.0	1044048.6
Water	28.2	0.3	1.1	0.0	0.0	0.0	66.9	0.7	16.6	0.2	153.9	1.6	9279.8	97.2	9546.5
Total 2020 (km^2)	55476.2	–	5.0	–	6.6	–	11998.8	–	3955.8	–	1040107.0	–	9817.9	–	–
Change (km^2)	1268.8	–	−6.3	–	0.0	–	−41.7	–	2449.3	–	−3941.6	–	271.4	–	–
Change (%)	2.3	–	−55.4	–	0.0	–	−0.4	–	162.6	–	−0.4	–	2.8	–	–

8.3.4 Crop Structure and Food Production Per Capita Change

A mixture of drought-tolerant crops, cash crops, and livestock dominates the agriculture of the Mediterranean region. The planting area, fruit area, and pastoral production are spatially separated. Wheat, barley, and maize are the main staple crops in the Mediterranean region, and they are mainly grown on flat terrain or in areas with low slopes. The share of food crops (wheat, maize, barley, rice, oats, potatoes, etc.) in Mediterranean countries ranges from 23.0 to 70.9%. Morocco, France, Turkey, and Egypt have 70.9%, 63.8%, 62.1%, and 58.4% of their agricultural land dedicated to food crops, respectively. The other countries have less than 50%, especially Israel, Greece, Portugal, and Lebanon, with only 29.8%, 31.4%, 23.0%, and 28.3% of their area dedicated to food crops, respectively. Olives are the most important cash crop in the Mediterranean region, with shares of 39.6%, 39.3%, 36.9%, 31.5%, 35.5%, and 32.1% in Tunisia, Israel, Greece, Libya, Portugal, and Jordan, respectively. Grapes and vegetables are also significant cash crops in the Mediterranean region. In Israel, for example, vegetables are grown on 10.7% of agricultural land. Cash crops occupy a large proportion of arable land, limiting the cultivation of staple foods and leading to insufficient crop production in the countries surrounding the Mediterranean.

Using crop production data from the FAO and CropWatch monitoring platforms, the change trends of crop production (Fig. 8.13a) and crop production per capita (Fig. 8.13b) were estimated in the countries surrounding the Mediterranean region from 2010 to 2020. Crop production in Egypt, Lebanon, and Algeria showed a significant upward trend from 2010 to 2020, while crop production in Italy, Greece, and Libya showed a significant downward trend from 2010 to 2020, and crop production in the other countries showed a variable trend. As 400 kg per capita per year is a criterion to eliminate food insecurity, all North African and West Asian countries were below this level and have not achieved food self-sufficiency. In contrast, the crop production per capita in France, Spain, Croatia, Bosnia, and Turkey was over 400 kg. The food produced by these countries is enough to meet their own needs and even export to other countries, e.g., France is one of the largest food exporters in the world.

8.3.5 Water Resource Analysis

Water limitation and scarcity are the greatest challenges to agricultural development in North and West Africa. Significant changes in agro-fruit and pastoral production in the Mediterranean have occurred in the past half-century. Large-scale farms and plantations have gradually replaced traditional small farms and plantations. The development of agriculture has significantly increased water consumption and poses a significant challenge to sustainable development. Based on NASA's Gravity Recovery and Climate Experiment (GRACE) (Landerer and Swenson 2012), the change in liquid

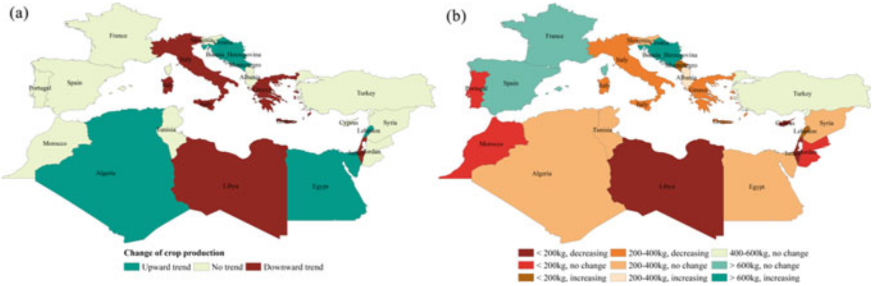

Fig. 8.13 **a** Change in food production and **b** food production per capita in 2020 and its change from 2010 to 2020

water thickness from 2003 to 2016 is shown in Fig. 8.14. West Asia, Egypt, Tunis, Algeria, and Libya have experienced the issue of declining liquid water thickness. The decline in the water table characterizes this region's climate, indicating a severe water resource crisis.

Figure 8.15 shows the distribution of the annual ET and its trends from 2003 to 2019. ET intensity is determined by land cover type, with a higher ET intensity in forests, shrubs, and arable land and a lower ET intensity in bare land and deserts due to the arid climate (Fig. 8.15a). Due to developed irrigated agriculture, the ET intensity is higher in the Nile basin and its delta. Figure 8.15b shows the different patterns of ET variation, with a strong increasing trend in the Nile delta and coastal

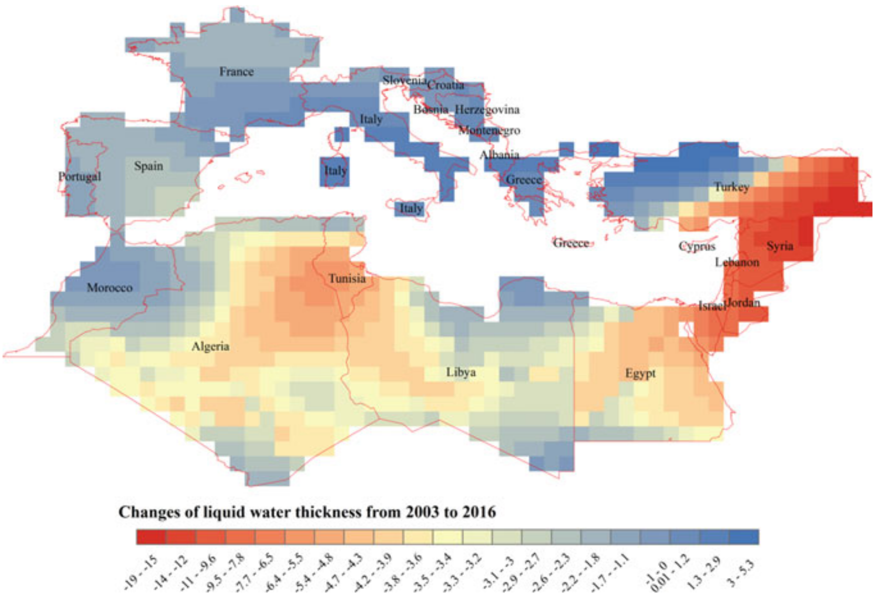

Fig. 8.14 Change in equivalent water thickness between 2003 and 2016

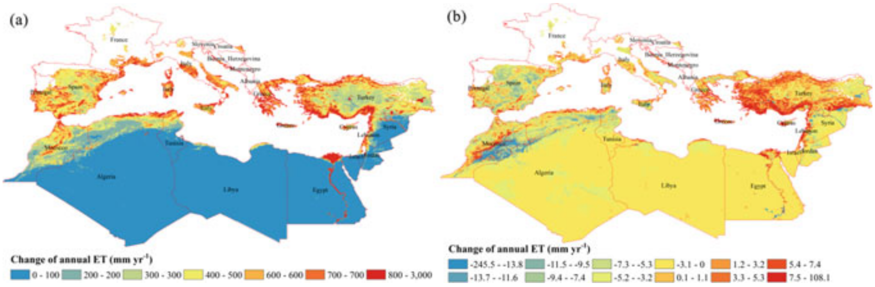

Fig. 8.15 **a** Annual ET spatial distribution and **b** its trend from 2003 to 2019

areas of West Asia, Turkey, and Greece and a significant decreasing trend in Spain and the mountainous regions from Morocco to Tunisia.

Agriculture is the largest user of water in the Mediterranean region. ET represents the actual water loss due to climate change and anthropogenic factors, and separating the contributions of natural and anthropogenic factors to ET variability can provide valuable information for water resource management. This study used a data-driven approach (Zeng et al. 2022) to quantify the impact of natural and anthropogenic factors on ET changes in the Nile basin, Tunisian agro-pastoral, Algerian agro-pastoral, Moroccan agro-pastoral, Libyan agro-pastoral, West Asian agro-pastoral, and Turkish agro-pastoral regions (Fig. 8.16).

The ET separation method first divided ET and environmental factors into a natural group $(ET, X_i)_n$ and an anthropogenic group $(ET, X_i)_a$ according to the natural and human-managed features of land cover types. Here, X_i included precipitation (P), air temperature (Tair), wind speed (Wind), downward longwave radiation flux (LWdown), downward shortwave radiation flux (DWdown), pressure (Psurf), specific

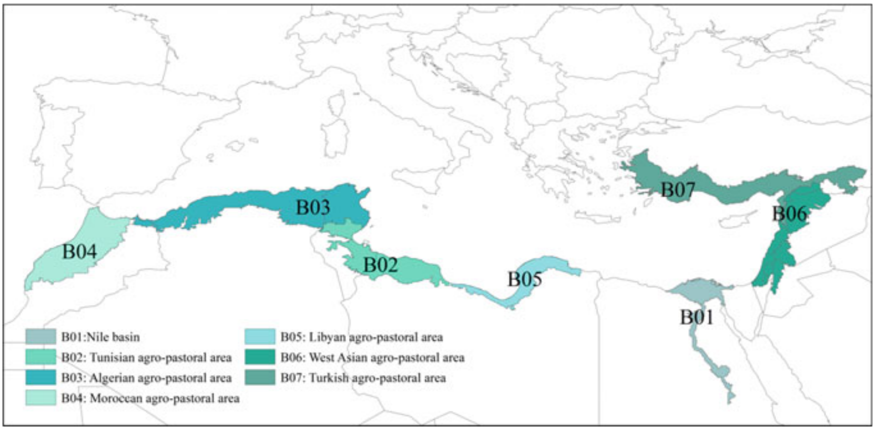

Fig. 8.16 Seven major agricultural regions in North Africa and West Asia

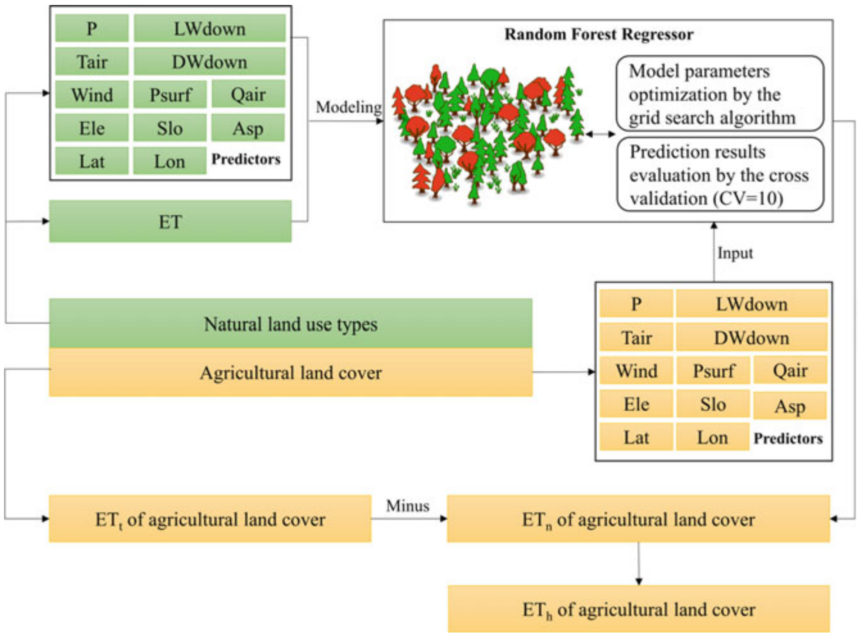

Fig. 8.17 The framework of natural ET and anthropogenic ET separation of agricultural land

humidity (Qair), elevation (Ele), slope (Slo), aspect (Asp), latitude (Lat), and longitude (Lon). Second, a random forest regressor that optimized the parameters by the grid search algorithm was employed to build the ET_n prediction model that explored the linkage between the ET and X_i of the natural group. Third, the ET_n prediction model was transferred to predict the ET of agricultural land cover caused by natural factors as X_i of the anthropogenic group as input. Finally, the anthropogenic ET (ET_a) of agricultural land cover was separated by calculating the difference between ET and ET_n. This approach is explained in Fig. 8.17.

Here, ET and environmental data came from different sources. The synthesized ET product at a 1-km spatial resolution for 2019 was used in this study (Elnashar et al. 2021a). Synthesized ET data were generated by using the simple average of the Penman–Monteith-Leuning (PML) (Zhang et al. 2019) and the Operational Simplified Surface Energy Balance (SSEBop) (Senay et al. 2013) remote sensing ET products. Precipitation data were collected from the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) (Funk et al. 2015). The CHIRPS is a daily $0.05° \times 0.05°$ (≈5 km) grid cell quasi-global rainfall dataset from 1981. It creates a gridded rainfall product by incorporating remotely sensed precipitation with in situ station data. The LWdown, DWdown, Tair, Wind, and Psurf with a spatial resolution of 0.25° (≈25 km) were provided by the Global Land Data Assimilation System (GLDAS-2.1) (Rodell et al. 2004). Elevation (Ele) information was extracted from the Shuttle Radar Topography Mission (SRTM) version 4 data (Jarvis et al. 2008) with a spatial resolution of 90 m. Slope (Slo) and aspect (Asp) data were calculated from SRTM data using spatial analysis. Land cover data were collected from the European Space Agency (ESA) Climate Change Initiative Land Cover (ESA-CCI-LC) (Bontemps et al. 2012) with a spatial resolution of 300 m. ESA-CCI-LC data extend from 1992 to 2019 with 37 land cover classes (ESA 2015, 2018). The longitude and latitude datasets were generated at a 1-km resolution by spatial analysis.

The ET_n prediction model was built independently in B01, B02, B03, B04, B05, B06, and B07. Good performance was found in 7 basins. The R^2 values were 0.83, 0.95, 0.98, 0.95, 0.99, 0.98, and 0.93, respectively; the Nash coefficients reached 0.83, 0.95, 0.98, 0.95, 0.99, 0.98, and 0.93, respectively; the mean absolute errors (MAEs) were 16.1 mm, 3.0 mm, 22.9 mm, 20.0 mm, 3.7 mm, and 21.7 mm; and the root mean square errors (RMSEs) were 58.0 mm, 7.6 mm, 34.2 mm, 30.2 mm, 10.9 mm, 35.2 mm, and 51.0 mm, respectively (Table 8.2).

The increase in ET intensity by agricultural activities was 453.9 mm, 68.0 mm, 56.1 mm, 64.4 mm, 137.8 mm, 93.3 mm, and 105.1 mm for the Nile basin, Tunisian agro-pastoral area, Algerian agro-pastoral area, Moroccan agro-pastoral area, Libyan agro-pastoral area, West Asian agro-pastoral area, and Turkish agro-pastoral area, respectively. The results indicated that ET will increase by 4539 m^3, 680 m^3, 561 m^3, 644 m^3, 1378 m^3, 933 m^3, and 1051 m^3 for B01, B02, B03, B04, B05, B06, and B07, respectively, if agricultural land increases by one hectare (Table 8.3). Taking Egypt as an example, from 2000 to 2020, the agricultural land area increased by 1508.12 km^2, this meant that the water consumption increased by 684.54×10^6 m^3.

Table 8.2 Performance summary of the ET_n prediction model in 7 basins

Major agricultural regions	R^2	RB	NSE	MAE	RMSE	P value
B01: Nile basin	0.83	4.13	0.83	16.1	58	0
B02: Tunisian agro-pastoral area	0.95	−0.2	0.95	3	7.6	0
B03: Algerian agro-pastoral area	0.98	−0.16	0.98	22.9	34.2	0
B04: Moroccan agro-pastoral area	0.95	−0.12	0.95	20	30.2	0
B05: Libyan agro-pastoral area	0.99	−0.2	0.99	3.7	10.9	0
B06: West Asian agro-pastoral area	0.98	−0.19	0.98	21.7	35.2	0
B07: Turkish agro-pastoral area	0.93	−0.19	0.93	35	51	0

Table 8.3 Summary of increased ET consumption by agriculture in 2019

Basin	Agricultural type	ET_t (mm)	ET_n (mm)	ET_h (mm)	Area (km^2)	Components		
						ET_t (mm)	ET_n (mm)	ET_h (mm)
B01	Rainfed agriculture	586	360	225	246	674.5	220.5	453.9
	Mosaic agriculture	321	150	171	6390			
	Irrigated agriculture	741	233	508	34,231			
B02	Rainfed agriculture	188	130	59	14,390	198.6	130.6	68.0
	Mosaic agriculture	387	150	237	774			
	Irrigated agriculture	245	37	208	18			
B03	Rainfed agriculture	352	286	66	51,593	407.0	350.9	56.1
	Mosaic agriculture	441	391	50	82,707			
	Irrigated agriculture	570	334	236	7			
B04	Rainfed agriculture	404	322	82	25,427	392.0	327.5	64.4
	Mosaic agriculture	384	331	53	48,063			
	Irrigated agriculture	527	238	289	451			
B05	Rainfed agriculture	441	221	220	121	330.5	192.7	137.8
	Mosaic agriculture	325	189	136	7504			
	Irrigated agriculture	614	411	203	102			
B06	Rainfed agriculture	359	278	81	38,501	405.7	312.4	93.3
	Mosaic agriculture	459	352	108	27,851			
	Irrigated agriculture	568	436	132	1732			
B07	Rainfed agriculture	498	459	39	25,175	556.8	451.6	105.1
	Mosaic agriculture	564	467	97	43,189			
	Irrigated agriculture	603	418	184	25,606			

8.4 Driving Forces of Dryland Change

8.4.1 Climate Change

The significant warming trend has had a strong negative impact on the extent of drylands, vegetation productivity, biodiversity, and stability of the dryland ecosystem. For example, warming may reduce soil water availability (Schlaepfer et al. 2017), soil fertility (Berdugo et al. 2020), plant productivity (Berdugo et al. 2020; Yao et al. 2020), leaf abundance, and species diversity (Maestre et al. 2016). It may also affect nutrient cycling and soil microbial communities (Maestre et al. 2016) and increase the risks of drought, land degradation, and desertification in dryland regions (Huang et al. 2017; Tietjen et al. 2017). Food security in the Mediterranean region has also been affected by a warming trend. Many studies have reported that warming has a negative impact on crop yield and livestock production due to increased frequency and intensified drought (Abd-Elmabod et al. 2020; Cammarano et al. 2019; Fraga et al. 2020; Mohamed and Squires 2018). New studies have indicated that the warming trend is critical for Syria's civil war (Selby et al. 2017). First, severe drought caused a significant decline in wheat production and resulted in large-scale migration. The latter exacerbated the socioeconomic stresses that underpinned Syria's descent into war.

Different climate scenarios have indicated that a significant warming trend would occur in the Mediterranean region (Fig. 8.10). A series of negative impacts of warming on the ecosystem would occur in the future if no reasonable human intervention occurred. As mentioned in Sect. 8.3.1, warming will increase the intensity of aridity and lead to a northward expansion of drylands (Feng and Fu 2013). Warming will increase the frequency of extreme climate events (droughts and heavy rains) and the risk of fire (Turco et al. 2018). Warming may lead to regime shifts and mediate the relationship between the biodiversity and stability of dryland ecosystems (García-Palacios et al. 2018). Warming trends will exacerbate environmental problems and increase the risks to water, ecosystems, food, and health (Cramer et al. 2018). For example, warming will aggravate soil erosion, salinization, soil carbon depletion (Lagacherie et al. 2018), land degradation (Yao et al. 2020), biodiversity loss (Verdura et al. 2019), and species richness loss (Newbold et al. 2020). In the future, the sustainable development of drylands in the Mediterranean region should pay close attention to warming trends and monitor and simulate the consequences caused by warming trends. More importantly, policy-makers should take suitable adaptation measures to reduce or even dismiss the negative impact of warming trends.

8.4.2 Anthropogenic Drivers

Population, wildfire, overgrazing, grazing abandonment, land intensity, land abandonment, and urban expansion are the main driving forces of dynamic changes in dryland ecosystems in the Mediterranean. North Africa and West Asia are experiencing rapid population growth. The populations of Egypt, Turkey, France, and Italy exceed 50 million, accounting for 21.1%, 17.3%, 14.1%, and 12.7% of the population of the Mediterranean region, respectively. The rapid population growth has led to the massive migration of the population from rural areas to towns and cities (Wolff et al. 2020). The population boom in the dry southern and eastern Mediterranean has put considerable pressure on the food supply that has aggravated the overexploitation of land and water resources (Mohamed and Squires 2018). Warming and water constraints reduce the productivity of cropland, leading to land degradation and abandonment. Land degradation and desertification would significantly reduce crop production, forcing population migration to more productive areas and even causing cross-border migration. Wildfires are another important anthropogenic driving force of dryland ecosystem change in the Mediterranean region. Studies have found that wildfire in abandoned terraces has resulted in significant soil degradation in the Mediterranean region (Lucas-Borja et al. 2018), and large fires led to the transition from Mediterranean oak woodlands to shrubland (Guiomar et al. 2015). Grazing and grazing abandonment also play a crucial role in modifying Mediterranean dryland ecosystems. Overgrazing could significantly reduce vegetation and biocrust cover and increase the risk of bare land exposure. Overgrazing is the main driver of land degradation in Spain, Greece, and Cyprus (Riva et al. 2017). The consequences of grazing on dryland ecosystems are controversial. Some studies have suggested that grazing abandonment could reduce soil fertility, carbon storage, soil multifunctionality, and soil microbial activity (Peco et al. 2017), while other studies found that grazing abandonment could increase the cover of biocrusts and benefit the stability of dryland ecosystems (Rodríguez-Caballero et al. 2018).

8.5 Summary and Perspectives

This chapter summarizes the characteristics and dynamic trends of Mediterranean dryland ecosystems and the impacts of climate change and anthropogenic drivers on dryland ecosystems. Mediterranean dryland is dominated by low productivity bare land and is experiencing a significant warming trend. Biodiversity, soil nutrients, carbon stocks, and microbial community viability are experiencing harm due to the warming trend. The sustainable development of drylands should pay more attention to the warming trend and predict the consequence of the warming trend. Due to the impact of the warming trend, land and water resources have uneven spatial distributions. North Africa and West Asia face extremely dry climate conditions and deeply suffer from water limitation and pressure from rapid population growth. The phenomena of cropland cultivation, degradation, and abandonment widely exist in the dry regions of North Africa and West Asia, even causing large-scale cross-border migration. Extreme climate events will become more frequent, widespread, and intense under the warming trend. The warming phenomenon may trigger population migration and social unrest. The lack of data and models are major issues for

Mediterranean dryland ecosystems. In the future, more models should be developed to simulate the dynamic change in Mediterranean drylands and predict the consequences of dynamic change. Reasonable measures should be taken in case catastrophic consequences occur. Moreover, to understand the changes in the Mediterranean dryland ecosystem, a series of critical products should be produced, such as biological soil crusts, shrub encroachment, and land cover, with a finer spatial resolution.

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Chapter 9
Dryland Social-Ecological Systems in Africa

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Abstract In Africa, dryland ecosystem is the largest biome complex, covering 60% of the continent and home to ~525 million people. Coupled with adverse climatic conditions and anthropogenic pressures make dryland highly vulnerable to environmental degradation. In this chapter, we elucidate an overview of dryland socio-ecological systems (DSES) in Africa. We examine dryland biodiversity as a basis for ecosystem services in Africa. Therefore, we investigate the research and technology

S. Diop died prior to the submission of this paper. This is one of the last works of him.

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gaps in African drylands. Finally, we conclude and highlight the future perspectives for sustainable DSES management. Sustainable development requires an understanding of and adherence to the proper functioning of DSES. We recommend to promote sustainable agricultural best practices and innovations as a tool to enhance community resilience and cope with climate change impacts on food security, use modern observational data and develop idealistic models to better understand the climate-drylands-food security nexus approaches, and strengthen dryland research and management effectiveness through emerging and affordable technologies.

Keywords Socio-ecological systems · Land degradation · Dryland biodiversity · Resilience · Food security · Africa

9.1 Drylands and Socio-ecological Systems in Africa

Stretching latitudinally from 37°N to 35°S and longitudinally from 52°E to 17°W, Africa accounts for 6% of the Earth's surface area and 20% of its landmass. In Africa, drylands occupy over 60% of the total surface area (~30.37×10^6 km^2 including its adjacent islands) (Kolding et al. 2016). Overall, African drylands are biophysically and socially diverse, characterized by challenging agroclimatic conditions, thus a big part of the global heritage, in terms of the crops and livestock (Cervigni and Morris 2016). Therefore, African drylands include a constellation of widely differentiated resources at spatio-temporal scales (i.e., macroscales to microscales). In drylands, the availability of potential input expands and contracts dramatically through under unpredictable intervals between years. The potential productivity of these resources, as well as their efficient and sustainable use, depends largely, or even entirely, on producers' micro-management and real time adjustments (Kratli 2020).

The Agro-Ecological Zones (AEZs) of Africa are classified into seven types of ecosystems: cultivated lands, scrublands, shrublands, grasslands, savannas, semi-deserts, and deserts. Therefore, the major physical regions of Africa include the African Great Lakes, the Ethiopian highlands, the Sahara, the Sahel, the Swahili Coast, and the rainforest, as well as the Southern Africa savanna (Kay and Kaplan 2015). High temperatures and low precipitation of the dryland regions result in limited organic matter decomposition, minimal nutrient cycling, and subsequently reduced primary productivity (Hartley et al. 2007). The dryland agroecosystems provide the inhabitants with important ecosystem services including "hotspots" of hydro-biogeochemical activities. In these regions, ~70% of the population relies on rainfed agriculture for subsistence and is deeply intertwined with nature (Jalloh et al. 2012), in absence of climate change adaptation and mitigation measures, 40–80% of people are expected to be affected from 2010 to 2030 (Nyberg et al. 2019). Figure 9.1 shows the example of the degrading impacts of anthropogenic activities on dryland ecosystems and landscape restoration in Africa.

The dryland ecosystems are described by dryland socio-ecological systems (DSES), which dynamically couple society, culture, and natural capital quantitatively

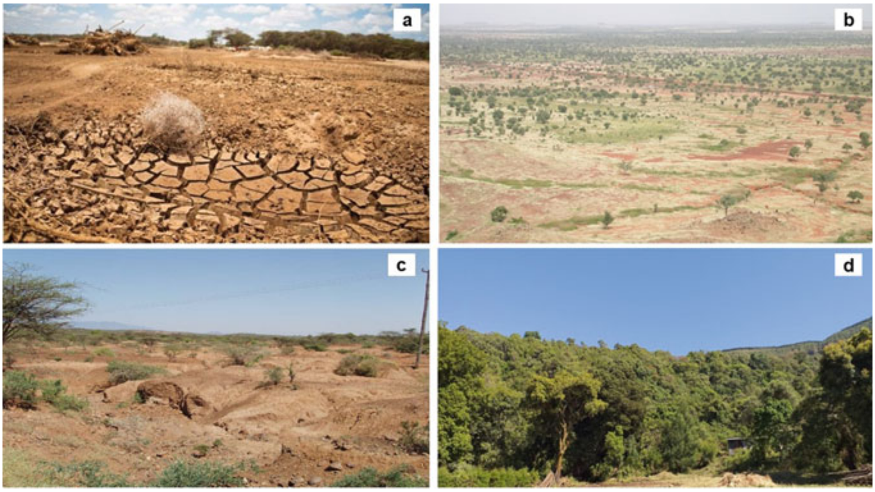

Fig. 9.1 Examples of dryland ecosystems in Africa. **a** Coupled changing weather patterns and over-abstraction are reducing river flows in Ewaso Ngiro River, Kenya. *Credit* Denis Onyodi/Kenya Red Cross Society. **b** West African Sahel and dry savannas, view of a degraded landscape in Niger. *Credit* Patrice Savadogo. **c** Degraded area by soil erosion in the rangelands near Lake Baringo, Kenya. *Credit* Lawrence M. Kiage. **d** Landscape restoration in Ethiopian highlands. *Credit* Mesfine

and qualitatively (Huber-Sannwald et al. 2012). Currently, DSES face challenges at different scales including climate variations, population growth, land degradation, desertification, water scarcity, economic and land-use changes, cultural perceptions, and governance and management practices that negatively affect DSES (Chapin et al. 2009; Linstädter et al. 2016; Villada-Canela et al. 2020). These challenges contribute to high levels of unpredictability, variability, and heterogeneity to shape the coupled human and nature co-adaptative process. For the sustainable management of natural resources, a comprehensive understanding of the complex relationships between human and natural systems is necessary (Aminpour et al. 2020). The DSES also provide guidelines for assessing the impact of social and ecological dimensions on resource use and management (Partelow 2018). There are two main conceptual pillars for DSES. Firstly, the first one is to understand the DSES function, taking into account the economic effect, and the second one is to consider all aspects related to the development, transformation, and implementation toward normative sustainability goals (Yu et al. 2021). Nowadays, in Africa, dryland management is one of the most pressing issues that need to be addressed. Presently, there is an urgent need to restore and protect the ecosystems in the semi-arid drylands of Sahel regions, Eastern Africa, Southern Africa, and Western Africa.

Using the DSES approach, this chapter aims to provide diagnostic information on Africa's drylands management. The chapter also reviews major features and trends of the DSES by assessing the driving factors of dryland change and their interactions. Attention is paid to potential future perspectives of sustainable dryland ecosystem management and efforts to enforce dryland community resilience in the face of adverse environmental changes. This chapter also addresses the research and innovation gaps in the reliable assessment of DSES assessment. This chapter could aid in enhancing the implementation of sustainability policy in Africa and worldwide.

9.2 Major Characteristics of Drylands and DSESs in Africa

9.2.1 African Dryland Distribution

Arid, semi-arid, and dry subhumid terrains in Africa make up 27% of the world's drylands and account for 11% of the Earth's land surface (Prăvălie 2016). The Sub-Saharan Africa (SSA) drylands cover ~13.9×10^6 km^2. According to a high-resolution assessment by UNDP/UNSO (1997), the total African aridity coverage is ~21.2×10^6 ha (equivalent two-thirds of the continent surface area) with three-fifths of its farming lands, and are home to two-fifths of its population. Insofar, the northern region comprises 38% of this area, which is largely in the hyper-arid category. Despite the fact that the economy of Central Africa lags behind Northern, Western, and Eastern Africa, they all share the similar aridity in their territories. The arid zone of Africa is barely 1% covered by forest in central Africa, which makes up a large part of the continent's forested areas (Kigomo 2003).

In Eastern Africa, the hyper-arid, arid, and semi-arid zones are mostly in Sudan, Djibouti, and Somalia. The semi-arid and arid zones are widespread in Kenya and Ethiopia, while the sub-humid drylands are the dominating ones in Tanzania and Uganda. Burundi has a dry sub-humid zone that covers only 5% of its land area. Nearly 51% of Tanzania is relatively dry, and over two-thirds of Kenya are in arid and semi-arid areas with 33.3, 51.8, and 12.3% land subject to degradation from slight, moderate to severe. Among the West African countries, six have a large dryland region including Cote d' Ivoire, Mali, Ghana, Niger, Nigeria, and Senegal with the index ranging between <0.03 and 0.5 (Fig. 9.2).

9.2.2 Climate, Soil, Land Uses, and Land Degradation

The Köppen-Geiger climate classification of Africa indicates three climate types for the land areas: the tropical A, arid B, and temperate C, representing 57.2%, 31%, and 11.8% of the land areas, respectively. The northern and southern fringes of Africa are dominated by deserts, and tropical rainforests, grasslands, and semi-arid climates are found in the central eastern regions of the continent (Fig. 9.3).

The Environmental Systems Research Institute (ESRI) digitized Africa soil data from the UNESCO/FAO Soils Map of the World at a scale of 1:5 m (sheets VI 1-2-3). From the original 1509 soil units of the World map, Africa's 106 units were increased

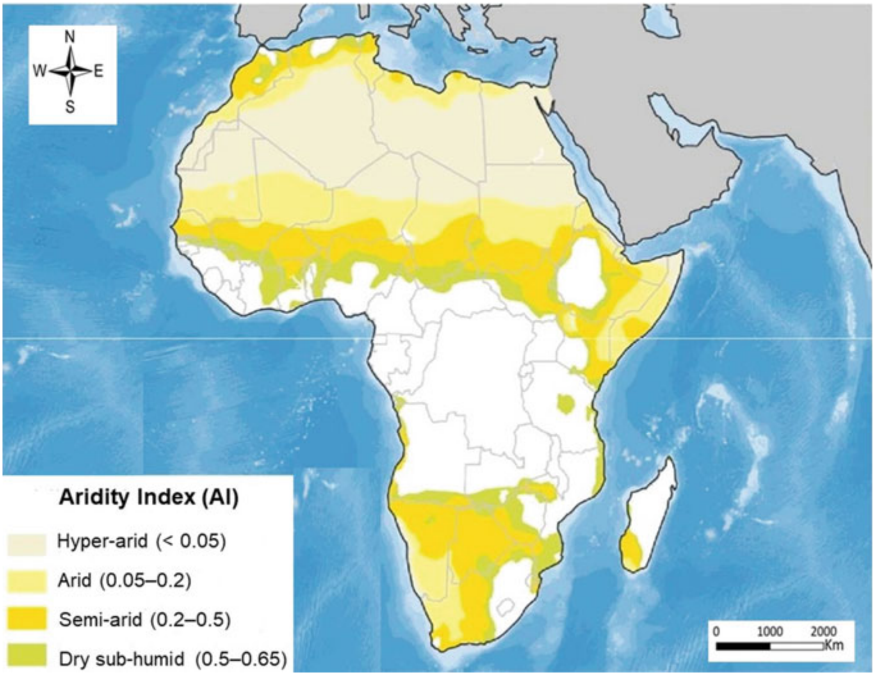

Fig. 9.2 A distribution of aridity classes in Africa based on global aridity index (AI) datasets (data from https://figshare.com/articles/Global_Aridity_Index_and_Potential_Evapotranspiration_ET0_Climate_Database_v2/7504448/3) (Trabucco and Zomer 2018). Drylands are demarcated based on the aridity index (AI), hyper-arid (AI < 0.05), arid ($0.05 < \text{AI} \leq 0.20$), semi-arid ($0.20 < \text{AI} \leq 0.50$), dry sub-humid ($0.50 < \text{AI} \leq 0.65$), and humid (AI > 0.65). Adapted from Wei et al. (2021)

to 133 to allow identification of associated soils (Batjes 2012). The major features of Africa soils are presented in Table 9.1.

The notable function of the soil in nutrient cycling. At the same time, it contributes to food production, water storage, and climate change mitigation in drylands (Safriel 2017). Soil organisms play a crucial role in the nutrient cycle of the land, contributing to its fertility and productivity through the accumulation of organic matter in the soil. Soil organisms include bacteria, fungi, insects, protozoa, worms, invertebrates, and vertebrates and all play an important role in the carbon, nitrogen, phosphorus, and water cycles by aiding decomposition processes that convert organically stored nutrients in plants like nitrogen to usable forms for living plants and maintaining the soil structure (Laban et al. 2018). The texture and structure of soil including the degree to which soil particles are bound together by organic matter, water holding properties, dissolved minerals, and oxygen contained in between the spaces are the rudimentary determinants of soil fertility (Safriel 2017). According to Parton et al. (1995), dryland soils typically have low organic carbon content due to primary productivity constrained by water scarcity, which affects the accumulation of soil organic content (SOC) and soil organic matter (SOM). Nevertheless, due to the longer residence

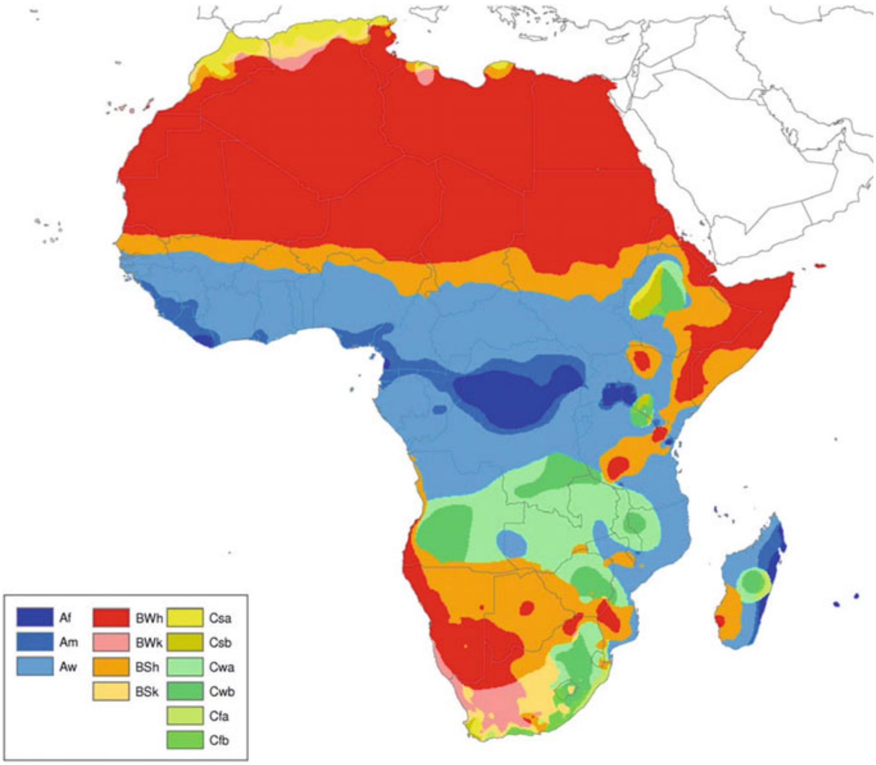

Fig. 9.3 Africa's major climates regions. The Equatorial (Af), Monsoon (Am), Tropical Savana (Aw), Warm Desert (Bwh), Cold Desert (Bwk), Warm Semi-Arid (Bsh), Cold Semi-Arid (Bsk), Warm Mediterranean (Csa), Temperate Mediterranean (Csb), Humid Subtropical (Cwa), Humid Subtropical/Subtropical Oceanic Highland (Cwb), Warm Oceanic/Humid Subtropical (Cfa), Temperate Oceanic (Cfb). *Source* Derived from World Köppen-Geiger classification (Peel et al. 2007)

time of soil carbon and low soil moisture, dryland ecosystems are quite significant for carbon sequestration and mitigation in comparison to other soils.

In Africa, land use patterns primarily depend on the population (Biswas 1986). More than 60% of the population of SSA is smallholder farmers, and agriculture contributes to about 23% of SSA's GDP (Bjornlund et al. 2020). Although 60% of the world's arable land is uncultivated in Africa (Gnacadja and Wiese 2016), but the continent's share remains low in global crop yields, therefore it has large rands in agricultural investment potential. Savannas, often known as grasslands, encompass about half of Africa, approximately 13 $\times$ 10^6 km^2 (Garrity et al. 2012). Grasslands cover the majority of central Africa, from south of the Sahara and the Sahel to north of the continent's southern point (Fig. 9.4).

Demographic growth, internal conflict, and wars with expanded refugee settlements, improper soil management, deforestation, shifting cultivation, insecurity in

Table 9.1 Main traits of African agro-ecological zones (AEZs) pertaining to land use (LU) and soil heterogeneity

FAO (AEZs)	Main soil type	Lus and agricultural systems
Hyper-arid (Kalahari, Karoo, Namib, and Sahara deserts)	Regosols and Arenosols	Oasis agricultural systems, nomadism, harvesting and hunting
Arid	Cambisols, Lixisols, and Leptosols	Millet based systems, semi-nomadism, and transhumance for livestock production
Dry and semi-arid (the Sahel and Sudan savannah)	Regosols, Solonetz, Arenosols, Lixisols, Plinthosols	Integrated crop (millet) and livestock production (agro-pastoralism) systems, transhumance sorghum, and maize
Sub-humid (Guinea Savannah)	Ferralsols, Acrisols, and Gleysols,	Agroforestry systems with sorghum, maize, root, and fruit plants
Humid (high rainforest)	Ferralsols, Acrisols, and Gleysols	Forest production (cocoa and coffee), agricultural systems with root and tuber crops
Mediterranean	Calcisols, Gypsisols, Regosols, Arenosols, Luvisols	Wheat-based system
Highlands	Ferralsols, Nitisols, Vertisols, Planosols	Grassland, pasture, cohffe, and tea plantations

Source Láng et al. (2016); Fischer et al. (2021); Garrity et al. (2012); Dewitte et al. (2013)

land tenure, variation in environmental conditions, and intrinsic characteristics of fragile soils in various agro-ecological zones are among the main causes of land degradation in Africa (Kiage 2013). According to the World Bank, Africa accounts for 65% of the total extensive cropland degradation of the world, negatively affecting ~485 × 10^6 people and resulting in ~US$ 9.3 billion annual costs (Fenta et al. 2020). Furthermore, among agricultural and non-agricultural land use, inadequate land-use planning and misuse of natural assets by the farming community, particularly poor farmers have significant negative impacts (Parikh and James 2012). For instance, 51% of the land in the Zambezi River basin, shared by Angola, Botswana, Malawi, Mozambique, Namibia, Tanzania, Zambia, and Zimbabwe is moderately degraded and 14% is highly degraded. As a result of this deterioration, soil water-holding capacity (i.e., soil water retention) and infiltration decrease, lowering the amount of water in the soil accessible for the crops. It also has the potential to undermine large-scale water storage, particularly for irrigation schemes (Abrams 2018).

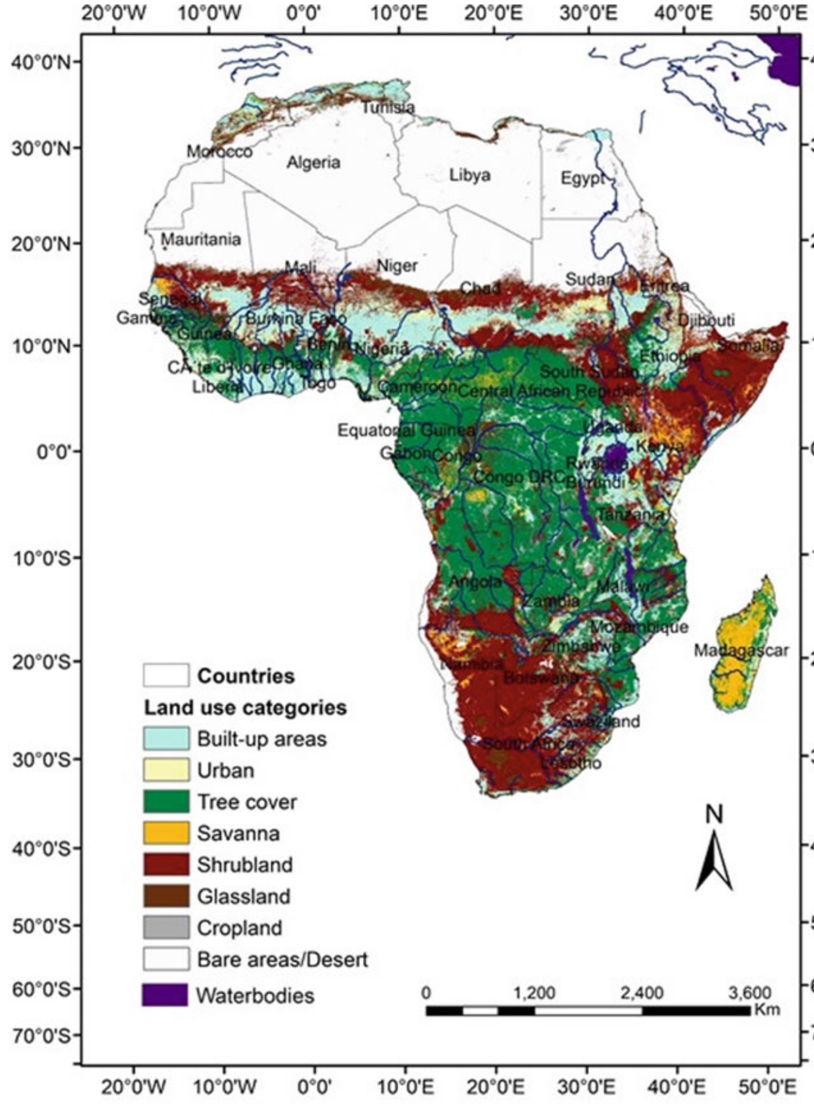

Fig. 9.4 Land use map of Africa. *Data source* European Space Agency (ESA 2017), Land Cover CCI Product User Guide Version 2

9.2.3 Water Resources

The considerable runoff typically benefits downstream regions. Rivers like the Nile, Niger, Senegal, and Orange carry water from rainy areas to arid areas that are often too arid to support life (Fig. 9.5). These high-elevation watersheds dubbed the "water towers of Africa", provide water sources for many transboundary rivers, and offer

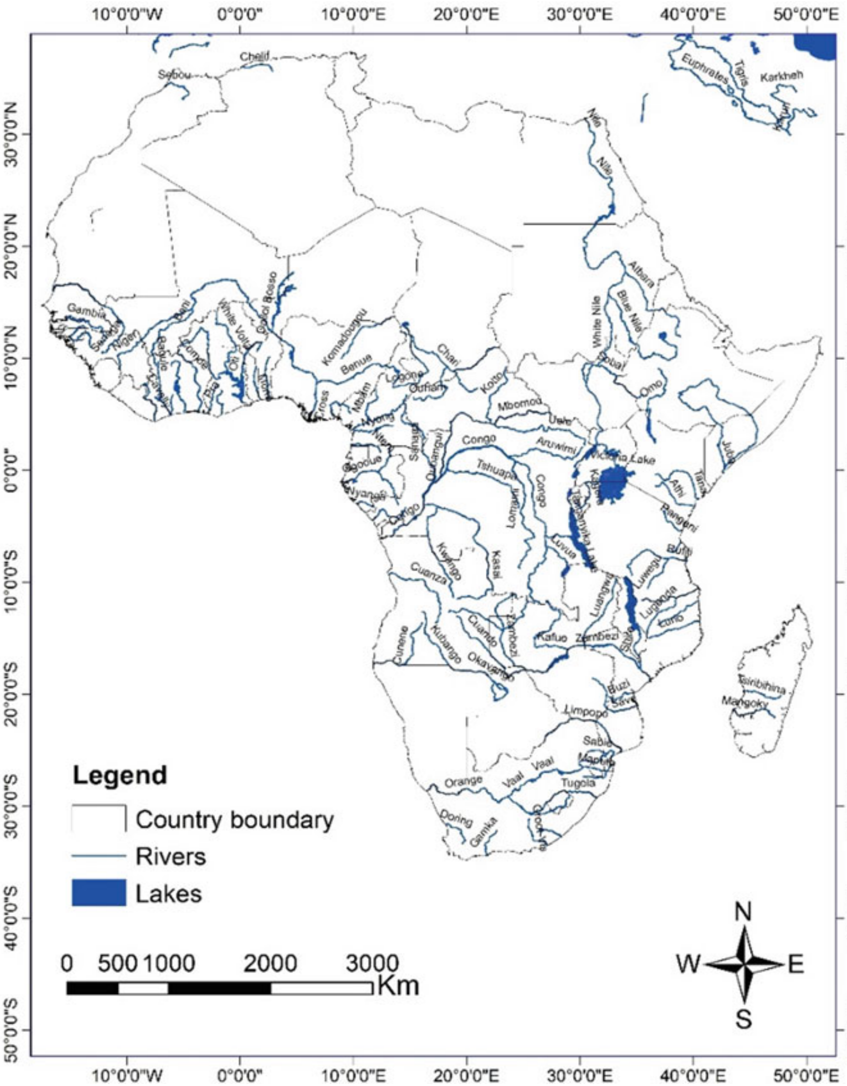

Fig. 9.5 Map of rivers and lakes in Africa. Data from WHYMAP GWR and UNESCO 2015—Bundesanstalt für Geowissenschaften und Rohstoffe (BGR) and UNESCO 2008. https://www.whymap.org/whymap/EN/Home/whymap-node.html. Accessed 10 October 2021

life-saving water to millions of people in downstream regions (MA 2005; UNEP 2010).

There are wide variations in average water availability (WA) per person among countries in the continent. For instance, the annual per capita WA for Nigeria, Africa's most populous country, is 1499.4 m^3, which is lower than that of fairly dry countries such as Botswana (5340 m^3) and Namibia (15,750 m^3) in Southern Africa. Annual per capita WA is high for countries such as Guinea (17,771 m^3), Sierra Leone (21,172 m^3), and Liberia (49,028 m^3) in West Africa; the Democratic Republic of Congo (15,773 m^3), Central African Republic (30,264 m^3) and Gabon (81,975 m^3) in Central Africa; and in the Indian Ocean Island of Madagascar (13,179 m^3). In the southern part of the continent, WA per capita is moderately low for South Africa (905 m^3), compared to North African countries such as Algeria (282 m^3) and Libya (110 m^3), as well as Kenya (618 m^3) in East Africa.

Likewise, most countries in Africa's desert regions, such as Libya, Egypt, Algeria, Tunisia, Namibia, and Botswana, usually have little rainfall and rely significantly on groundwater resources. In Botswana, for example, groundwater fulfils 80% of residential and livestock needs (Black 2016), and the same occurs in Namibia (Ndengu 2002). In North Africa, groundwater is the only source of water (Braune and Xu 2010). Some of Africa's important aquifers are losing water faster than recharge in large sedimentary basins of Lake Chad and under the Sahara Desert (Pham-Duc et al. 2020).

9.2.4 Understanding Dryland Biodiversity as a Basis for Ecosystem Services in Africa

Ecosystem multifunctionality (i.e., the interaction between biodiversity and different ecosystem functions) is context-dependent (Hu et al. 2021). Africa is home to a wide range of biodiversity and about one-quarter of the world's biodiversity is found in this region. Africa is also home to some of the largest intact concentrations of large mammals, which graze freely throughout many countries. African dryland biodiversity is unique and varies among ecosystems, species, genetic, and functional diversity (Bonkoungou 2001). The instant main threats to dryland biodiversity in Africa are the ecosystem and habitat degradation caused by new and powerful forces of environmental deterioration such as climate change, deforestation, desertification, mining operations, poverty-induced overexploitation of natural resources, and increased wildlife trade (Fig. 9.6) (Archer et al. 2021). Within each relative aridity category, the dryland ecosystems and species are highly heterogeneous with wide variations in topographic, climatic, geological, and biological situations and the most limiting factors are soil nutrients and hydrological resources. For instance, the findings of Hu et al. (2021) revealed that the microbial diversity (e.g., fungi) is positively associated with multifunctionality in more arid regions, whereas in less arid regions, there is a strong positive correlation between soil and species richness multifunctionality. Besides, the dryland biodiversification also comprises the seasonal rainfall pattern, fires, and herbivore pressure (Venter et al. 2017).

The dryland biodiversity in Africa mostly consists of Mediterranean systems, the Southern Africa region, the Saharan, and the Sahel of Africa. In 2014, 3148 plants

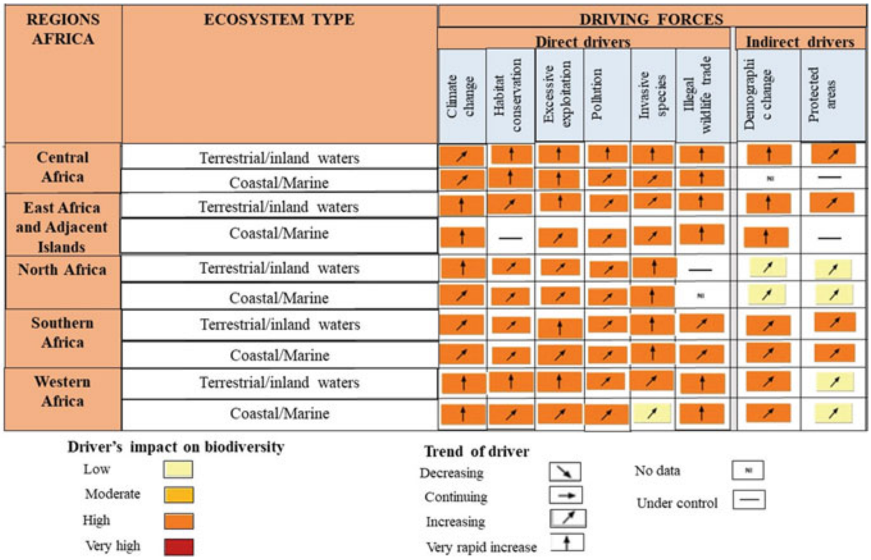

Fig. 9.6 Trends in biodiversity, drivers associated with them per subregion, and ecosystem type

and 6419 animals in Africa were recorded as threatened with extinction on the International Union for Conservation of Nature (IUCN) Red List. Although 21% of all freshwater species are threatened, 58% of freshwater plant species and 45% of freshwater fish are over-harvested (Dulvy et al. 2014; Darwall et al. 2011). In addition, over the past two decades, African birds showed a decline, resulting in a high probability of extinction. The trends for other groups also revealed a negative correlation. Since 1970, the combined population of African vertebrate species have declined by around 39% (McLellan et al. 2014). Declines are more rapid in Western and Central Africa than in Eastern or Southern Africa (Craigie et al. 2010). On the contrary, smaller species' population trends (e.g., insects) are often unknown (UNEP-WCMC 2016). In 2020, the population of Africa grew by 2.49% (about 1.36 billion inhabitants) compared to that in the previous year (Anoba 2019), which results in an increased demand for natural resources, leading to land-use changes and unsustainable species utilization such as wetland drainage for agriculture, inappropriate and unsustainable fish harvesting (UNEP-WCMC 2016). As a result of these changes, natural areas, biodiversity, and ecosystem services provided by natural habitats are all under threat. Furthermore, the water contamination from excessive nutrients, domestic and industrial organic loads, pesticides and heavy metals, and the impacts of invasive species lead to the depletion of biodiversity in freshwater habitats, especially in East Africa's Lake Victoria, the Mediterranean and Atlantic coasts of Morocco, and numerous major African rivers (Darwall et al. 2011).

Biodiversity Conservation and Preservation

Many dryland societies have strong values of environmental custodianship and a rich knowledge of their environment and rely heavily on a range of biodiversity (Mortimore et al. 2009). Re-enabling communities to use this knowledge is a powerful way to enhance biodiversity and build resilience in Africa. Restoring biodiversity through ecological restoration contributes to major gains in ecosystem services (Cowie et al. 2011; Bonkoungou 2001). For instance, soil biodiversity is critical for the supply of ecosystem services, and its protection is central to achieving Land Degradation Neutrality in Africa (Von Maltitz et al. 2019). Sustainable land management (SLM) practices protect the ecosystem functions that sustain productivity. Clearing land for cultivation may initially increase food production, but it comes at a significant cost in terms of water supply, climate regulation, carbon sequestration, forest resources, pollination, and many more services. Vegetation cover can play a major role in reducing surface flows of water and improving infiltration of water. In return, soil biodiversity improves both infiltration and water storage in the soil (Cowie et al. 2011; Collentine and Futter 2018).

However, biodiversity conservation is not the exclusive preserve of environmental and wildlife agencies. Instead, a shared responsibility of many sectors, including agriculture and water is the key. Sustainable agriculture offers one of the most important ways to achieve sustainable development by simultaneously protecting biodiversity and ecosystem services, raising agricultural productivity, and promoting the resilience of people and ecosystems (Adenle et al. 2019; Agula et al. 2018). SLM practices often rely on protecting biodiversity to boost soil organic carbon, soil nitrogen, and soil moisture. Practices like agroforestry and low tillage agriculture are based on indigenous practices that have been revived and improved to protect soil moisture and fertility of croplands as well as provide supplementary benefits. Other SLM practices, such as contour bunds and zai, also contribute to building up soil moisture and organic matter to improve productivity and resilience (Cordingley et al. 2015; Adimassu et al. 2016). In Africa, there is a need to motivate new approaches and actions such as conserving, valuing, restoring, and preserving biodiversity and ecosystems. Considerable effort and endeavor are needed to promote and consolidate these approaches between all sectors.

9.2.5 Socio-economic Development Indicators

Economic growth will lessen the proportion of people in Africa's drylands who are vulnerable to droughts and other stressors. However, it may not offset the effects of population growth. By 2030, the population living in rural areas of the dryland countries is projected to grow by 15–100% (Morris et al. 2016). By 2050, the human population in SSA will double (UNDP 2015). About 70% of Africans rely on dry and sub-humid lands for their livelihoods (UNEP 2007). With 10–20% of drylands already degraded, these drylands, which are mostly used for cattle production, are highly vulnerable to land degradation (FAO 2009).

In the past four decades, population growth in SSA was correlated to the corresponding economy with a rise in the number of people living in extreme poverty (Burian et al. 2019). Since the early 1990s, SSA's poverty rates have remained steady, and half of the global population now lives on less than US$ 1.90 day^{-1} (World Bank Group 2016). Africa's GDP is projected to gradually recover by ~3.4% and ~3.7% in 2021 and 2022, respectively, after shrinking by 2.1% in 2020 (IMF 2021). The per capita GDP is estimated to have contracted by 10% in nominal terms in 2020, which makes it insufficient to accelerate the socio-economic growth and reduce the poverty (Fig. 9.7). Drylands and poverty are interlinked at all scales of geography, from regional to subnational (Middleton and Sternberg 2013).

The COVID-19 pandemic will also exert a greater effect on African living conditions and socioeconomic prospects. In Ethiopia, Malawi, Nigeria, and Uganda, for example, the crisis is evidenced by increased unemployment, poverty, and inequality (Josephson et al. 2020). The health and economic problems facing dryland residents in Africa threaten to overwhelm the healthcare system, undermine livelihoods, and stain long-term prospects for economic growth.

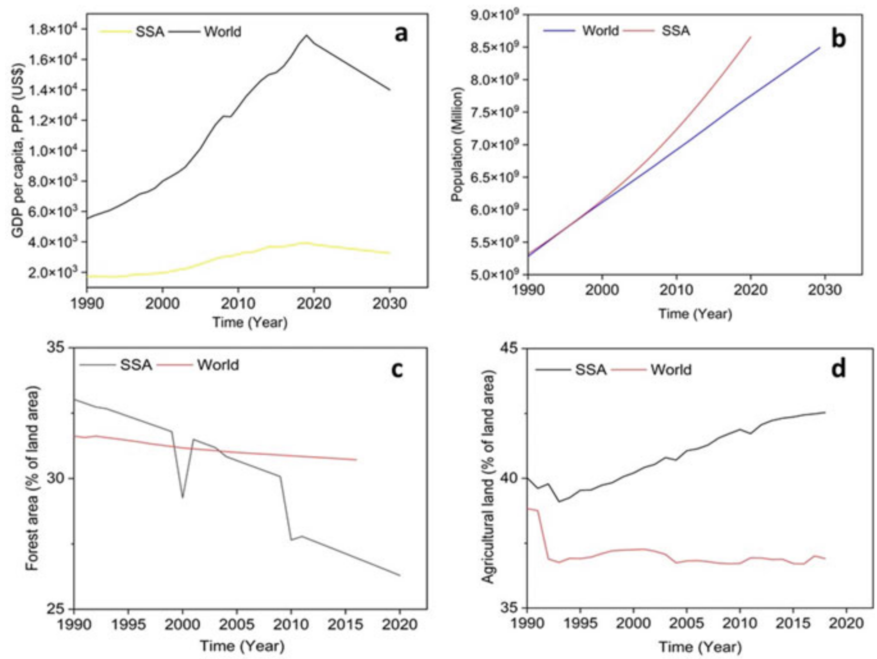

Fig. 9.7 Global and Sub-Saharan Africa (SSA) development indicators from 1990 to 2020, **a** GDP per capita, purchasing power parity (PPP), **b** population, **c** forest area, and **d** agricultural land. *Source* International Monetary Fund (IMF) database (World Bank 2021)

Table 9.2 The extent of African dryland

Sub-region	Hyper-arid		Arid		Semi-arid		Dry sub-humid		Total
	km^3	%	km^3	%	km^3	%	km^3	%	km^3
Central Africa	0	0	6	0	66	2	144	4	216
Eastern Africa	878	14	1670	27	1768	28	767	12	5083
Southern Africa	96	2	823	13	2579	42	924	15	4422
Western Africa	2363	33	1465	20	1278	18	514	7	5620
Total	8072	27	4604	16	6100	21	2392	8	21,170

Adapted from Koohafkan and Stewart (2008); UNDP/UNSO (1997)

9.3 Changing Aspects of African Drylands

9.3.1 Dryland Dynamics in the Past Decades

In the last four decades, the overall dryland temperature rose at a rate of 0.032 °C·year^{-1} and precipitation dropped at a rate of 0.074 mm·month^{-1}·year^{-1}, respectively. Although precipitation generally decreased over the drylands, summer precipitation increased over southern Africa as well as northern Africa's dryland areas (Daramola and Xu 2021). High precipitation years in southern Africa caused an initial spike in fire rates, which then declined in the subsequent years (Wei et al. 2020). Spatially, the North and Southern parts of Africa are dominated by hyper-arid regions (e.g., Sahara, Kalahari, and Namib desert) (FAO 2019). Most African regions are dominated by drylands, of which dry sub-humid, semi-arid, arid, hyper-arid dryland account for 8%, 21%, 16%, and 27%, respectively (Table 9.2).

9.3.2 Structure and Functions

It has been widely accepted that dryland function and structure are influenced by climatic factors (Maestre et al. 2016). The structure and function of drylands can be traced down to the basic processes that underpin the dryland ecosystem services that provide benefits to people (Hoover et al. 2019). Africa drylands ecosystem structural and functional dynamics consist of carbon dynamics, woody plant increase, and change in vegetation greenness (Ross et al. 2021). Therefore, in order to understand dryland carbon sequestration potentials, it is crucial to monitor spatiotemporal dynamics of dryland structure and function.

Auditing vegetation greenness is a key measure of photosynthetic activity and subsequent availability of green biomass for livestock feed (Diouf et al. 2015). Lu et al. (2016) indicated that over recent decades, changes in vegetation greenness were spatially diverse in African drylands despite overall greening trends (Wei et al. 2019). Over the last decades of the twentieth century, the field data and satellite observations commonly revealed a positive trend in vegetation greenness and rainfall across much of the Sahel (Brandt et al. 2015). Moreover, in recent decades, East Africa has been identified as a hotspot of vegetation browning driven by rising soil water shortage (Wei et al. 2019). As a result, in water-stressed southern Africa, vegetation greening has become common, which is linked to an increase in woody cover fueled by abundant rainfall and enhanced by CO_2 fertilization (Venter et al. 2018). Similarly, woody plants are becoming more common in African drylands (Stevens et al. 2017). In Southern Africa, for example, woody encroachment is displacing herbaceous vegetation (Skowno et al. 2017).

It has been observed that the Sahel's woody vegetation is shifting towards drought-resistant shrubs at the expense of forests (Brandt et al. 2015). The trend of rising leaf area index/vegetation greenness in African drylands mirrors the extensively reported increase in woody plant cover in tropical arid regions worldwide (Tian et al. 2017). Meanwhile, estimations of woody cover obtained from vegetation optical depth (VOD) have shown significant increases from 1992 to 2011 (Brandt et al. 2017). Further, from 2010 and 2016, there was a carbon loss in African drylands (Fan et al. 2019). With a climax in the unusually rainy year of 2011, African drylands were identified as a major carbon sink (Yue et al. 2017; Poulter et al. 2014). Consequently, due to intense El Niño event of 2015–2016 led to drylands being classified as a carbon source (Brandt et al. 2018), but the detailed mechanisms underlying this transition need to be investigated (Yue et al. 2017). In 2017, the carbon stock of African drylands (i.e., shrublands and savanna) had nearly restored to the pre-El Niño 2015–2016 levels (Wigneron et al. 2020). African drylands carry a huge amount of the continent's carbon stock, with soil carbon accounting for a significant portion (Robinson 2007). Nevertheless, widespread woody plant increase is often connected with an upsurge in aboveground carbon biomass (Venter et al. 2018). There is thus a need for observational evidence to identify the effects of woody intrusion on soil carbon (Mureva et al. 2018).

9.3.3 Ecosystem Services, Human Well-Being, and Resilience

Vegetation shifts and ecosystem resilience-related studies are increasing over time, which lead to accurate measurement at larger areas and enrich human understanding of the changes in the terrestrial ecosystem, carbon exchange system, and climate-biosphere interactions in the environment (Bao et al. 2014; Zhang et al. 2003). The usage of remote sensing approaches such as the Gross Primary Productivity (GPP) and evapotranspiration (ET) is becoming feasible to evaluate the ecosystem Water Use Efficiency (eWUE) and ecosystem resilience. The assessment of eWUE centered

Table 9.3 Ecosystem resilience using dimensionless ecosystem Resilience Index to drought (eRId)

No.	Resilience status	Range
1	Resilient	>= 1 *eRId*
2	Slightly non-resilient	0.9 <= *eRId* < 1
3	Moderately non-resilient	0.8 <= *eRId* < 0.9
4	Non-resilient	*eRId* < 0.8

on GPP and ET can be essential to understand the ecosystem carbon–water coupling (Sun et al. 2018), which further provides information for the accurate prediction of ecosystem resilience and ecosystem management (Huang et al. 2016). A recent study on eco-hydrological resilience over Africa using coarse spatial resolutions showed that 31.22% of the terrestrial ecosystems were non-resilient to ecosystem shifts (Kayiranga et al. 2020).

Ecosystem resilience to drought in the Horn of Africa (HA) (A case study): This case study highlights and demonstrates the status of the eWUE and ecosystem resilience to drought in the HA with higher spatial resolution and specific focus. The eWUE was extracted from daily Global GPP and annual ET datasets using MOD17 algorithm. The study generated the annual eWUE from the fraction of average annual MODIS GPP to the average annual ET for 15 years. The ecosystem resilience calculation was conducted using the dimensionless ecosystem Resilience Index to drought (*eRId*) from the ratio of mean values of multi-annual eWUE to the annual eWUE of the driest year as initially defined by Sharma and Goyal (2018), which was further applied in other studies (Sharma and Goyal 2018; Guo et al. 2019). The driest year of high drought severity in the Horn of Africa, i.e., 2009, was identified from the spatial and temporal patterns of the high-resolution annual SPEI images, which was consistent with the UN *Emergency Events Database* (EM-DAT) record of drought (EM-DAT 2018). Table 9.3 shows the four major classes of the *eRd*.

Figure 9.8 shows the spatial distribution maps for average annual GPP, ET, and eWUE across the HA during 2000–2014. The highest values of GPP and ET were concentrated in southwestern highlands of Ethiopia and Kenya with less extents in its southeastern coast. The accuracy evaluations of the mean annual GPP were achieved using GPP_{EC} from Global FLUXCOM observations with R^2 and RMSE values of 0.78 and 5.4 g C m^{-2}, respectively. By comparing the average annual MODIS-ET values to the annual averaged ET from the monthly evapotranspiration data of USGS early warning system for the study area, R^2 of 0.76 was obtained.

The mean annual eWUE in the HA was 1.58 g C kg^{-1} H_2O, which indicated a large spatial variability with a standard deviation of 0.51. The highest mean annual eWUE was in most regions of Ethiopia followed by Eritrea, while Somalia and Djibouti had the lowest mean annual eWUE. Based on Zonal statistics extractions using the ESA land cover types, the highest mean annual eWUE was in the croplands and forestlands, and the least mean annual eWUE was in the sparse vegetation and wetland areas. The shrubland, which had the overwhelming land cover (38.9%) in the HA, had relatively lower mean annual eWUE, whereas the grassland and sparsely vegetated lands had higher standard deviations and eWUE values.

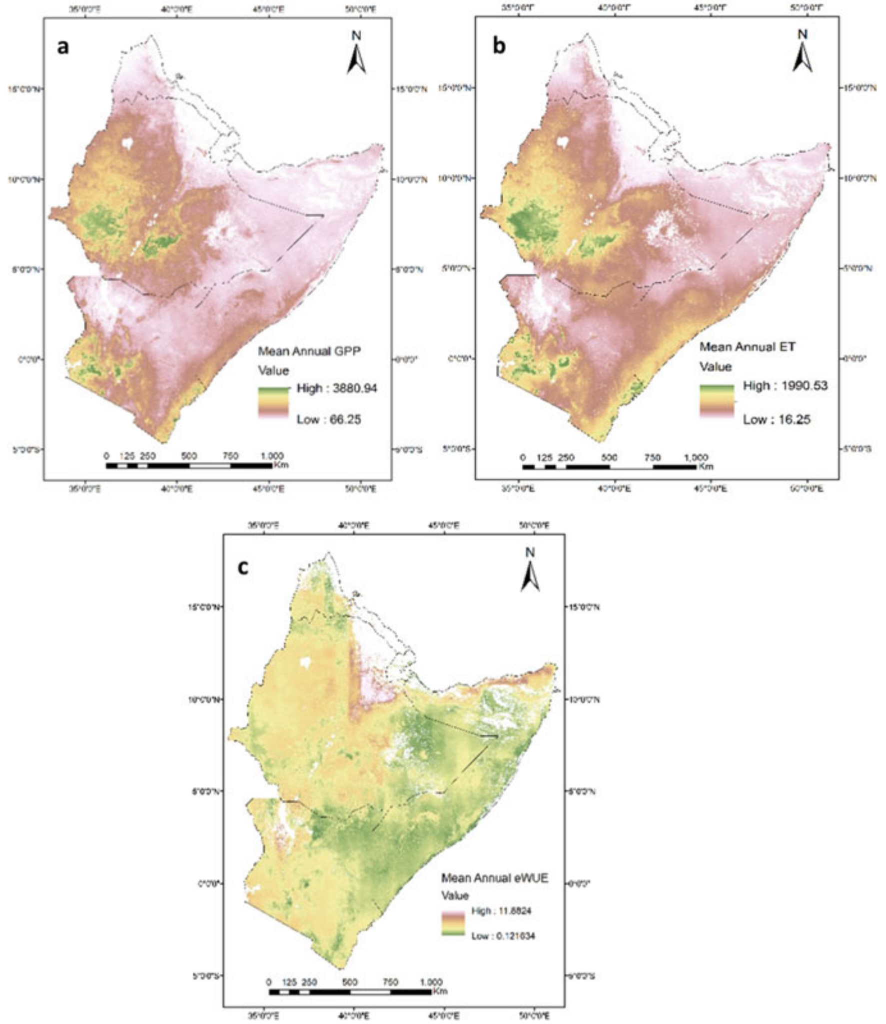

Fig. 9.8 Mean annual distributions of **a** GPP (g C m^{-2}), **b** ET (mm), and **c** eWUE (g C kg^{-1} H_2O) during 2000–2014 in the HA

Figure 9.9 shows the ecosystem resilience to drought (eRd) during 2000–2014. Overall, 54.9% of the study areas were found to be resilient to drought. Most of the resilient ecosystems were in the central highlands of Eritrea, southeast of Ethiopia, northeast Kenya, and large parts of Somalia. In contrast, 32.6%, 9.6% and 2.8% of the regions were non-resilient, moderately non-resilient, and slightly non-resilient, respectively. The strictly non-resilient ecosystems were mainly observed in southeast parts of Kenya, south-west of Eritrea, and areas near the triple junction of Ethiopia, Djibouti and Somalia.

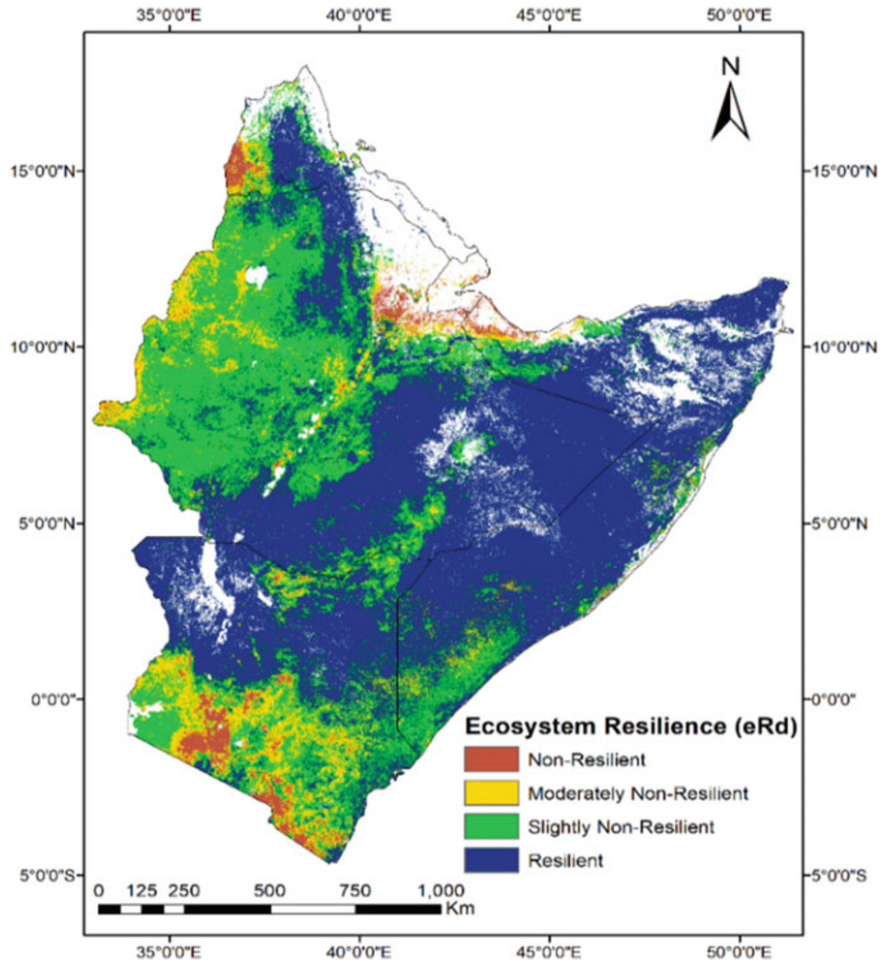

Fig. 9.9 Ecosystem resilience to drought (eRd) in the HA during 2000–2014

The aggregation of the final ecosystem resilience with multiple land cover types and agroecological zones of the region showed that the cropland and wetland were slightly non-resilient to drought with mean eRd values of 0.97 and 0.99, correspondingly. On the contrary, grassland and sparse vegetation were relatively the most resilient to drought with mean eRd values of 1.12 and 1.1, respectively. Likewise, the tropic warm-humid (eRd = 0.92) and the tropic cool-humid (eRd = 0.94) were slightly non-resilient to drought, whereas the tropic warm-arid (eRd = 1.13) and cool-arid agroecological zones showed relatively the highest resilience to drought in the region.

Even though the HA was considered as a drought-prone region, the final map of ecosystem resilience (Fig. 9.9) showed that 54.9% was resilient to drought, while

32.6% was completely non-resilient. This is mostly in agreement with the recent study on eco-hydrological resilience to ecosystem changes over the African continent (Kayiranga et al. 2020). However, this research showed that cropland and wetland were slightly non-resilient ecosystems to drought conditions rather than the savannahs and barren lands. The ecosystem resilience to drought revealed that the warm-humid and cool–humid agroecological zones were slightly non-resilient to the most severe drought conditions; this indicated the vulnerability of these ecosystems to the warming trends and climate variability impacts in the region. The variations in eWUE provide useful spatial and temporal information for policy and decision-makers and can play a vital role in rangeland and ranch management, vegetation degradation protection and management, and drought and climate change mitigation at national and regional levels.

9.3.4 Livelihoods and Food Security of Local Communities

The sustainability of livelihood in Africa's drylands is being jeopardized by a wide variety of environmental, political, and socioeconomic changes that are all intertwined (Fraser et al. 2011). The livelihoods of local communities are inextricably linked to the landscapes in which they live, which are especially sensitive to changes in these environments (Shackleton et al. 2019). Changes in livelihood activities could have detrimental consequences for ecological services. Environmental and socioeconomic development are putting increasing pressure on rural regions across much of Africa (Suich et al. 2015). Eventually, the food security both in developing and developed countries (e.g., the case of Tigray crisis) is compromised by political instability, conflicts or economic crises (García-Díez et al. 2021; Peng et al. 2021). Importantly, climate-related shifts are commonly overlaid on and feedback on a wide range of cross-scale socioeconomic stresses that relate to social vulnerability in the first instance (Niang et al. 2014). The detailed cross-cutting dimensions of resources and local livelihoods are shown in the Fig. 9.10.

As the livelihoods of most people in drylands in Africa depend upon natural resource-based activities, including agriculture and animal husbandry, the capacity of the natural resources to generate stable and sufficient incomes is increasing (de Haan 2016). Hasty demographic growth increases the pressure on dwindling resources, creating conditions of extreme weather events, food price spikes, or other exogenous shocks that can trigger acute humanitarian crises and disasters and fuel violent social conflicts. Many households in Africa's drylands turn to unsustainable practices to address pressing short-term needs, resulting in significant land degradation, water scarcity, and massive biodiversity losses (Cervigni and Morris 2016). Vulnerability is thus expanding as a result of complex interactions between several causes, compromising the long-term livelihood prospects of hundreds of millions of people. Climate change is anticipated to compound the situation by increasing the frequency and severity of extreme weather occurrences (IPCC 2021). Food prices are predicted to

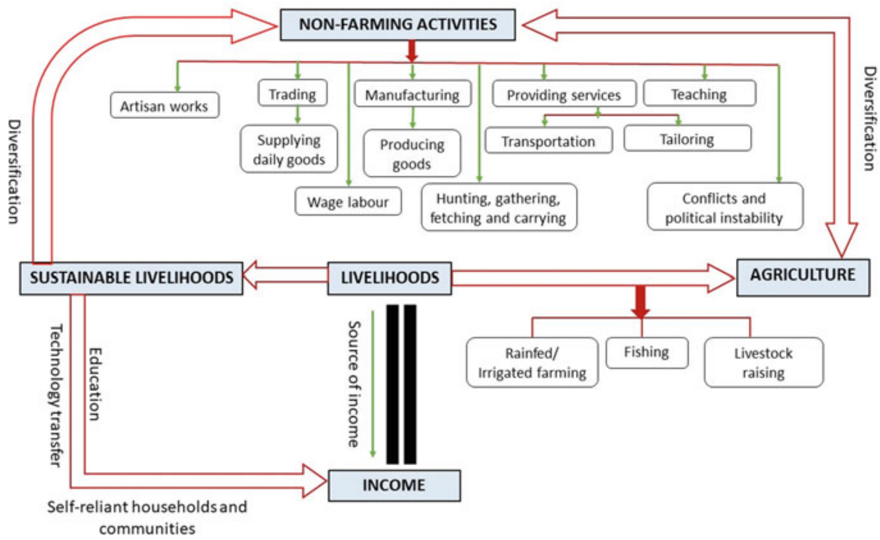

Fig. 9.10 Cross-cutting dimensions of rural resources and local livelihoods

rise dramatically, making food affordability an increasing concern for many African societies (Scholes et al. 2015).

SSA is plagued by food insecurity, with some cases reaching catastrophic proportions, for example, in the HA regions and southern Madagascar (Hirwa et al. 2021). Food insecurity is not just about availability, unsatisfactory food production, and intake, it is also about the poor food quality and nutritional value (Sasson 2012). Food riots and escalating food prices are two of the numerous signs of the current food crisis and instability. Nowadays, the key roots of food insecurity are inadequate food production and the influence of the COVID-19 crisis (Ayanlade and Radeny 2020). Africa's governments have enacted initiatives to promote staple crop cultivation and increase the productivity of local farmers, particularly smallholders.

9.3.5 Dryland Conservation and Effective Practices

The major conservation issues for drylands of Africa are habitat loss or degradation and habitat fragmentation, largely caused by agriculture, charcoal production, and infrastructural development (Githiru et al. 2017). Effective conservation methods are required to maintain crop output in arid regions (Hammel 1996). Adaptability and conservation strategies can help to offset dryland ecosystem service losses. Although the conservation status of dryland biodiversity is not well monitored, several recognized drivers of biodiversity of the African drylands have declined. As a result of population shifts and urbanization, agricultural expansion, land-use changes, weakening governance arrangements, and the introduction of alien invasive species, the declination of these drivers will become more obvious (IUCN 2012). It is crucial to foster sustainable intensification approaches based on conservation agriculture and community-based adaptation, with functioning support services and market access including the introduction of adapted cultivars (Mbow et al. 2014). In addition, identifying sustainable land management practices (e.g., agroforestry, crop rotation, and intercropping systems) for enhanced land-based climate change adaptation and mitigation (i.e., food production, biodiversity, GHG emissions reduction, soil carbon sequestration) is also important (Sanz et al. 2017; Francis 2016). Attention should also be paid to the food-energy-water-biodiversity-health (FEWBH) nexus, particularly water usage and re-utilization efficiency as well as the rainwater management (e.g., water harvest practices) (Albrecht et al. 2018; Hirwa et al. 2021) and water-energy-food (WEF) nexus security (Muhirwa et al. 2023). Obviously, establishing institutional designs centered on youth and women through new economic models that facilitate access to credit and loans to enact policies that balance cash and food crops will be beneficial (Palacios-Lopez et al. 2017). Last not least, enhancing local expertise, culture, and customs while exploring dryland ecosystem management innovations should be enhanced.

Uncertainties are quite crucial in agriculture because they influence decision-making and might potentially lead to inefficiencies as well as food poverty (Thornton and Wilkens 1998). Mortimore and Adams (2001) highlighted five key elements of the 1972–1974 drought catastrophe. Specifically, diversification of livelihood and crops, migration, negotiating the rain, managing biodiversity, animal integration, off-farm income-generating activities, and livestock integration were all prevalent among the mix of resilience techniques identified in the literature (Batterbury and Forsyth 1999; IPBES 2021). Additionally, as a reflection of the diversity, farmers grew multiple varieties of the same crop on the same field at the same time as insurance against future risks, which was a demonstration of system resilience (Jellason et al. 2021).

Diversification within and without agriculture has been used as a resilience management approach to help farmers endure extreme weather conditions (Ayana et al. 2021). In West Africa, household heads were discovered to be the decision-makers in terms of diversifying income sources (Ifeoma and Agwu 2014). Apart from livelihood diversification, food sources and farming systems were also diversified to serve as insurance against pest and disease infestations that could lead to losses or for balanced nutrition (Jellason et al. 2021). Research conducted in west Africa also illustrated the efficiency and the flexibility of livelihood and farming systems through the rationing of family labor for priority farm operations, which were determined by the variability of rainfall as to what and when to grow (Mortimore and Adams 2001). Some authors asserted that the current resilience strategies displayed by African smallholders were insufficient to tackle climate change impacts due to new dimensions of challenges such as increased poverty, population growth, and food insecurity (Awazi et al. 2021).

By 2030, structural changes driven by economic growth will enable a few dryland dwellers in Africa to switch to off-farm livelihood and thereby reduce their vulnerability. Many more will continue to depend on animal husbandry and the cultivation of the land. Advanced agricultural production technology can generate significant resilience improvements by enhancing the productivity of rain-fed agriculture. If nothing is done, households that depend on agriculture and are susceptible to droughts and other crises are anticipated to rise by roughly 60% in the Sahel and the HA by 2030. Interventions to improve the productivity of rain-fed crops can significantly mitigate this increase (Cervigni and Morris 2016). Enhanced agricultural production technology, soil fertility management, and the incorporation of trees into existing farming systems can all provide resilience benefits by increasing yields and crop drought and heat tolerance. Trees growing in farming fields, in fact, can function as fertilizer providers while also lowering crop water and heat stress. Trees can also improve household food security by providing food when crops and animal-source meals are in short supply, as well as increasing coping ability by offering assets that can be cut and sold in times of need.

In West Africa, among the current strategies for managing the resilience of arid zones, there is the very ambitious project of the Great Green Wall (GGWI). The GGWI's overarching goal is to combat desertification using established principles of sustainable land management, as well as the enhancement and preservation of natural resources and production and management systems. Through multipurpose activity platforms, transition is achieved while guaranteeing the socioeconomic development of local communities. For example, the Samise implements aim to (1) create income, (2) improve access to basic needs, (3) oversee the transition to a circular economy as a way to foster the emergence of rural production sites, (4) consolidate ecological sustainability to eradicate poverty and food insecurity, and (5) boost local population adaptation and resilience abilities (Diop et al. 2018). Since its creation in 2005, the Pan-African Agency of the Great Green Wall (with a total distance of ~7,775 km and total area of ~11,662,500 ha) has set up successful examples in its headquarter of Nouakchott. Now, it has transformed the arid areas of the Sahel into sustainable development hubs, which are integrated into the national economic fabric, an essential action for the entire Sahel region of West Africa (Fig. 9.11).

Furthermore, due to the fragility of Sahelian environment and its ability to adapt to climate variability and change, the choice of species to be reforested at the GGWI level is also influenced by two other factors: (1) they must not be edible to local fauna, and (2) they must have ecological relevance and economic worth (e.g., fruit and gum arabic production). The species should also meet criteria such as resistance to water stress, adaptability and plasticity, and multiple uses and utilities as perceived by local populations (Diop et al. 2018). The potential benefits from the construction of the GGWI to resolve biodiversity losses, environmental degradation, desertification, and climate change will have a legitimate chance of success if they are coherent with significant matters related to local communities' livelihoods such as satisfaction of domestic needs in terms of wood and non-wood products, raising

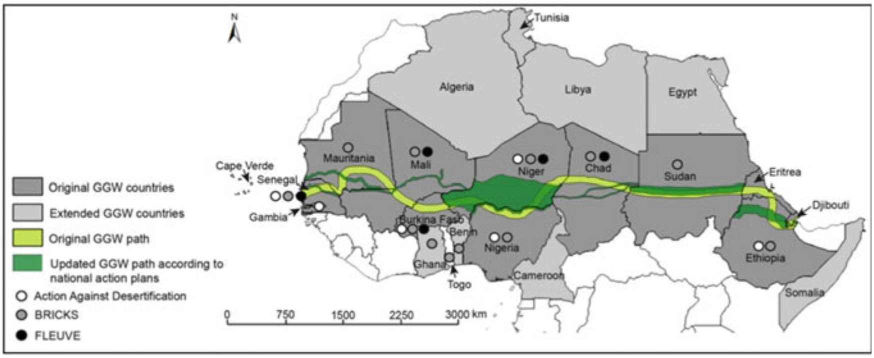

Fig. 9.11 The GGW situation map and participating countries. The Sahel and West Africa Program in Support of the *Great Green Wall* Initiative (SAWAP) consists of several transnational projects, e.g., Building Resilience through Innovation Communication and Knowledge Services (BRICKS) and Front Local Environmental pour une Union VertE (FLEUVE). Adapted from Goffner et al. (2019)

of household incomes through promotion of sustainable income-generating activities, and commitment to sustainable income-generating activities. Figure 9.12 shows exemplar tree nursery beds for the purpose of combating desertification in Senegal.

From this vantage point, "polyvalent" village gardens, as beneficiaries and focal points for practically all domestic and rural population activities, are a meaningful

Fig. 9.12 Nursery grounds in north Senegal—photo by A. Guisse, June 2009

method that is congruent well with GGWI program's goals. They supplement a regulated concentration of many activities that are frequently part of the inhabitants' ordinary everyday life. They may include new activities that rely on resources, as well as local dynamics and demographic proximity that allow for the formation of circumstances for self-sufficiency in the supply of services such as food, housing, medicinal plants, and other socioeconomic items (NAGGW 2016).

Dryland conservation practices in Ethiopia: The dryland area covers over 70% of the landmass in Ethiopia (Amanuel et al. 2019). According to a recent FAO guideline, dry forests account for 80% of Ethiopia's forests, and are a crucial element of Ethiopia's tropical forests, which span from the lush alpine forests of central Ethiopia's Bale Highlands to the hot and dry woods of southern Ethiopia's Borana rangelands (Atmadja et al. 2019). The dryland agro-ecological zone including the dry sub-humid areas of the country are under continuous human and livestock pressures, which are vulnerable to the effects of climate variability and climate change. In Ethiopia, traditional conservation practices reverted to 400 BC; however, established conservation activities such as soil and water conservation (SWC) became effective since the 1970s (Fig. 9.13) (Haregeweyn et al. 2015). Soil erosion (i.e., gully, rill, and sheet erosion) remains as one of the main environmental problems in large parts of Ethiopia, and it is likely to deteriorate with the predicted increase in population and climate variability in the twenty-first century (Field and Barros 2014). The SWC practices in Ethiopia showed mixed results as affected by the type of intervention involved and the agroecology considered for implementation, yet the relative performance of the interventions was effective in the dryland areas compared to the humid lands of the country (Haregeweyn et al. 2015).

The Federal Democratic Republic of Ethiopia has carried out some on-site conservation practices to conserve and promote sustainable utilization of its forest genetic resources in the dry forests managed by the civil society and/or the government (Atmadja et al. 2019). The Ethiopian Biodiversity Institute (EBI) has saved 2,000 accessions of 260 species of trees in its gene banks and created 15 in vivo locations in Ethiopia's three regional states (Atmadja et al. 2019). According to the EBI, a reduction in biodiversity at various spatial and temporal scales has become a concern in the country, necessitating national biodiversity protection and initiatives.

Fig. 9.13 Example of Ethiopia's common SWC measures; gully plugs constructed across gullies. **a** Gullies, **b** before (2012) and **c** after (2013) the interference. *Source* Haregeweyn et al. (2015)

Ethiopia has been very proactive in implementing the Participatory Forest Management (PFM) for conservation of its natural forests and forestry restoration. However, the rate of forest gain has been approximately 19,000 ha annually largely in the Dry Afromontane areas during 2000–2013, one-fifth of the annual forest loss in the country (Johnson et al. 2019).

An inception appraisal regarding the landscape restoration in Ethiopia's drylands elaborated that ~1×10^6 ha of degraded land were restored in northern Ethiopia over the past two decades (Sola et al. 2020). The main restoration practices and techniques implemented in the drylands of Tigray included area exclosures to enable for natural vegetation regeneration, conservation tillage, and water harvesting as well as building of small dams to hold water for infiltration or irrigation, tree planting, and pasture extension (Sola et al. 2020). Gebremeskel Haile et al. (2019) also highlighted the success and exemplary conservation practice In the Abraha Atsbaha watershed (Tigray, Ethiopia), where drought-prone degraded areas were converted into well-established sustainable landscapes as the groundwater levels amplified. The sustainable agricultural development practices have significantly contributed to diet self-sufficiency and economic benefits. The Government of Ethiopia built a dryland agriculture bureau to support research and development in the drylands, where the majority of Ethiopia's food is produced. A modern dryland management agenda calls for more participatory and collaborative planning and design of area enclosures, an unified landscape strategy engaging many sectors, and an endeavor to achieve socioeconomic sustainability guided by both professionals and knowledge systems (Sola et al. 2020).

9.3.6 Dryland Nature-Based Solutions (NbS) for Sustainable Management

NbS are facing challenges in semi-arid and arid lands in Africa including climate change, water security, food security, human health, socio-protection, socio-economic development, disasters, ecosystem degradation, and biodiversity loss (IISD 2022). NbS benefits and ecosystem services in African drylands include providing clean water to communities; maintaining diversity of plants and animals which are crucial for resilience to changes and shocks; stabilizing the soil while ensuring good quality soil and enhancing the carbon sequestration on agricultural lands and peatlands; providing flood control and regulate the quality of water; and promoting the aesthetic, spiritual and human well-being benefits such as ecotourism for the country and improved livelihood (Thorn et al. 2021). Moreover, NbS provide means for DSESs to successfully navigate the linkages between systems such as food, water, energy and climate, thus enhancing livelihood resilience and diversification. For instance, urban agriculture, as a form of NbS, can increase food security and improve human being livelihood. Immense benefits as well as the innovative governance, institutional, business, and finance models and frameworks inherent to NbS implementation provide a wealth of opportunity for social transformation and increased social inclusiveness in cities. Given the range of interventions by NbS and the cross-sectoral co-benefits, new processes and designs for informal area upgrading are interrogated and implemented. Opportunities for NbS implementation should be explored and, where relevant, upgrading activities should make use of NbS. Indeed, investing in NbS will let African drylands meet urgent global challenges sustainably as well as benefits biodiversity and livelihoods.

In summary, the NbS in arid and semi-arid lands (ASALs) can be grouped into five core principles as listed below (Cohen-Shacham et al. 2016; Seddon et al. 2020; Thorslund et al. 2017; Raymond et al. 2017).

(1) Environmental restorative capacity: Ecological rehabilitation (ER), forest landscape rehabilitation (FLR), ecological engineering (EE);
(2) Issue-specific: Ecosystem-based adaptation (EbA), Ecosystem-based mitigation (EbM), Ecosystem-based disaster risk reduction (Eco-DRR), and Climate adaptation services (CAS);
(3) Infrastructure development: Natural infrastructure (NI), Green infrastructure (GI);
(4) Managerial functions: Ecosystem-based management (EbMgt), e.g., Integrated coastal zone management (ICZM); Integrated water resources management (IWRM); and
(5) Protection measures: Area-based conservation (AbC).

The five categories of NbS are summarized as conceptual representation in Fig. 9.14.

To comply with the NbS principles (Fig. 9.14), there is a need for effective involvement of different actors with civil society organizations and the private sector (Leone et al. 2021), integration of hybridized approaches of green, blue, and grey infrastructure (Depietri and McPhearson 2017), maintain dryland soil biodiversity by planting indigenous trees along roads and in households (Thorn et al. 2021), linking informal transport networks with green spaces, shifting perspective from "unplanned" to "unserviced", experimentation of "untried beginnings" (Cilliers et al. 2021), and generation of and use relevant data for evidence-based decision making (Frantzeskaki et al. 2019).

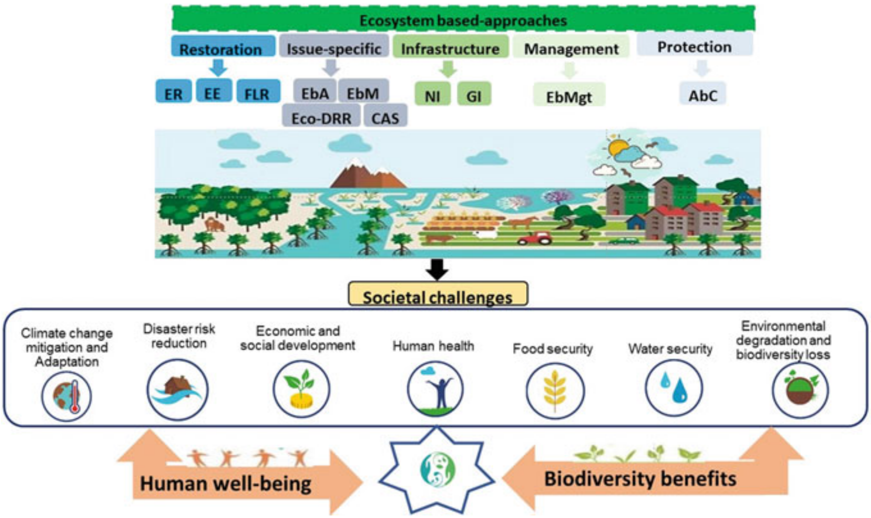

Fig. 9.14 Hypothetical representation (prototype) of NbS in drylands in Africa. Adapted from Cohen-Shacham et al. (2019)

9.4 Driving Forces of Dryland Change

9.4.1 Climate Change and Extreme Events

The World Meteorological Organization (WMO) declared the confrontation of a mix of changing precipitation patterns, increased temperatures, rising sea levels, and more frequent extreme weather and climate events (Blunden and Arndt 2020). Extreme weather events such as droughts, floods, and landslides, are likely to occur more frequently and/or with greater intensity in the twenty-first century according to climate measurements and models (Niang et al. 2014; Orimoloye et al. 2019). Extreme temperature occurrences have a severe impact on agriculture in Africa since many crops are already planted at the boundaries of their thermal tolerance and water stress resilience. Meanwhile, much of Africa's agricultural production takes place in semi-arid regions which are expected to get drier in the future (Scholes et al. 2015).

Reduced agricultural productivity as a result of heat and drought stress, as well as increased insect, disease, and flood damage will have significant consequences for regional, national, and household food security and livelihoods (Blunden and Arndt 2020). Under RCP 8.5, reductions in mean yield of 13, 11, and 8% are projected in West and Central Africa, Northern Africa, and East and Southern Africa. Wheat and rice are expected to be the worst-hit crops, while millet and sorghum are likely to be the least afflicted. Seasonal weather patterns are anticipated to be affected by global climate change. Concerns have been raised that converting Africa's dry tropical forests and savannahs to croplands for agricultural production may undermine the biomes' natural carbon reserves (IPCC 2019). According to a study based on 20,000 historical maize trials in Africa and daily weather data, the productivity of African maize declined by 1% for every 1 °C increase above 30. Under the same temperature situations, the yield was reduced by 1.7% in drought conditions (Lobell et al. 2011). To build the resilience of ecosystems and livelihoods in drylands, there is a need for a harmonized framework that integrates multiple hazards, including droughts, floods, and fires (Cervigni and Morris 2016). This will assist in determining the link between extreme climate occurrences and African populations' socioeconomic well-being.

9.4.2 Anthropogenic Activities

Agriculture

Agriculture in the drylands is dominated by small-scale and resource-poor farms, which suffers from limited investment in agricultural technologies and inputs, resulting in declined crop yields and livestock productivity. Dryland farming expansion is thus a leading stressor to biodiversity. Dryland economies and societies have always been driven by agriculture and related land use. African dryland operations face problems of failing in providing basic services due to rapidly rising population growth and economic development and are often unsuccessful in producing enough food. These issues are, therefore, compounded by socioeconomic and ecological factors of resource degradation (e.g., water, land, and biodiversity) (Twomlow et al. 2006). Over 94.5% of African food production is rainfed, with over 728×10^6 ha rainfed cultivable area. Maize, millet, and sorghum occupy the highest crop areas for all of Africa, but with significant diversity among regions. Even so, rainfed agriculture also has low productivity and yields. For example, maize yields are 1.8–2 tons ha^{-1} in Africa as compared to 5.11 tons ha^{-1} of the world average. The low productivity is due to improper farming techniques, including the impacts of land degradation, inadequate pest control, inefficient water usage, low fertilizer use, low mechanization, and poor support structure. The level of public expenditure on rainfed agriculture is insufficient to reinforce viable, productive, and sustainable rural lifestyles (Abrams 2018). The four intrinsic features of dryland agriculture that demonstrate its dynamism and potential are: (1) diversity, (2) people resiliency and adaptability, (3) sustainable intensification in a fragile ecosystem, and (4) complementary investments in infrastructure and policy reform (Bantilan et al. 2006). The need to increase the productivity of dryland agriculture is vital to ensure world food security.

Nitrogen Deposition

In drylands, the atmospheric concentration of greenhouse gases (GHGs) has abruptly augmented in the last decades, resulting in ongoing climate change (Pachauri et al. 2014). Therefore, assessing nitrogen (N) deposition to drylands is intricated by the manifold forms and paths of N loading from the atmosphere (Sickman et al. 2019). Many studies on soil N balance in Africa provided evidence of widespread soil N depletion through harvested crops, plant residues transported out of the fields,

Table 9.4 Nitrogen flows at the farm level in Africa's dryland smallholder farming system. Adapted from Dlamini et al. (2014)

Flows	Nutrients
Inputs	Mineral fertilizers Organic inputs including • Animal/farmyard manures • Applied composts • Crop residues application Biological N fixation • Intercropping • Inoculant application Atmospheric N Biomass transfer
Outputs	Harvested crops Crop residues removal Runoff and erosion Leaching below the root zone Gaseous losses • Volatilization • Denitrification

overgrazing and/or leaching, erosion, and volatilization, which altogether surpass the amount of nutrient inputs through fertilization, atmospheric deposition, biological fixation, and organic inputs (Manlay et al. 2004). Removal of crop products and residues, leaching, gaseous losses, runoff, and soil degradation are all examples of N output processes. Figure 9.15 represents the Nitrogen cycle or flows while Table 9.4 represents the summary of inputs and outputs.

GHG fluxes are projected to be low in dryland ecosystems, such as those in the Mediterranean Basin, due to water and nutrient limitations, particularly N (Dalal and Allen 2008). 80% of the agricultural system in SSA is composed of smallholder farms (farm size < 10 ha) with low N application and organic and/or synthetic fertilizer use. This type of agricultural crop production at the national level has low inputs, with mean annual synthetic N fertilizer use in SSA ranging from 7 kg N ha^{-1} to 13 kg N ha^{-1} in West Africa and East Africa respectively (van Bussel et al. 2015). Farmers in some countries, such as Burkina Faso, acquire government or assistance group help for using mineral N fertilizers to increase the crop productivity. N_2O emissions from the Africa's agriculture sector are estimated to account for about 6% of all global anthropogenic N_2O emissions (Pelster et al. 2017; Brümmer et al. 2008). Low N inputs cause soil N reserves to deplete, a process known as soil "N mining", which is one of the main causes of soil fertility loss and low crop yields (Vitousek et al. 2009). Land degradation/desertification also leads to adverse loss of soil nitrogen stocks (Dlamini et al. 2014). N_2O emissions from agriculture in SSA will possibly double the contemporary anthropogenic N_2O emissions if current yield gaps are addressed (Leitner et al. 2020). Further, it has been demonstrated in SSA that increasing fertilizer application rates beyond a particular threshold (between 100 and 150 kg N ha^{-1}) causes a non-linear rise in direct N_2O emissions (i.e., N_2O that is discharged on-site from soils to which N is added) (Shcherbak et al. 2014).

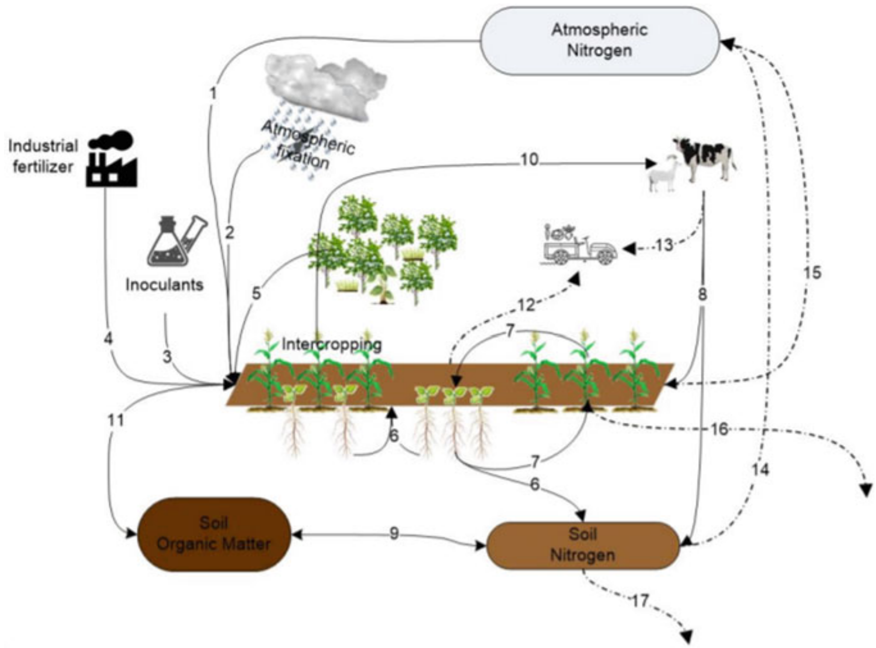

Fig. 9.15 Nitrogen and flows in smallholder farming system in drylands. Arrows represent flows, with solid lines representing N additions and exchanges, and dotted lines N losses. (Where 1 is biological nitrogen fixation, 2 atmospheric fixations, 3 microbial inoculations, 4 inorganic fertilizer application, 5 biomass transfer, 6 nutrient recovery, 7 crop rotation, 8 animal manure, 9 mineralization/immobilization, 10 animal feeds, 11 crop residues incorporated into the soil, 12 crops produce (goods), 13 livestock products, 14 denitrification, 15 volatilization, 16 runoff, and 17 leaching). Adapted from Kiboi et al. (2019)

Consequently, N is one of the foremost factors limiting agricultural thruput in African dryland agroecosystems (Rütting et al. 2018). Thus, this has had a great impact on the semi-arid cropping systems practiced in the continent. For instance, soil type together with the crops grown in the Sahel region, e.g., millet in Northern Burkina Faso largely contributes to N loss from the fields (Krogh 1997). To enhance or maintain the quality of the environment and conserve natural resources, alternative low-external-input approaches that involve the utilization of organic inputs have been developed for the farmers (De Jager et al. 2001) including the use of livestock manure for nutrient cycling and transformation of present agricultural land to other N-recycling efficient farming opinions in semi-arid conditions. However, fertilizer application results in higher soil fertility, together with the rise in N_2O emissions. Increased fertilization should be considered alongside GHG emissions that can be evaded (e.g., by deterring soil degradation and SOM mineralization), counterbalanced via C sequestration due to improved soil management (e.g., by building up additional SOM owing to enhanced residue input), and mitigated with emissions that would otherwise eventuate elsewhere due to cropland expansion (e.g., via deforestation, and grassland conversion).

Livestock Grazing and Fencing

Livestock is the main (and often only) land-use option in Africa's drylands. This sector is the keystone of the national economy in many of the countries of East and West Africa, the majority of which have a vast area of drylands (FAO 2018). Pastoralism is performed in Africa's major areas, covering 43% of the continent's territory. It covers about 36 countries (in 53 countries), elongating from the Sahelian West to the rangelands of Eastern Africa and the Horn and the nomadic populations of southern Africa (FAO 2018). About 25×10^6 pastoralists and 240×10^6 agro-pastoralists rely on livestock as their main source of income. In the SSA, 35% is permanent pasture (Kiage 2013). Further, in broad terms, pastoralism prevails in eastern Africa's drylands, whereas limited crop-livestock integration and agro-pastoralism prevail in western Africa's drylands, which can be traced in part to bi-modal against unimodal weather patterns (Milne et al. 2016).

Additionally, livestock has both positive and negative effects on the dryland resource base. In global drylands, livestock production sustains millions of livelihoods (Zhang et al. 2021), and pasturage is expected to increase in the next decades (Chillo et al. 2017). However, desertification occurs largely in drylands as a result of overgrazing (i.e., biodiversity loss and degradation of ecosystem functions). Grazing has been reported to create lower litter quality (i.e., low N and high secondary compound content) in drylands, which, when combined with a reduction in litter quantity and soil moisture, has a detrimental impact on the decomposition rate (Campanella and Bisigato 2010). Plant diversity changes also have an impact on animal communities by altering habitat structure and food security (Chillo and Ojeda 2014).

Nonetheless, in parts of SSA (especially Ethiopia), fencing or protecting an area for livestock fodder has become a useful strategy for supplying the animals with feed during times of stress (Catley et al. 2013). Pastoralists in drylands use livestock mobility as the primary strategy to deal with and exploit natural resource unpredictability (e.g., the case of Botswana McGahey (2011)). Pastoral mobility, diversification of livestock species, and maximization of herd numbers are some pastoralist insurance strategies that communities use to manage extreme uncertainty in their environment. In addition, while considering efforts to improve carbon management, pastoral organizations must be recognized and developed upon. Farmers and livestock keepers use a variety of management strategies across the varied land-use systems to obtain lucrative benefits (i.e., food and nutrition security, livelihoods, and revenue, etc.) as well as to improve the "condition/health" of the grazing areas. The main priorities of the management practices are to (a) decrease and combat land degradation, (b) restore or rehabilitate the land, and (c) increase land productivity for cattle production. Grazing management or pasture improvement (e.g., increased productivity, nutrient management, forest management, and species legumes), livestock management (e.g., improved feeding practices, precise agents, and nutritional additives), and restoration of degraded rangelands are examples of these management practices (e.g., erosion control, organic amendments, nutrient amendments).

There is a myriad of reasons why people in Africa's drylands face food insecurity and are unable to satisfy their nutritional needs and targets. Although there is no single reason why food shortages, insecurity, and the prevalence of malnutrition continue to plague Sub-Saharan Africa, failed internal economic policy tools and international policy prescriptions are identified as the culprits or causative factors (Dodo 2020). The main factors that have aggravated the problem of food production, supply, and accessibility are drought and conflict. Within an already tough setting of fragile ecosystems, high rates of population increase and ratio of poverty have also played a role. Because about 80% of the population in the region is rural and relies almost entirely on agriculture for consumption and income, solutions to the challenges of poverty and food insecurity must be predominantly found in the agricultural sector. The link between poverty and food insecurity is critical. Food production is important since agriculture is the primary source of income for the majority of the poor, and agriculture employs around 76% of the IGAD population. However, the level of food insecurity is lowered only when poverty is relieved or reduced. As a result, the long-term approach to food insecurity goes beyond increasing food production and involves the need to strengthen rural livelihoods in generally. Social safety nets of many kinds are also part of the solution to extreme poverty and food insecurity, not only in exceptional conditions like drought but also over the long periods needed to arrive at inclusive societies and as lasting solutions.

Chronic food insecurity is the most common and devastating consequence of these concerns in Africa's drylands. According to the African Union Commission's (AUC) Food Security Report, 27% of Africa's overall population is undernourished, nearly half of Africa's children are stunted, and acute malnutrition (>10%) is reported in more than 15 nations. Africa is currently attempting to cover its food insecurity with imports worth approximately US$ 20 billion per year, in addition to requesting food aid (AUC-NEPAD 2006). The majority of the victims of food insecurity in the region are the poor inhabiting the drylands who depend heavily upon natural resources for their livelihoods, either by growing crops or managing livestock.

Climate change impacts and continues to impair the subsistence of communities in Africa's drylands, which has now become a critical concern for the long-term development of the region (Epule et al. 2017). This challenge consists of the potential consequences of agroforestry systems on ecological services, agricultural productivity, and livelihoods. Agroforestry systems are traditional land-use methods that incorporate trees into agricultural grounds. These systems are widespread in Africa's drylands and have been practiced for generations. Unfortunately, Africa's dryland is highly susceptible to the effects of climate change (Epule et al. 2014) because of its reliance on rainfed agriculture. These areas' rural lifestyles are heavily reliant on agriculture and non-timber forest products, both of which are threatened by climatic changes. As a result, the regions are no longer able to provide good yields in ecological systems to sustain rural people's livelihoods.

9.4.3 Wildfires

Wildfires are an extreme threat to the dryland environments (e.g., grasslands, savannas, or dry forests) and are increasing due to increasing ignitions caused by humans, the spread of fire-prone invasive grasses and shrubs, and warming, drying climate. The dramatic increase in wildfire prevalence in recent decades poses serious threats to human safety, infrastructure, agricultural production, cultural resources, native ecosystems, watershed functioning, and others. Wildfires are especially prevalent in Africa, with up to 9% of the continent burnt on an annual basis (Andela et al. 2013), contributing to 70% of the global burned area (Andela and van der Werf 2014). More extensive and later dry season fires lead to wet season rainfall deficits of up to 30 mm (Saha et al. 2016). Recently, the MODIS tool on NASA's Aqua satellite detected multiple dozens of fires burning in southwestern Africa. Similarly, using the albedo model, the study of Saha et al. (2019) identified the strongest brightening in the Kalahari region as well as more intense and long-lived initial darkening in the Sahel region. In some biomes, the frequency of wildfires is widespread and alarming, such as in the forests and savannahs of West and East African countries. As fire frequency depends on fuel production, it is influenced in arid and semiarid regions by the total rainfall (Fig. 9.16).

Forest fires regimes are also responsible for woodland degradation in dry regions (Nichols et al. 2017). Fires, sometimes set to clear the land for agriculture, leave the soil susceptible to erosion and exposed to sunlight and other elements, which may change the makeup of the soil and prevent the tree species from regenerating (Fig. 9.17). Fires can also place neighboring stands at risk as grazing animals move into new areas to find forage, intensifying the pressure on resources and leading to overgrazing. Fires are a primary cause of desertification in the SSA regions, where the degradation of drylands is especially pronounced (Wei et al. 2020).

The occurrence and impacts of wildfire can be reduced through prevention, preparedness, and pre-fire management. The post-fire response such as erosion control and replanting in burned areas also helps to reduce the immediate impacts of wildfire and the establishment of nonnative grasses, which can reduce the risk of future fires. Given limited resources for land management and the ability of wildfires to cross property boundaries, building collaborative relationships among land managers, landowners, scientists, fire responders, and the public is key to addressing wildfires in African drylands.

9.4.4 Resource Conflicts in African Arid and Semi-arid Areas

The present state of natural resource degradation in the African drylands is explained in terms of factors related to ecological and demographic pressures, land-use conflicts, and inefficient land administration policies (Reda 2015). The armed conflicts strongly affect the agricultural activities (Demissie et al. 2022). Today, many

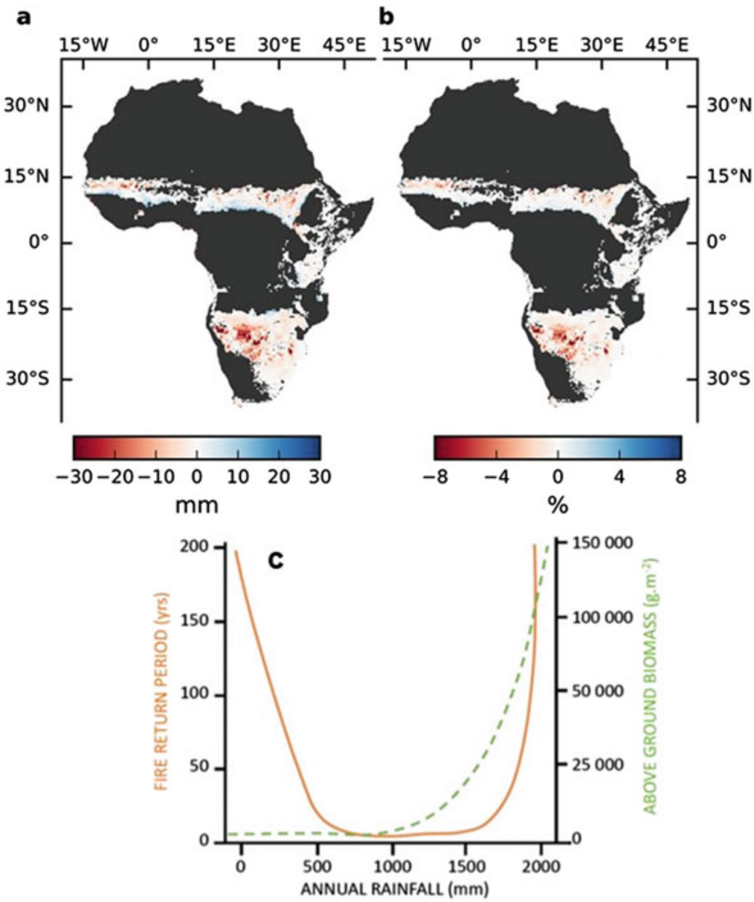

Fig. 9.16 Fire-induced rainfall suppression in drylands. **a** Rainfall lost as the difference between modeled wet season rainfall and wet season rainfall with no fire. **b** Rainfall modifications (expressed as a percentage of mean annual precipitation). Adapted from Saha et al. (2016). **c** The relationship between rainfall, fire frequency (continuous line), fuel accumulation (discontinuous line) in the southern Africa region. Adapted from Hély et al. (2019)

protected areas in SSA are located in areas of conflict (IUCN 2018). The potential conflictive areas in African drylands include the Senegal valley, the Niger Delta, the Kenyan highlands and wetlands, Tanzanian game-reserves and protected parks, and conflicts between Botswana and Namibia over the use of water resources as well as national politics and land tenure conflicts, the (Le Meur et al. 2006). Moreover, in the HA, i.e., Ethiopia, Eritrea, Somalia, and Djibouti, the first three countries are among the 20 countries in the world, where the most threatened herbivore species are found (Ripple et al. 2015; Sterzel et al. 2014; Pettersson and Öberg 2020), the case of Tigray (Balehegn et al. 2019). The HA has the highest conflict density in global drylands (8 out of 42 conflicts) (Fig. 9.18).

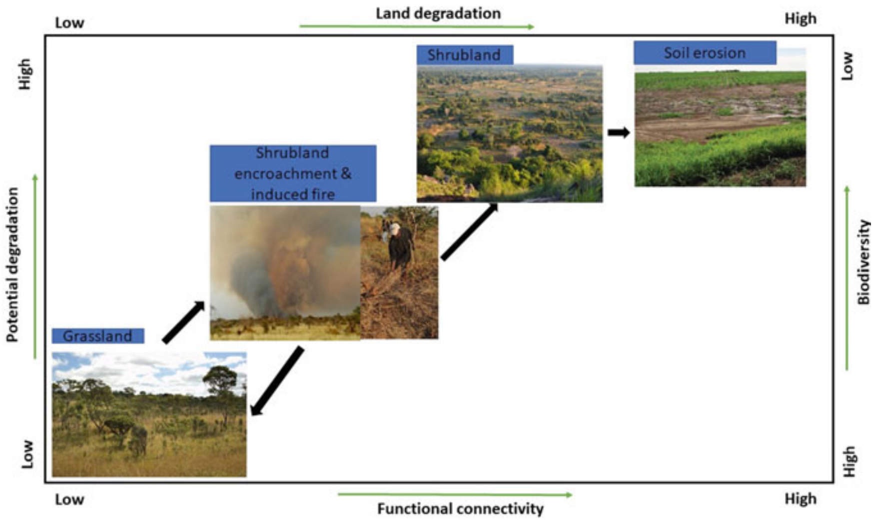

Fig. 9.17 Conceptual framework illustrating the stages of land degradation in typical Miombo woodlands, Southern Africa, with changes in biodiversity, functional connectivity, and soil erosion. Modified from Ravi et al. (2010)

Many researchers found that more than 70% of Africa's protected areas experienced conflicts during the last two decades. Several large nations experienced an average of 20 or more years of conflict per protected area, including Chad, Namibia, and Sudan (Daskin and Pringle 2018; Wigley et al. 2010). However, the large-mammal populations, including many threatened species have declined sharply (Ripple et al. 2015). Nowadays, almost all the countries in SSA experienced net encroachment, with only Congo, Kenya, Madagascar, Niger, and Somalia undergoing a net decline in woody cover. The highest rates of encroachment occurred in areas with moderate initial woody cover (i.e., 30–60%) in 1986. Areas with more than 75% initial cover experienced the highest rates of loss, probably due to human-induced clearing (Venter et al. 2018). Grazing herbivores, which dominate most African rangelands, reduce grass competition with woody plants and reduce fuel loads for fires, thereby releasing woody plants from the fire trap (Hempson et al. 2015; Roques et al. 2001). As result, shrub invasion is often associated with "overgrazing", and high browsing pressure can, in contrast, prevent the establishment of woody seedlings and retard the growth of shrubs, prolonging their exposure to fire and suppressing their recruitment into the mature stage (Roques et al. 2001). Figure 9.19 illustrates the major drivers of encroachments in protected areas (PAs) and their consequences on the dry environments.

The reasons for pAs encroachment in African drylands are still something of a puzzle. Multiple drivers likely interact to cause pAs encroachment. The uncertainty lies mainly in quantifying the importance of these drivers and understanding the extent to which they interact with one another. Factors such as herbivory, fire, and

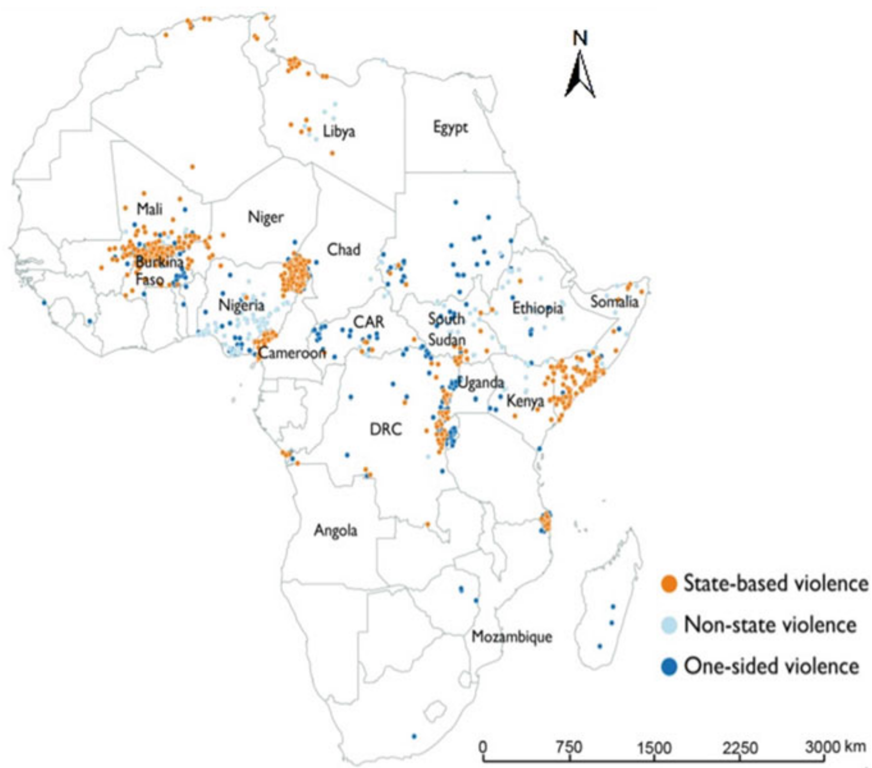

Fig. 9.18 Spatial distribution of armed conflicts and African drylands. Data from Uppsala Conflict Data Program (UCDP) georeferenced event dataset (GED). https://ucdp.uu.se/. Accessed 10 December 2021

soil properties are likely to alter woody cover and rates of encroachment in both wet and dry savannas at all levels (Devine et al. 2017). There is a need to enforce the best practices (BPs) approach and integrated conservation and development projects (ICDPs) to encourage conservation and development in rural communities adjacent to protected areas (Mutanga et al. 2015). The involvement of all stakeholders is very crucial. There is also a need for vastly elevated funding for PA management and research from both African and international governments and institutions.

9.4.5 Interactions Among Different Drivers

The DSES concept explicitly implies that humans and nature are inextricably linked. The effects of anthropogenic activities and climate change on ecosystems change their structure and function, thereby facilitating the provision of goods and services that contribute to human well-being (Fig. 9.20). For instance, livestock production in

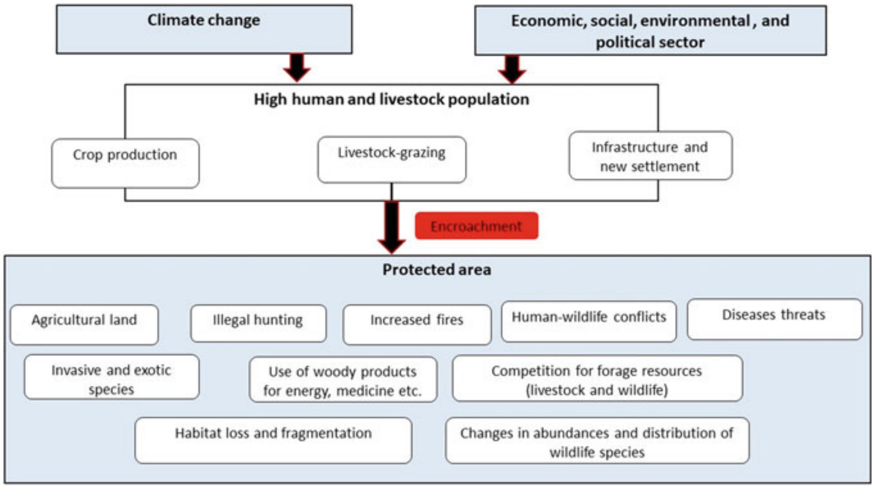

Fig. 9.19 Conceptual framework defining the major factors driving encroachments into a PA and their impacts in arid and semi-arid areas of Southeastern Africa

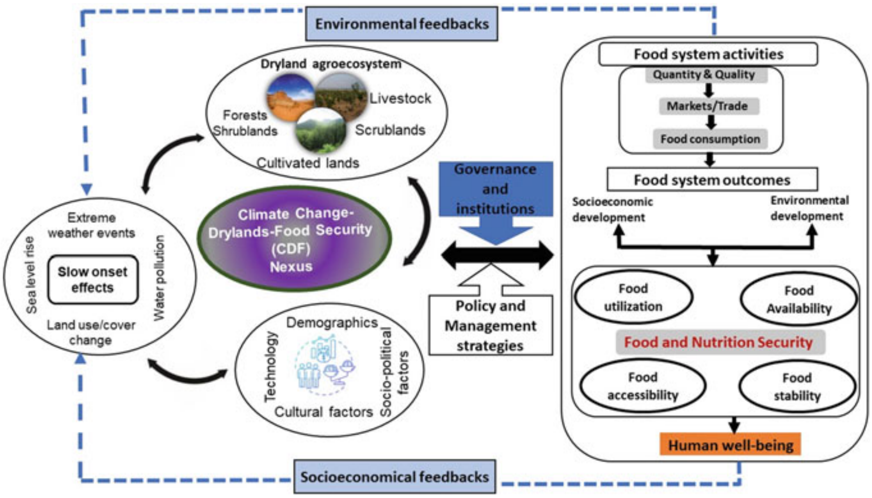

Fig. 9.20 Interconnection between food system, socioeconomic development, and dryland agroecosystem. Adapted from Hirwa et al. (2022)

drylands for semi-arid and arid societies, climate change mitigation by carbon sequestration, and cultural services such as distinctiveness and place for touristic activities (i.e., parks) all contribute to the well-being of dryland communities. However, drylands are directly affected by climate variability, human activities such as urbanization, and agricultural activities. The proximate drivers are influenced by distal socioeconomic dynamics, such as human population growth, technology, socio-economic development, governance, institutional arrangements, and others. These indirect drivers do not always have direct effects on the dryland environment, but they do have an impact on the natural environment through moderating the effects of proximate causes.

Additionally, these interconnected ties between people and DSES can contribute to the community's resilience in different ways. Nonlinear dynamics and social and ecological feedbacks can promote negative system states. The proximate human actions such as overexploitation that directly influence ecosystems are shaped by underlying causes, or distal social, economic, cultural, and institutional forces. DSESs' assessments require understanding the factors that can help or hinder resilience. Likewise, the study of Cinner and Barnes (2019) indicated the adaptive capability were referred to six broad categories of social characteristics that contribute to dryland social-ecological change resilience: (1) the resources available to persons, (2) the ability to switch strategies, (3) the ability to plan and act collectively, (4) adaptation to changes and recognizing them, (5) the socio-cognitive structures that allow or limit social actions, and (6) the agency to decide whether or not to modify.

9.4.6 Research and Technology Gaps in African Arid Ecology

Long-term data series are important tools to answer ecological and evolutionary questions that need broad spatial and temporal monitoring. The lack of temporal information (i.e., long-term data series) leads to serious misjudgments that can interfere not only with attempts to understand and predict changes but also with efforts to manage the environments (Barbosa et al. 2020). Predictive models would be useful to understand and implement restoration programs that include the interactive effect of environmental variables and aquatic communities (Tessarolo et al. 2017). Several models have been built to prevent or reduce the adverse environmental impacts in arid and semi-arid zones. For instance, eutrophication (Mooij et al. 2010), flood forecasting and control (Refsgaard et al. 1988), drought prediction (Mishra and Singh 2011), crop growth modelling and crop yield forecasting (de Wit and van Diepen 2008; Khaki and Wang 2019; Di Paola et al. 2016), and among others have been developed and implemented.

The number of Earth Observation Networks (EONs) and Ecosystems Research Networks (ERNs) in Africa is relatively low compared to other regions worldwide. Therefore, this challenge results in huge uncertainties and subsequently affects decision-making at the international watershed levels, rendering the design of efficient adaptation measures much more difficult. These uncertainties are due to limited scientific understandings of the climate drivers and their interactions (e.g., West African (Klein et al. 2017), East Africa (Rowell and Chadwick 2018; Bornemann et al. 2019), and Southern Africa (Davis and Vincent 2017), resulting from a lack of high quality, long-term observation data, and specific data mining capabilities. For instance, Climate Risk and Early Warning Systems (CREWS) showed that the West African countries were most vulnerable to weather extremes because their national hydrological and meteorological services or agencies had limited early warning capabilities (i.e., the low infrastructure, observation systems, and human capacities), weak or non-existent dissemination systems, and a lack of effective emergency planning in case of alerts and warning information (Salack et al. 2015).

Therefore, there is an urgent need to enhance near-surface measurements and observation infrastructure in African drylands in order to develop coherent procedures of climate services delivery to national civil protection, humanitarian support agencies, and vulnerable communities. Droughts, flooding, air pollution, and dry spells, among other extreme events, can be detected using the network of near-surface observatories (Giannini et al. 2013; Knippertz et al. 2015; Salack et al. 2019), and to underpin climate services for mitigation, adaptation measures, and risks assessments (Ouedraogo et al. 2018; Jones et al. 2015). The strong commitment of African governments to ensure the sustainability and continuation of the transnational observation networks will empower African and world scientists as well as national meteorological and hydrological agencies to conduct research and deliver ecosystem services at a high level of accuracy and achieve Sustainable Development Goals (SDGs).

9.5 Summary and Perspectives

African DSES is considered as hotspots of vulnerability to environmental variability. African drylands have notably experienced change in land-use shifts and management in different regions of Africa. Therefore, understanding how Africa's drylands adapt to climate change and anthropogenic influence and maintaining the functional integrity of DSES is fundamental for sustainable development in the context of global environmental change.

This chapter provides a synopsis of African drylands as a DSES. The major features, trends, driving forces, potential future perspectives of drylands are reviewed, thereby informing policymakers, decision-makers, and stakeholders to harmonize strategies for DSES management in a sustainable way. The DSES are complex adaptive systems composed of connections between different people and dryland ecosystem factors. Biophysical and socioeconomic factors contribute to the emergence of DSES dynamics, which combine nonlinear and linear patterns with gradual yet abrupt developments. Comparing identical dryland DSES and diverse responses to global change, a better understanding of the context-specific DSES traits will be easier to be obtained. More study is needed to reduce the uncertainty in projecting system change trajectory and to investigate how synergies and trade-offs in drylands DSES are linked to spatial and temporal scales. Finally, this chapter highlighted the immediate future investment approaches and perspectives for climate-adapted development in Africa drylands:

- Empowerment of the local i.e., investing in infrastructure (e.g., communications and transport); investing in the development of governance systems that empower locally-led adaptation; prioritizing vulnerable stakeholders in order to provide the basis to address distributional outcomes and equity and improve community-level resilience.
- Supporting and exchange of local practices i.e., promoting investment in the demonstration of locally-led management practices that enable land resources restoration and sustainable production using NbS, carbon and biodiversity credit markets; implementation of participatory/joint research approaches to engage academic entities in supporting local knowledge, innovation, and technologies along with associated policies, so as to speed up the local adaptive learning.
- Involvement of the public in developing solutions: Building social and human capacity, fostering environmental mainstreaming education, and creating public awareness campaigns and trainings to educate citizens in dryland regions of Africa which provide the basis for adaptation to on-going future. Instead, providing the incentives and support for local communities to drive transformation and create job opportunities for youth and women.
- Providing enabling and integrated environment for strategic technical and operational partnerships and policy coordination and knowledge management. Undoubtedly, investing in major climate-adapted initiatives that run across multi-disciplinary sectors, people, and countries. Additionally, coordinate investor partnerships (i.e., dissolving climate finance mechanism) to drive free trade investments at large scale and over multiple funding cycles that accumulate to build resilience and reduce regional conflict.

In the final analysis, there is a need to promote sustainable agricultural best practices (e.g., NbS and Ecosystem-based Adaptation programs) and innovations as a tool to enhance community resilience and cope with climate change impacts on water-food security, use modern observational data and develop idealistic models to better understand the climate-drylands-food security nexus approaches, and strengthen dryland research and management effectiveness through emerging and affordable technologies. The above-mentioned recommendations should be seriously considered in future research and policy-making on DSES not only in Africa but also globally.

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Chapter 10
Dryland Social-Ecological Systems in Americas

Yangjian Zhang, Ke Huang, Jianshuang Zhang, Ran Zhao, Dennis S. Ojima, Jie Gao, and Yixuan Zhu

Abstract American drylands account for circa 20% of the global drylands and form a critical part of the global ecosystems. This study comprehensively assessed the ecology and socio-economic status of American drylands by analyzing original and published data. The research findings reveal that North and South American drylands have more differences than commonness. In terms of commonness, both North and South American drylands have higher productivity and soil fertility than other drylands of the globe. Under this high ecosystem productivity context, North American drylands are the high agricultural productivity regions and South America is the largest beef exporter in the world. There are several aspects of differences between North and South American drylands. North American drylands possess an ecosystem productivity twice that of South American drylands. Precipitation has significantly decreased in North America drylands, while South American drylands have become wetting over the past three decades. Population in both North and South American drylands have increased. Vegetation coverage trends exhibit a weak rising trend in South America, while North America drylands have become significantly greener, mainly due to croplands irrigation. The driving forces on land use change and ecosystem productivity in North American drylands comprise a variety of factors, while those on South American drylands are relatively simpler, mostly caused by one driving agent. In dealing with the dual pressures of climate change and socio-economic developments, countries in both North and South America have implemented a series of drylands ecosystem protection measures, such as setting national park and conservation agriculture. These efficient and successful experiences can be examples for other dryland ecosystem protection around the world.

Keywords Drylands · North America · South America · Climate change · Socio-economic developments

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10.1 Introduction

North and South America are called America in combination, and they are two separate continents in the Western Hemisphere. The two continents are under the umbrella of totally different climates. For North America, climates transit from the prevailed subarctic climate in the North to the tropical climate in the south, sequentially harboring arctic, subarctic and tundra, desert and semiarid, savanna, and tropical rain forest ecosystems along the climate gradient. Among them, the arid and semi-arid climates are prevailed in the interior regions, where rain-bearing westerly winds are obstructed by Rocky Mountains. The wide variety of climates breeds diverse vegetation, including conifer taiga forests of Canada, Pinus ponderosa and Pinus edulis dominated ecosystem in Colorado plateau and Canyon-lands regions, and grasslands in great plains. Shrubs, like Artemisia tridentate and Cercocarpus montanus, are extensively grown in open spaces between trees.

For South America, the climate transits gradually from tropical in the north to marine in the South. Fed by adequate rainfall, the Amazon River basin accommodates the most extensive tropical rainforest in the world. On the other hand, moistures carried by the westerly winds mostly precipitate on the west side of the Andes and leaves its eastern part extremely dry. The cold Peru Current also causes northern Chile dry. Typical dryland forests are mainly located in the Gran Chaco, primarily composed of Maranhão Babacu and Caatinga.

10.2 Major Characteristics of Drylands in the Region

10.2.1 Dryland Distribution

Drylands occupy approximately 30% of American continents and American drylands account for circa 20% of the global drylands. They stretch from central Canada to the central and western parts of the United States, the entire northern half of Mexico, parts of the Caribbean, the Pacific coast and southern parts of South America (Fig. 10.1). According to the definition of the United Nations Convention to Combat Desertification (UNCCD), the aridity index (AI), calculated by P/PET (P annual precipitation, PET annual potential evapotranspiration), define drylands as regions with AI < 0.65. The AI also classifies drylands into four different types, e.g., dry subhumid ($0.5 \le AI < 0.65$), semi-arid ($0.2 \le AI < 0.5$), arid ($0.05 \le AI < 0.2$) and hyper-arid ($AI < 0.05$) regions (Middleton and Thomas 1997).

A high proportion of Americas' drylands belong to temperate drylands (97%), except the small proportion of tropical dryland distributed in Latin Americas. More than half of the North America drylands (54%) can be assigned to the semi-arid type. The second most prevalent type is dry subhumid (22%), which is mostly distributed along the edges of the drylands. Approximately a quarter of the North American drylands are distributed in the arid zone, primarily in the interior western part of the

Fig. 10.1 The drylands in North and South America determined by the average aridity index from 1981 to 2019

United States, the Baja Peninsula, and coast of the Gulf of California in Mexico, along with one region in central Mexico and some regions straddling the border between Mexico and the United States. The hyper-arid zone covers only less than two percent of North America's drylands, mainly located at the northern tip of the Gulf of California. Chihuahuan desert, as the largest desert in North America, stretches all the way from the southwestern United States deep into the Central Mexican Highlands.

The drylands of South America are approximately 552 million hectares, covering circa 31% of the region's total land area. They are primarily distributed in the semi-arid zone (46%) and dry subhumid zone (41%), with only eight and five percent in the arid and hyper-arid zones, respectively (Table 10.1). South American drylands are mostly distributed in two main topographical areas, which are the high mountains of the Andes in the west South America and the Brazilian and Guiana Highlands in the east South America.

The United States and Argentina are home to the largest area of drylands over North America and South America, respectively. In the United States, Argentina, Canada, Mexico, Brazil, Bolivia, Colombia, Ecuador and Praguay, the drylands are mainly classified as the semi-arid. The drylands in island countries mostly belong to the sub-humid type (Fig. 10.2).

The primary factor limiting vegetation growth in drylands is water shortage. Low soil moisture supply and high atmospheric water demand are considered as the two main drivers causing dryness stress on vegetation. Temperature and humidity are the two basic factors defining vapor pressure deficit (VPD) (Fig. 10.3). As a

Table 10.1 Area of each type of drylands in Americas

Sub aridity zones	North America		South America	
	Area (km^2)	Percentage (%)	Area (km^2)	Percentage (%)
Dry sub-humid	1,431	22.45	2,527	45.78
Semi-arid	3,473	54.50	2,267	41.07
Arid	1,355	21.26	444	8.04
Hyper-arid	114	1.79	282	5.11
Total	5,982		5,520	

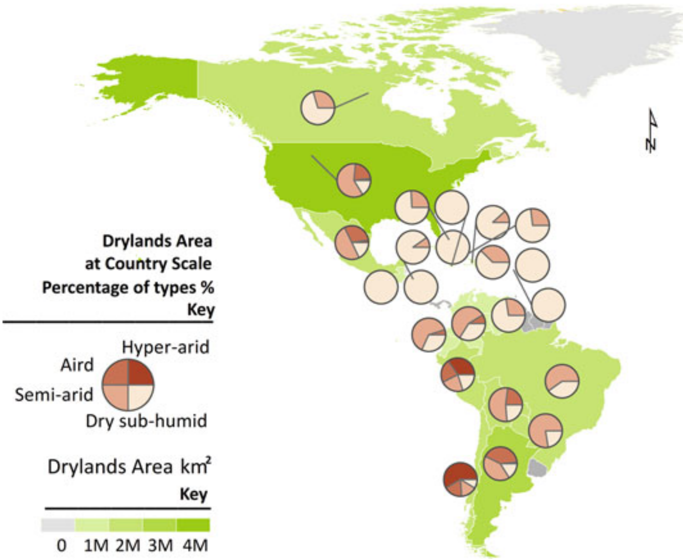

Fig. 10.2 The area and percentage of each type of drylands in North and South America

proxy for plant water stress, VPD is what actually affects plant growth via moderating the transpiration process, and reflects the effect of temperature and precipitation on the relative humidity and transpiration demand (Seager et al. 2015). The warming-driven increases in vapour pressure deficit hasten evaporative water loss and deplete surface moisture, in turn amplifying atmospheric drying through the land–atmosphere feedbacks (Lian et al. 2021).

The distribution of drylands and arid climate are the joint results of atmospheric circulation and large-scale topography interacting with synoptic-scale and mesoscale weather systems. The drylands over southwestern North America are strongly influenced by the subtropical highs together with the descending branch of the Hadley cells (Scheff and Frierson 2012). Moreover, some dryland regions in South America and the western United States are heavily impacted by topography because high mountains produce the foehn effect and block the passage of rain-bearing air

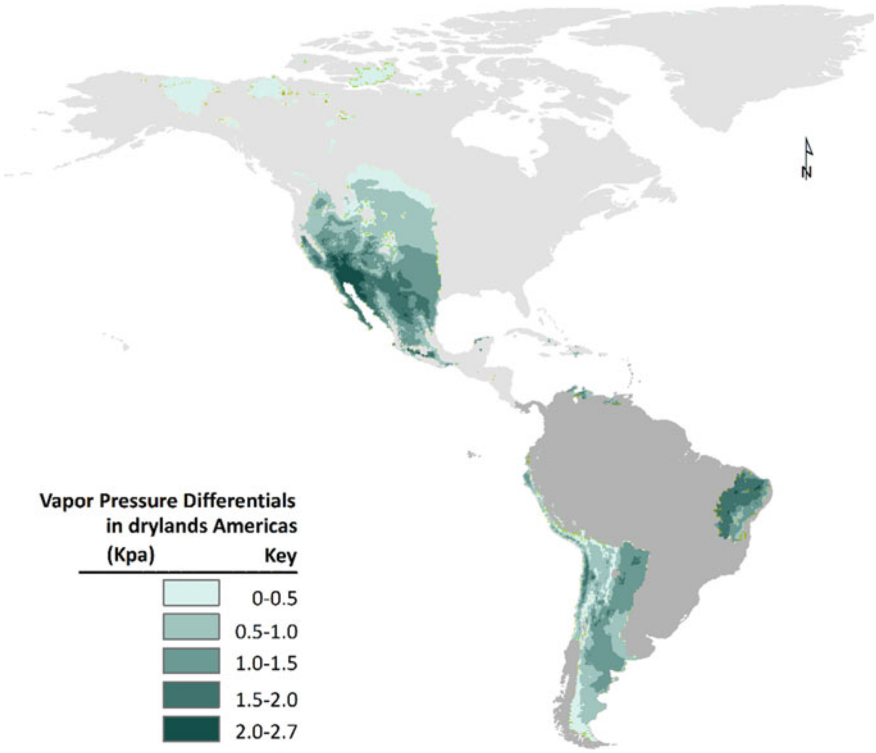

Fig. 10.3 The vapor pressure deficit (VPD) of American drylands

(Huang et al. 2017a). Over the past half-century, the semi-arid regions of the American continents have expanded significantly. The newly formed semi-arid regions were mainly developed from arid regions in southwestern North America that had become wetter caused by enhanced westerlies in recent years (Huang et al. 2016a; Li et al. 2019).

It is predicted that drylands would expand under future climate scenario (Morales et al. 2011; Koutroulis 2019). In North and Central America, the arid regions will occupy most of New Mexico, western Texas, and most of northern Mexico. By the end of this century, the semi-arid regions will expand eastward by 2–3° of longitude in the Great Plains. Only a few dry regions in southern South America may get wetter. Potential dryland expansion means lower ecosystem carbon sequestration and a greater risk of desertification (Huang et al. 2017b), severely affecting usable land availability and threatening food security.

10.2.2 Dryland Ecology and Biogeographical Characters

Dryland Climate and Soils

The dry condition of American drylands is due to the co-influences of the Pacific currents and Andes Mountain barrier. For the ocean current, warming phases are known as El Niño and cooling phases are known as La Niña. The prevailing climates in the drylands of North America are mainly formed due to the planetary-scale atmospheric circulation in the subtropical and mid-latitudes. The westerlies and the mid-latitude cyclones produce the dryer climate in the west and southwest of North America. Changes in atmospheric circulation patterns, in combination with the oceanic temperature rhythms regulated by El Niño and La Niña events, result in annual climate variations comprising severe drought years and wetter-than-average years throughout the region. The southwestern region of the drylands is also affected by monsoon events, which are localized climate patterns characterized by seasonal fluctuations in temperature and precipitation.

In South America, Andean areas feature dramatic temperature fluctuation and decreasing rainfall from east to west. The high Andes accommodate cold areas in central Peru, Bolivia and Chile with temperatures ranging from −2 to 12 °C and precipitation ranging from 610 to 1,420 mm. Temperatures in the tropical wet-dry areas of the Brazilian highlands and Ecuador can reach 18–35 °C. In eastern Brazil, the area around Parnaíba and the São Francisco River is characterized as an interior warm zone, receiving only 100 mm annual rainfall. In southern Chile, the annual rainfall can reach 2,500 mm. The warm and cold deserts in Patagonia and northwest Argentina are characterized by an arid climate. In Patagonia, the highest temperature is about 20 °C. Temperatures in the Atacama Desert can reach 18 °C, with almost no rainfall in the whole year.

Soil provides foundation to support the ecosystem functions and services, which includes nutrient cycling, carbon storage, water security, food, and fiber production. Tracing down to the basic processes underpinning other ecosystem function and services is the nutrient cycling. Unlike other global drylands, such as in Africa and Australia, the drylands in Americas have generally less nutrient constrain according to FAO Harmonized world soil (Fig. 10.4). The extensively distributed Cyanobacteria in arid and semiarid regions of North America play a significant role in nitrogen fixation (Eldridge et al. 2020; Maestre et al. 2013). Higher nutrient availability, which means less nitrogen limitation and higher soil organic matters content, can improve soil carbon storage capacity and vegetation carbon sequestration capacity.

Biodiversity in Drylands

Species diversity pattern highly hinges on their origins and evolution. In South America, dryland plants were developed in the Paleocene (66–56 million yr ago (Ma)) while in North America, they were developed in the beginning of the Late Cenozoic (33.9 Ma) (Thompson and Anderson 2000). The long developing history of dryland plants across the continents, and their roles as the origin of many unique plant lineages make them an important host to a diverse flora. There are some typical

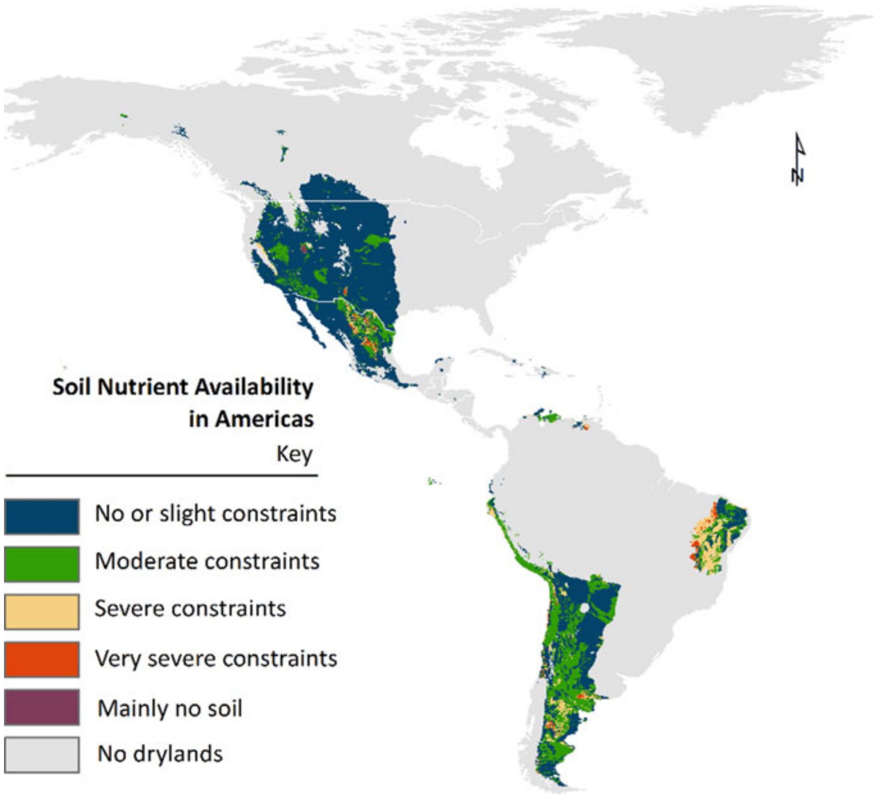

Fig. 10.4 Soil nutrition availability of North and South America drylands

plant species distributed in drylands over Americas, such as the Cactaceae in Sonoran Desert, Mexico and the southern United States; Caatinga in southwestern Andes and Pinus edulis in Canyonlands.

North America harbors a vast array of dryland ecosystems, including the Sonoran Desert, the northernmost drylands of the world, and the conifer taiga forests of Canada, etc. In Mexico and the southern United States, the Cactaceae family has the highest diversity. Forests of Pinus ponderosa and Pinus edulis are found all throughout the Colorado Plateau, with Pinus ponderosa and Pinus edulis being the most common species in the Canyonlands. Artemisia tridentata and Cercocarpus montanus, for example, might occasionally find a home in the open spaces between the trees (Maestre et al. 2021; Shreve 1942).

South America is home to a large area of important dry forests, mainly located in the Gran Chaco, the Maranhão Babaçu, and the Caatinga, as well as the driest forest of South America that features a xeric shrubland composed of succulents and thorny trees with a high *degree* of endemism (Fernandes et al. 2020). The Caatinga is also an important center accommodating the diversified Cactaceae family along the southwestern Andes (Ortega-Baes and Godínez-Alvarez 2006).

Soil moisture is an important environmental filter on plant species composition for dryland ecosystem, and drought is especially harmful to endangered species because of their narrow physiological tolerance and poor competitiveness (Bartholomeus et al. 2011). Future climate is expected to impact dryland plants, particularly threatening endangered plant species such as *Magnolia dealbata* in Mexico, *Artemisia tridentata Nutt.* in western U.S., and many other vascular plants.

Human activities like grazing, fire, deforestation, and farmland agriculture are also having resonant impacts on plants. These activities lead to the fragmentation or destruction of plant habitats, as well as the introduction of invasive competitors from other habitats (Garza et al. 2020). Habitat loss is the most widespread cause of species endangerment in some regions of America, including but not limited to *Tabebuia chrysantha*, *Astronium* graveolens, *Manihot walkerae* in U.S. and Mexico, and *Caesalpinia echinata Lam* along the Atlantic Coast. Apart from endangered plants, human activity explains a significant portion of variations in wildlife animals, such as terrestrial mammal in Argentinian. Intensified human activities could threaten species' persistence in biomes, which could be worse if climate changes act as a negative layer on biodiversity (de Oliveira et al. 2012).

Climate change and human activities pose the greatest threat to biodiversity in America drylands (Darkoh 2003), especially on those endangered species. Much work remains to disentangle the respective effects of the above two driving factors. American drylands are expected to experience increasing climatic aridity and land use pressure in the future (Ferner et al. 2018). To protect endangered species, identifying the factors that determine their distribution and abundance is critical (Amat et al. 2013).

Land Cover and Land Use

Grassland/cropland and shrublands are the two dominant vegetation types in the drylands of Americas as in other global drylands. In North America, the two land-use categories constitute 45% of the drylands in this region. In North America, the Great Plains represent a broad swath of the semiarid agroecosystem, bordered by Rocky Mountains to the west and high-rainfall areas to the east, stretching from the Canadian border in the north to Texas and New Mexico in the south (Hansen et al. 2013). Rainfed cropland, perennial cropland, irrigated cropland, and fallow are the several forms of dryland croplands in North America. The Canadian Prairies, the United States and Mexican Great Plains, and the inland Pacific Northwest of the United States with wheat are all areas of North America with high density dryland farming (Tritcum aestivum L.). Dryland farming is important in northern and central Mexico, mainly about the cultivation of maize (Zea maize L.), sorghum (Sorghum bicolor L.), pulses, and oil seeds, in addition to wheat. A two-year cycle of wheat and summer fallow is the traditional and still widely used farming strategy. The two most common cropping systems in the western margin of the south Great Plain are winter wheat (Triticum aestivum L.)-summer fallow and winter wheat-sorghum (Sorghum bicolor L.)-fallow. In South America, grassland and croplands together occupy 38% of the drylands, with grassland making up 81% of the total and cropland making up the remaining 19%, while other wooded land accounts for 45% of drylands and barren land for the remaining 17%. Two widespread uses of those lands are extensive livestock production or pastoralism and the rainfed or irrigated cropland. The rangelands include the Patagonian rangelands and the Dry Chaco rangelands, etc. Rangelands are the second primary land use in drylands of South America. For example, two thirds of continental Argentina are arid and semiarid rangelands. These rangelands include five phytogeographic regions: (1) Puna, (2) Chaco Occidental, (3) Monte, (4) Caldenal, and (5) Patagonia.

Scientific management and technology application on dryland agriculture represent a frontier line in American drylands, especially in North America. Even with the support from science and technology application, dryland agriculture suffered declines in agricultural productivity over the past few decades as a result of drought. Irrigation in Americas' dryland agricultural system is considered as a potential adaptation strategy to reduce the negative impact of drought on crop yields (Tack et al. 2017), and the sustainable irrigation strategies were widely applied in Great Plains to increase the water use efficient of crop (Comas et al. 2019; Himanshu et al. 2019). On the other hand, the agricultural insurance program in U.S. is the world's largest in premium volume. It has been developed since 1920s and then severed as a powerful and efficient tool to help secure the income of the American farmers as compared to other countries. Recently, the program was expanded to a wider horizon of crop products (Smith and Glauber 2012). From 2000, new private commercial agricultural insurance system was also introduced in Brazil and Chile to help the producers against losses due to disasters or price declines (Mahul and Stutley 2010).

As a lesser-known treasure, the North and South America's drylands are covered by extensive forests (Bastin et al. 2017; FAO 2010). In total, forests cover 37% of the region's drylands. The South America's drylands contain 197 million hectares of forest, which corresponds to 18% of the global dryland forest area and 5% of the global forest area. Forest area follows a clear decreasing gradient with increasing aridity. An estimated 61% of the dryland forest is in the dry subhumid zone, 38% in the semi-arid zone, 1% in the arid zone and less than 1% in the hyperarid zone. Forest is the second most common land use (30%) in North America's drylands. It comprises 206 million hectares of forests, equal to 19% of the global dryland forest area and 5% of the global forest area. More than half of the forests grow in the dry subhumid zone, and the remaining 41% grow in the semi-arid zone. A small portion (5%) is in the arid zone, and no forests are identified in the hyperarid zone. The forests of North America's drylands are composed of 40% coniferous, 38% broadleaved and 21% mixed coniferous and broadleaved. Forests in drylands generate a wealth of environmental services, which normally exhibit higher resilience in response to global changes than other vegetation types (Table 10.2).

Forests play a critical role in offsetting atmospheric CO_2 levels rising by sequestering CO_2 (Huang et al. 2020). U.S. initiates the first wave of forest carbon study in the 1980s (Sharpe and Johnson 1981; Cooper 1983). During 1990 and 2015, forest C stocks in North and Central America have increased, while that of South America has decreased substantially (Köhl et al. 2015). Vegetation in drylands can contribute

Table 10.2 Areas with forest and ≥10% tree canopy cover in the drylands in 2015. The estimations are based on satellite images and following the same definition of drylands (in mega hectares). Dashes indicate non-existing information for a given source because estimates are expressed either in terms of "tree cover" or in terms of "forest" (Bastin et al. 2017)

Source	FAO (2010)	Bastin et al. (2017)		
Sensor	Landsat	Very high-resolution imagery		
Method	Sampling	Sampling	Sampling	Sampling
Year	2005	2015		
Type	Forest	Forest	>20% tree cover	>10% tree cover
South America	123	197	192	208
North America	166	204	201	238

significantly to interannual variations of ecosystem carbon stock. Considering the high proportion of drylands area in America, assessing their capacity in sequestering carbon should be a research priority in the future dryland study.

10.2.3 Disturbance and Degradation

Two primary types of disturbances on grassland, savanna, and shrubland in the drylands are fire and grazing. Grazing is normally characterized as a combination of human interventions and herbivory (grazing or browsing by livestock and wildlife) in grassland, savanna, and shrublands. Rangelands are extensively managed to support grazing animals, whereas pastures are more intensively managed and may involve seeding, fertilization, irrigation, and weed control. On the other side, grassland and savanna in Americas are fire-prone ecosystems. There are multiple fire-dependent biomes distributed in Pantanal region, and the extensively distributed cerrado in Brazil, Venezuela, and Chile. Mesic savannas need fire to maintain their structure and biodiversity. In 2000 alone, savannah burning represented some 85% of the area burned in Latin American.

Land use change is the main type of disturbances on forests in South America (Abril et al. 2005). Conversion between soybean land and neotropical deforestation has existed in South America for a long time (Gasparri et al. 2013). According to satellite observations, 3.8% dryland forests disappeared between 2001 and 2010, mainly because of soybean cultivation and livestock production (Clark et al. 2012). As the largest tropical dry forest, Caatinga is considered as one of the most endangered ecosystems in the world. "Slash and burn" practices are traditional in this area, whose abandonment has caused soil salinization. Forest succession and health are highly dependent on frequent fire in North America. However, the series of human management, particularly fire suppression, logging, and livestock grazing, have totally modified their succession cycle and growth environment, and make them increasingly vulnerable to large-scale severe wildfires and insect pest.

A significant portion of dryland ecosystem are highly frequent fire adapted. With fire exclusion and suppression, woody encroachment has replaced grasslands in many places (Li et al. 2022; Miller et al. 2017; O'Connor et al. 2020). The vegetation transformation will in return reshape the fire regimes and alter the water and nutrition resource availability. Fire can not only affect the bi-stable dynamics between grasslands and shrublands, also is highly relevant to forest sustainability. Forests of western North America mainly consist of Ponderosa pine and dry mixed-conifer species. Those forests are subject to a relatively short fire return interval of less than 35 years. The frequent low-severity fires maintain the key compositional and structural elements in these forests, also helping remove the old-growth and overmature stands in achieving sustainable forestry (Hurteau et al. 2014). On the other hand, fire disturbance should be paid mounting attention in face of projected warmer and drier environment, as well as an extended drought period. "Precision restoration" such as logging to lower the unnatural high tree density and improve the diversity of tree species should be taken into consideration as a more reasonable conservation strategy (Copeland et al. 2021).

The modified fire regime also conveys high pressure on the sustainability of the social, economic and the ecological components. Both fire frequency and burned area increased across the Southwest of US, especially the high-severity fire occurrences in xeric mixed conifer and mesic mixed conifer/spruce-fire ecosystem from 1984 to 2015 (Singleton et al. 2019). The ecological and socio-economic impacts of fire have been increasing drastically in California in recent decades (Hurteau et al. 2014; Keeley and Syphard 2021; Miller et al. 2009). Such as in 2017 and 2018, the devastating fire years, 147 people died in fires, about 35,000 homes and businesses were destroyed, and approximately US$ 34 billion in insured properties were lost (Safford et al. 2022). These lessons teach us that we should put more focus on restoring key ecosystem function instead of suppressing fires for those fire-frequent ecosystems.

A large proportion of Americas' drylands have undergone some levels of degradation. Assessments by the USDA Natural Resource Conservation Service suggests that ~21% of the western rangeland area has been degraded to some degree (Herrick et al. 2010). Today very little, only 1–2%, of the original prairies still exist. Much of the prairies has been turned into agricultural uses (Squires 2018). In North America, the arid and semi-arid western rangelands, together with cultivated drylands of the southwest and Great Plains, comprise the regions of the United States most susceptible to wind erosion and associated soil loss. Specifically, the Great Plains are particularly prone to flash droughts from episodic precipitation deficits (Mo and Lettenmaier 2016). And projected future ecological drought has shown that the western Great Basin will face an increasing chronic drought stress (e.g., longer dry periods) (Bradford et al. 2020).

10.2.4 Dryland Livelihoods

In social dimensions, the multi-stakeholder of agriculture and livestock production systems engagement is needed for the sustainable management of drylands in Americas. Grazing livestock is the principal practice of exploiting natural vegetation in Americas' drylands. Because pastoralism is the sole practice that can simultaneously provide secure livelihoods, conserve ecosystem services, promote wildlife conservation, and honor cultural values and traditions, it is considered the most economically, culturally, and socially appropriate strategy for maintaining the well-beings of communities in drylands. The grazing industries and ranching systems are the prevailing land-based resource utilization model in drylands of North America. Livestock products are the main outputs of grazing lands and continue to be the fastest growing agricultural subsector. Central and South America provide 39% of the world's grassland-based meat production (beef) (Irisarri et al. 2019). Moreover, rangelands are coupled socioecological systems, shaped through interdependent land use practices and ecological processes. External forcing, such as those from regional precipitation patterns or episodic shocks, and the non-equilibrium nature of most rangelands systems (Reynolds et al. 2007) complicates the relationships among climate, management, and forage availability. Under the ongoing socioeconomic and environmental transformations in drylands, all these needs imply the necessity of cross-disciplinary work among livestock production, sociology, natural resources, economy, and rural development.

Except the agropastoralism in drylands as supporting the fundamental livelihood in Americas, the iterate biofuels production systems in the west of South America are promising. They not only have climate change mitigation potential, also can fulfill the desire for economic growth in the agriculture sector supported investment in biofuels as a rural development strategy (Correa et al. 2021). To minimize the conflicts between energy exploitation and biodiversity conservation, policy amendments and new governance initiatives have emphasized the social and environmental dimensions of biofuels. For example, the United States has modified their biofuel use targets and policies by adding sustainability requirements (Hunsberger et al. 2014).

10.2.5 The Economy of the Drylands in Americas

Human Population Over Drylands and Regional Variations

The gridded population data were obtained from the Socioeconomic Data and Applications Center (SEDAC). In 2015, North America and the South America population account for 13.51% of the global total. In North and South America, 26.8% of the total population live in the drylands (0.17 billion in North America and 0.09 in South America), mostly concentrated over the semi-arid drylands of both North America and South America. The average population densities in hyper-arid, arid, semi-arid, dry sub humid of North and South America are 8.6, 51.53, 245.85 and 32.23 person

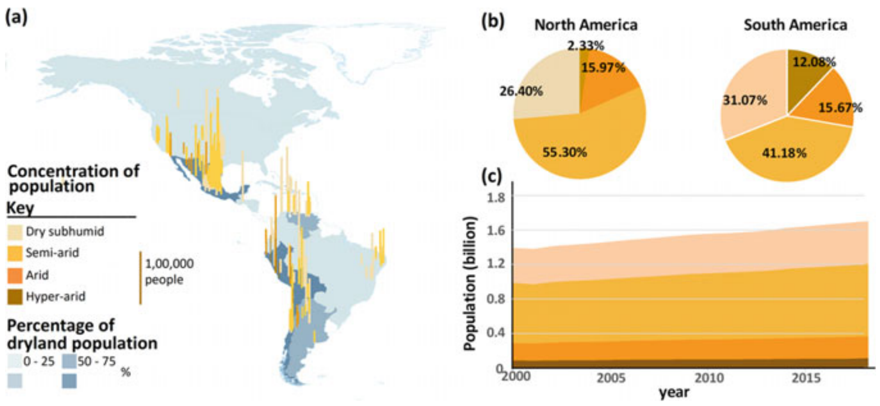

Fig. 10.5 Distribution of population over drylands in Americas. Countries without dryland distribution are not shown

per km^2, respectively. Mexico, home to the largest amount of population, is also the only country listed as the top ten largest dryland population countries over the world. Population in drylands (arid, semi-arid, and dry sub humid regions) of South and North America increased from 1.36 billion in 2000 to 1.71 billion in 2018 (Fig. 10.5).

Net-Migration from 2010 Through 2015 Over Dryland Regions

The Net migration (immigration minus emigration) is obtained through an indirect estimation technique, as the difference between population change and population natural growth. Net migration represents the difference between immigration and emigration (Fig. 10.6) (Neumann et al. 2015). Migration over drylands in Americas generally is in-migration. This trend indicates that the development conditions are beneficial for population growth, opposite to most of the other drylands over the world. Hyper arid and arid regions have the strongest appealing for the in-migrations, possibly caused by therein mega-city, such as Las Vegas and Phoenix in western U.S.

Artificial Lighting and GDP Reflected Human Activity

Both GDP and Nighttime lights can be used to indicate economic developments. Economic development can also be measured by Gross Domestic Product (GDP). According to the current developed global gridded GDP maps (Kummu et al. 2018), drylands account for more than 30% of the global GDP from 1990 to 2015 in Americas. The mean GDP are 0.31, 1.15, 3.59, and 1.65 $\times$ 10^{13} US Dollars in hyper-arid, arid, semi-arid, and dry sub-humid regions, respectively. GDP in drylands of Americas almost doubled since 1990, increasing from US\$ 3.6 $\times$ 10^{13} in 1990 to US\$ 6.7 $\times$ 10^{13} in 2015 (Fig. 10.7a–c). The GDP increasing rates were slightly higher in hyper-arid and arid regions (2.57 and 2.00%/yr) than in the semi-arid and arid sub-humid region (1.97 and 1.94%/yr) in Americas. Nighttime lights (NTL, the unit of Nighttime lights intensity is nW cm^{-2} sr^{-1}) generally represent the degree of urban socioeconomic development to some extent. The high NTL areas are mainly located

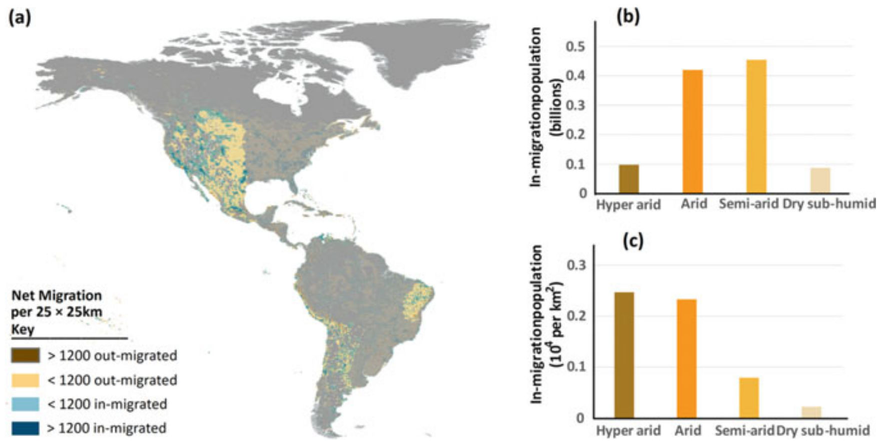

Fig. 10.6 The population migration over drylands in Americas. The upper right inset shows the total in-migration population of each sub-region of drylands, while the bottom right inset indicates the density of in-migration. Countries without drylands are not shown

in urban areas with high population density (Fig. 10.7d, e). The average intensity of nighttime lights were 0.98, 1.17, 0.61 and 1.00 in hyper arid, arid, semi-arid, and dry sub-humid regions in 1992. The average nighttime lights intensity were 2.31, 3.06, 2.14, and 2.76 in hyper-arid, arid, semi-arid, and dry sub-humid regions in 2015. The average nighttime lights intensity nearly tripled from 1.05 in 1992 to 2.43 in 2015. It is also interesting to note that the arid (0.025 nW cm^{-2} sr^{-1} yr^{-1}) and hyper-arid (0.023 nW cm^{-2} sr^{-1} yr^{-1}) regions became brighter at a doubled speed as compared to semi-arid (0.013 nW cm^{-2} sr^{-1} yr^{-1}) and dry sub-humid (0.011 nW cm^{-2} sr^{-1} yr^{-1}) regions during 1992–2010. This phenomenon indicates that the drylands in Americas experienced a balanced development as in other continents. A stable economy development in such western states of U.S. as California, Texas and Nevada contribute significantly to the social well-being boosting in drylands of North America.

10.3 Change and Driving Factor of Drylands in Americas

10.3.1 Dryland Climate Trends

Figure 10.8 shows the climate trends from 1982 to 2020. TEM and PET exhibit high correlations. Most drylands in North America exhibit significant warming trends (TEM, $p < 0.05$), which likely drive increased PET. PRE and ET have a similar pattern. PRE was observed to significantly decrease ($p < 0.05$) in North America over the past three decades, usually associated with decreases in ET.

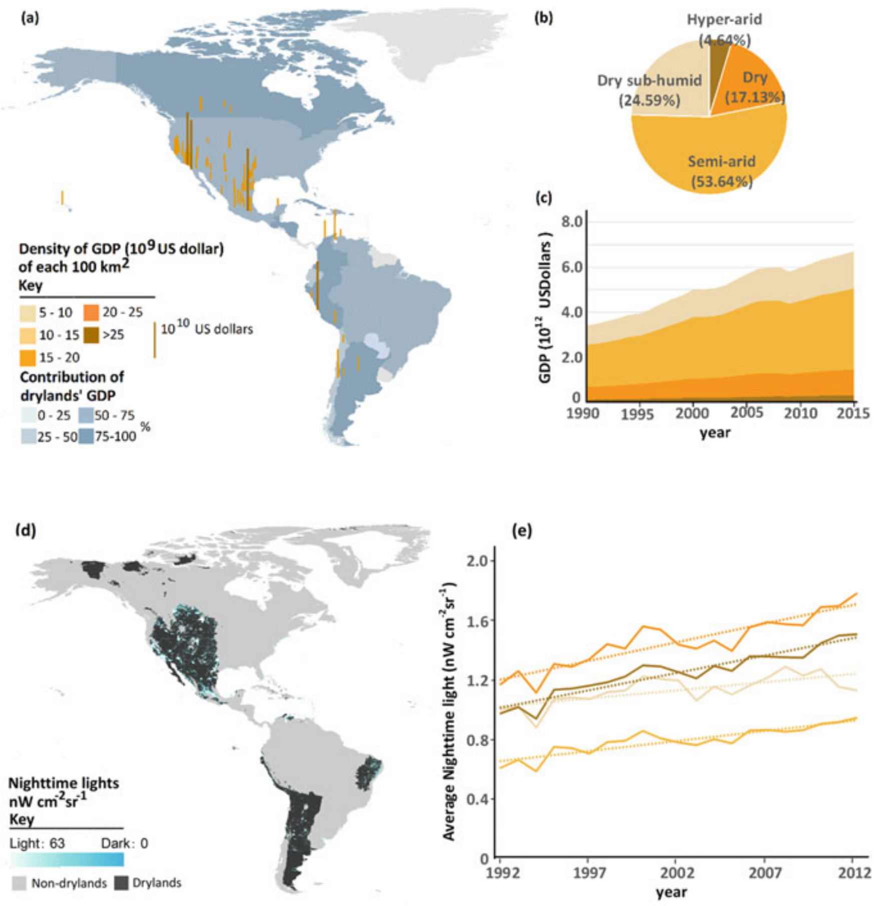

Fig. 10.7 **a** The GDP over drylands in 2015. Countries without drylands are not shown. **b** The contribution of GDP over different aridity level regions to the whole drylands GDP in Americas. **c** The total GDP changes in different aridity level regions in drylands of Americas. **d** The nightlight over drylands in Americas in 2015. Countries without dryland are shown as white. **e** Temporal variation of nighttime lights averaged in different aridity level regions in drylands of Americas. The lights detected are from cities and towns, gas flares, and fires

The spatial pattern of the SM trends is also roughly similar to that of AI. The spatial distribution of AI has similar trend with PRE but exhibits dissimilar pattern from that of the TEM trends. The drylands in eastern South America have become climatically wetting; but the southern North America and southern South America have become climatically drying. Simultaneously, the area ratio of drylands calculated in accordance with the standard of annual AI < 0.65 shows a significantly decreasing trend ($p < 0.05$) in most of North America, which indicates the area of drylands has been significantly reduced in Southwestern North America. SM shows

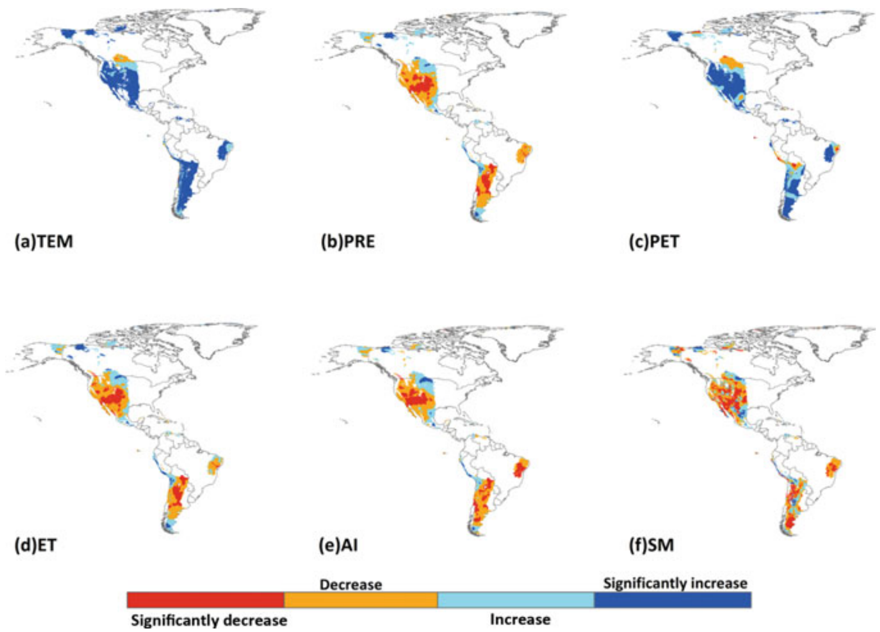

Fig. 10.8 Spatial distributions of trends in **a** temperature (TEM), **b** precipitation (PRE), **c** potential evapotranspiration (PET), **d** evapotranspiration (ET), **e** AI, and **f** soil moisture (SM) over drylands from 1982 to 2020

a significantly decreasing trend in southwestern North America drylands over the past four decades, indicating a decreased water yield over there.

Under the global context of warming, climate change will further exacerbate the vulnerability of dryland ecosystems by increasing PET globally. Warming is projected across American continent in the twenty-first century, and the most apparent will occur in winter of high latitude regions, where the greatest temperature increase approximates 15 °C in the vicinity of Hudson Bay (Maloney et al. 2014). Precipitation is projected to decrease significantly in the southwest of South America and south of North America (Cook et al. 2018). Mean annual rainfall can decrease by 8–14% in the Central United States under moderate to high emissions scenarios. Projected changes to drought characteristics under these scenarios are pronounced, with seasonal-scale droughts projected to lengthen by 12–30%, intensify by 17–42% and increase in frequency by 21–24% by the end of this century (Depsky and Pons 2021).

10.3.2 Land Cover Change and the Driving Force

North America drylands have been expanding, including semiarid and arid lands for 1997–2011 relative to 1982–1996. On the contrary, the southern portion of South

America has exhibited a wetting trend, resulting in the conversion from arid to semi-arid and hyper-arid to arid (He et al. 2019). By classifying land use change into types of a single event change (e.g., deforestation) or multiple events change (e.g., crop-grass rotation), we see clear patterns over South and North America (Fig. 10.9) (Winkler et al. 2021). About half of the areas are assigned to a single event change, such as deforestation in tropical South America. In contrast to single event changes, multiple event changes dominate in developed countries of North America (e.g., in the United States). Here, agricultural intensification (such as the United States) and/or major transitions in the agricultural sector, have taken place in the past few decades. Most agricultural land use changes (land transitions related to cropland or pasture/rangeland) occur in the form of multiple events change. Some of these changes are directly or indirectly linked to land management and agricultural intensification. The type of cropland-pasture/rangeland transitions can indicate areas of crop rotation or mixed crop-livestock systems as in the United States (Rosenzweig et al. 2018). Most multiple event land use changes occur between managed and unmanaged land, such as the abandonment of cropland.

Figure 10.10 shows land use/cover change dynamics (forest, cropland and pasture/rangeland) per 1 × 1 km grid cell from 1960 to 2019 (Winkler et al. 2021). The difference between North and South America is more pronounced in term of pasture/rangeland change, since pasture expansion in Brazil occurs in a large area while a

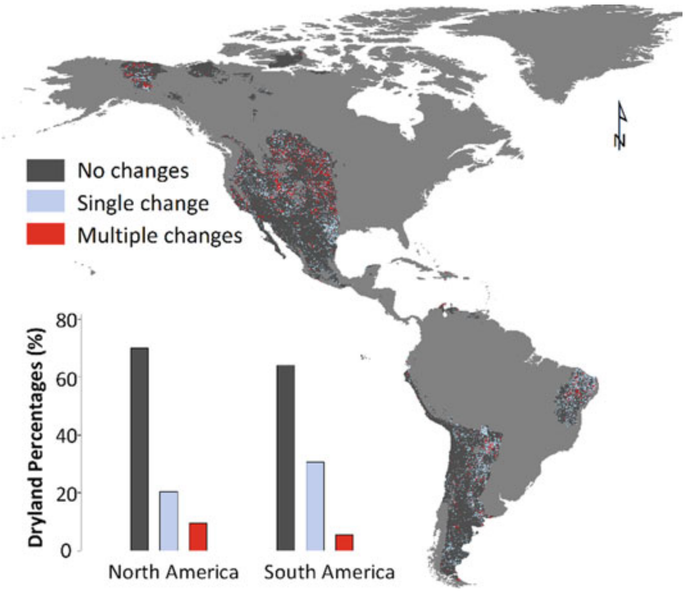

Fig. 10.9 Spatial extent of North and South America land use/cover change per 1 × 1 km grid cell from 1960 to 2019. The spatial extent of land use/cover change is displayed in light blue (areas with single event change) and red (areas with multiple event change) during 1960–2019. The bottom left barplot shows the percentage of land use/cover change over North and South America drylands

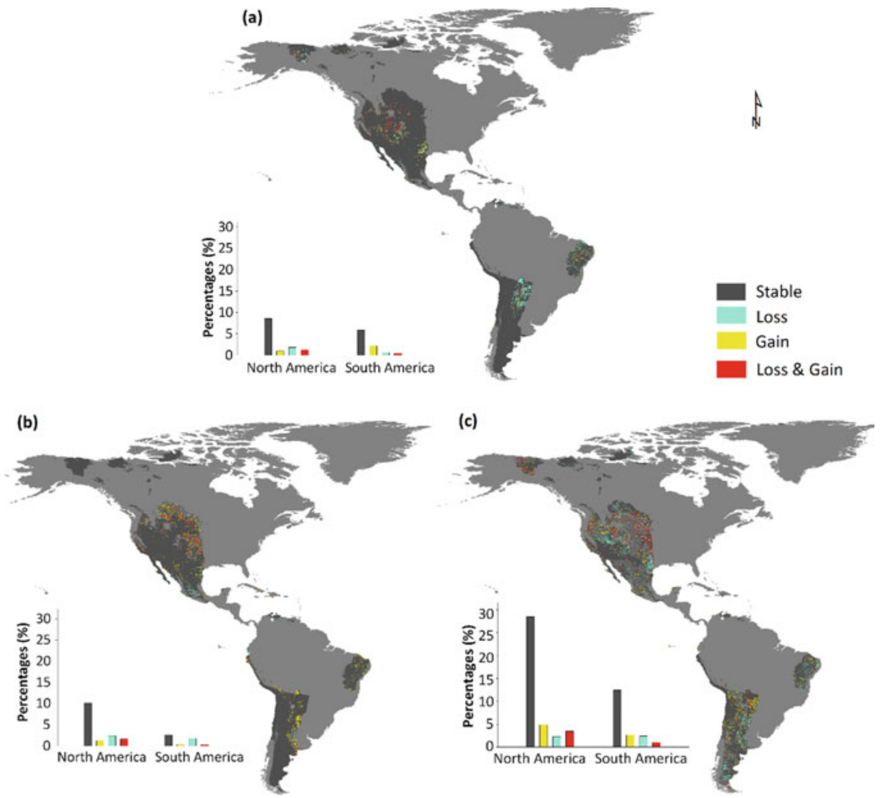

Fig. 10.10 North and South America forest, cropland and pasture/rangeland change. Spatial distribution of **a** forest, **b** cropland, and **c** pasture/rangeland extent (stable area) and change (gain and loss) during 1960–2019

widespread pasture was lost in North America. These land use change processes were supported and exemplified by numerous studies, e.g., agricultural land abandonment and woody encroachment of rangelands in the United States (Auken 2000; Ramankutty et al. 2010).

Global financial status also has a close relationship with the temporal dynamics of land use change. There was an abruptly slowed rate of land use change in South America since 2005. Before the financial crisis in 2005, rising demand stimulates global agricultural production, which in turn accelerates global land use change (Rajcaniova et al. 2014). The globally rising demand in the several developed countries of North America stimulates the expansion of bioenergy crop in South America (e.g., production of oil crops in Argentina, Brazil of South America). Global food price surges rapidly due to climatic extremes, biofuel policies, and export bans in 2007–2008 (Akram-Lodhi 2012) and 2010 (Bellemare 2015; D'Amour et al. 2016). In South America, land use changes are tightly associated with foreign investments and cross-border land acquisitions in agriculture (Arezki et al. 2015; Chen et al. 2017;

Krausmann and Langthaler 2019). The land use change follows a pattern of sudden increase (2000–2005), the subsequent fluctuations (during 2006–2010), and sharp decrease (after 2010), which demonstrates fluctuated developments in countries of South America, e.g., Brazil and Argentina. After the economic crisis of 2007–2009, the slowdown of land use change is mainly induced by a declined agricultural expansion, particularly in Argentina. With the end of the economic boom during the Great Recession, the reduced agricultural production has pushed higher the expansion rate of agricultural land in Argentina and Brazil.

10.3.3 Vegetation Structure/Function Changes and the Driving Factor

The natural climate and grazing are the two major factors determining drylands ecosystem structure and functioning in Americas' drylands. Increasing aridity is likely to aggravate imbalances among soil nutrient stoichiometry, and undermine Ecosystem functioning (Maestre et al. 2016). The intensified grazing and rising aridity have been widely reported to cause vegetation degradation (Eldridge et al. 2016).

Figure 10.11 shows that the overall NDVI of savannas demonstrates an increasing rate in South America. But forests mainly distributed in central South America have exhibited a significant decreasing trend. Overall, vegetation browning is observed in southern South America. North America dryland region displayed a significant greening trend on barren vegetated land, shrublands, and grasslands.

The drylands in North America experienced significant drying, where vegetation coverage has been increasing. The largest coverage increments were for croplands as a result of irrigation activities (Mueller et al. 2016) and increased SM. Grasslands in North America are also heavily irrigated. Significantly increased shrublands NDVI

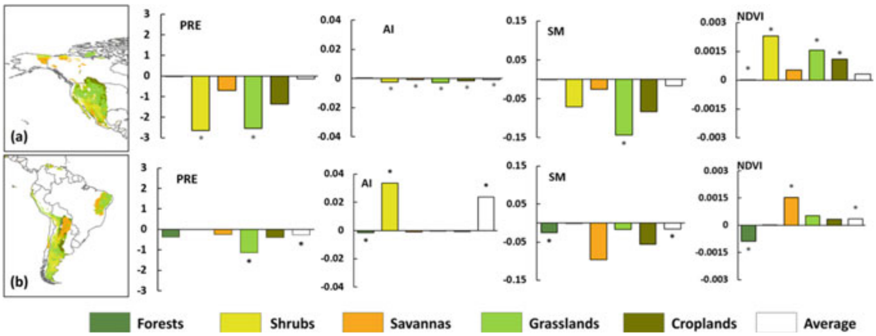

Fig. 10.11 Land cover types and the corresponding trends of PRE, AI, PRE-ET, SM, and NDVI for **a** North America and **b** South America during 1982–2020. * indicates a significant variation with P < 0.05, and ** indicates a highly significant with P < 0.01

were concentrated in the southwestern United States. Snowmelt in spring and summer is critical for vegetation growth in this area (Notaro et al. 2010), especially for shrubs that require deep soil water (Kurc and Benton 2010). The increased NDVI may be due to enriched water storage in deep soils stemmed from warming associated snowmelt. The above analysis suggests that both human activities and climate change contribute to the increased NDVI in the drylands of North America. Biological invasion has become widespread in the southwest United States (Herrick et al. 2010).

Obvious drying was observed in South America drylands forest, consequently decreasing therein vegetation growth. The drought-related NDVI reduction mainly occurs in forests, which causes ecosystem degradation. NDVI and drought indices showed a relatively high consistent trend in South America drylands. All the four drought indices point to dryness trends, but the average NDVI exhibits a weak rising, mostly caused by increased NDVI in the eastern savanna. The NDVI increasing is also observed in croplands, mostly related to irrigation practices (He et al. 2019).

The current droughts in South American are related to both El Niño and La Niña events, between which La Niña has played a more significant role. Warming atmosphere alone seems certain to make severe droughts more frequent, especially in Southwest South America (Voosen 2020).

Grazing is the most widespread land use in drylands, which provides food for a significant proportion of people worldwide (Asner et al. 2004). Grazing causes apparent effects on ecosystem structure and functioning in drylands (Hanke et al. 2014). Proper grazing rest, season-off-use, stocking rates, and subsequent management after fire are essential to restore resilient sagebrush ecosystems before they cross the breakdown threshold and become an annual grassland (Chambers et al. 2014; Miller et al. 2011).

The impacts of grazing on ecosystem are related to livestock type, grazing intensity, and some environmental factors. In North America grasslands, strengthened grazing intensity leads to a moderate expansion of bare soil soil (Augustine et al. 2012), while productivity and coverage of some grasslands can be partially increased by compensation growth, especially for grazing-resistant C_4 shortgrasses (Irisarri et al. 2016). Research shows that in Patagonian steppes (South America), sheep grazing alters the structure of plant communities. Compared to permanent grazing exclusion, moderate grazing keeps the sheep preferred plant species (Oñatibia and Aguiar 2019). At the same time, grazing impact on biodiversity is also regulated by different environmental factors. In North America, light and moderate grazing results in a decreased biodiversity in high-grassy grassland ecosystems with poor soil fertility and an increased biodiversity in high-grassy steppe with fertile soil (Fahnestock and Knapp 1994). For aboveground net primary Production (ANPP), in Argentina, ANPP based on live biomass increment is significantly higher in 4- and 15-year non-grazed sites than in 2-year grazed and 2-year non-grazed sites (Pucheta et al. 1998). Meanwhile, grazing intensity may regulate the response of ANPP to environmental factors. Studies have shown that the relationships between precipitation and ANPP are sensitive to grazing intensity (Irisarri et al. 2016). In North America, in the long-term grazed rangelands (>30 years), doubling grazing intensity in shortgrass steppe (SGS) and 175% increase in grazing intensity for northern mixed-grass prairie (NMP) reduce ANPP and precipitation-use efficiency (PUE) by approximately 24% and 33%, respectively (Irisarri et al. 2016).

Grazing also has a certain effect on livestock production in the semi-arid short grass prairies of North America. Beef production grows with increased grazing intensity in normal moisture conditions or wet years, but causes no increase in drought years (Irisarri et al. 2019). In Patagonian rangelands, compared with continuous grazing management (CGM), the weight of animals under holistic grazing management (HGM) is reduced and the body condition scores of HGM is also lower than that of GCM (Oliva et al. 2021). In addition to the above-mentioned, there are also impacts of growing grazing costs in North America. Riverbanks are the most bio-abundant zones in arid and semi-arid regions. Livestock will choose to live along riverbanks most of the time. Then the ecological risk will correspondingly escalate. Under this grazing mode, the adverse effects of grazing are amplified and need to be addressed (Fleischner 1994). Overall, to adapt to the changing climate and promote sustainable development, appropriate climate prediction tools are critical for managing rangelands. Also the quantity and quality of the current and predicted food need to be incorporated into the grazing management plan (Derner and Augustine 2016).

10.3.4 Carbon Dynamic and Nitrogen Dynamics

Gross primary production (GPP) is a key component of ecosystem carbon cycle (Fig. 10.12). The average annual GPP of North America drylands is 0.50 kg C m^{-2} yr^{-1}, which is more than double the value in South Americas (0.20 kg C m^{-2} yr^{-1}). In 2020, the mean annual GPP of forest, shrublands, savanna, grassland, and cropland in North Americas is 1.21, 0.30, 0.72, 0.46, 0.70 kg C m^{-2} yr^{-1}, respectively. The mean annual GPP of forest, shrublands, savanna, grassland, and cropland is 1.63, 0.32, 1.2, 0.84, 0.97 kg C m^{-2} yr^{-1} in South Americas, respectively. During the last two decades, nearly 87.1% of the drylands in Americas show growing vegetation GPP. The average vegetation GPP has increased from 0.17 to 0.20 kg C m^{-2} yr^{-1} and from 0.42 to 0.5 kg C m^{-2} between 2001 and 2020 in South Americas and North America, respectively. The savanna and cropland in North America show a significant ecosystem GPP growth at a rate of 5.12 g C m^{-2} yr^{-2} and 7.95 g C m^{-2} yr^{-2} during the last two decades, respectively. The average annual GPP are increased at a rate of 4.7 g C m^{-2} yr^{-2} and 1.6 g C m^{-2} yr^{-2} for drylands in North America and South America, respectively.

Climate change effects on GPP trends contain much uncertainty. Increased GPP around North and South America is mainly due to elevated atmospheric CO_2 concentration, except for some small parts of the Brazilian plateau (Sun et al. 2019). In the temperate steppe ecosystems of North America, precipitation significantly promotes vegetation growth, also GPP (Sun et al. 2019). In western North America with high water stress, the spatial continuity of GPP sensitivity to precipitation is not significant (Sun et al. 2019). GPP is more limited by water constraints through decreased SM and increased VPD in western and central United States (Madani et al. 2020).

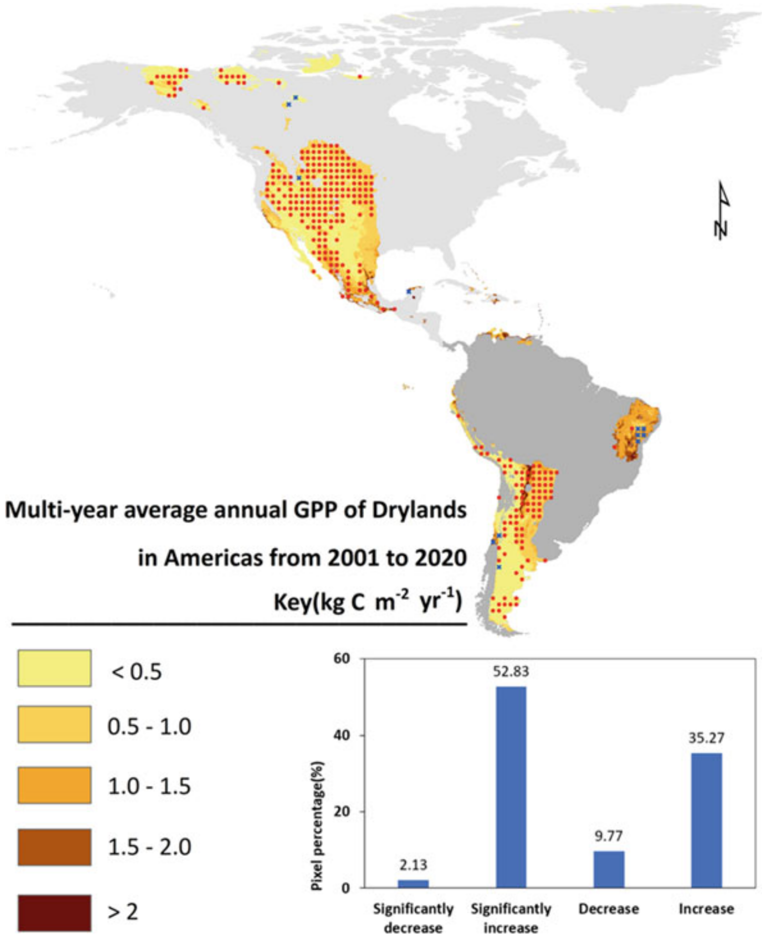

Fig. 10.12 The multi-year average GPP of drylands in Americas based on MODIS17H2 datasets. The • and X indicates the significant increased and decreased GPP area, respectively. The bottom panel shows the percentage of pixels exhibiting increasing and decreasing ecosystem GPP over the drylands in Americas

GPP increment trends are mainly regulated by increased solar radiation and temperature in humid temperate North America, and in many dry forest regions of South America, land-cover change is responsible for reduced GPP (Sun et al. 2018). Forest loss rates in temperate North America are relatively low, causing a lower impact on GPP than in South America (Sun et al. 2018). Rising temperatures play a primary role in stimulating GPP in northern high latitudes, while it suppresses ecosystem in South America (Cai and Prentice 2020).

Both soil and vegetation carbon storage in drylands contributes considerably to the terrestrial carbon storage. Soil organic matter levels in the top soil are mainly negatively correlated with mean annual temperature and positively correlated with precipitations in the Northwest agriculture systems of inland USA (Morrow et al. 2017). The land cover shifts, such as grasslands encroachment into drylands tend to boost soil organic carbon content. However, woody invasion tends to boost soil organic carbon content in semiarid and subhumid drylands while decreasing it in arid drylands in North America (Barger et al. 2011). Research has found that woody plant encroachment shifted soil organic carbon from an annual loss of 6200 g C m^{-2} to annual gains of 2700 g C m^{-2}, with an annual average accumulation of 385 g C m^{-2} in North American drylands (Barger et al. 2011).

Nitrogen availability is the second critical factor limiting dryland ecosystem primary productivity after water availability (Hooper and Johnson 1999; Yan et al. 2010). Even though the belowground parts account for more than half of the total net primary productivity in the drylands, studies across three typical dryland ecosystems in North Americas show that as compared to aboveground productivity, root productivity is less responsive to nitrogen addition (Swindon et al. 2019). Nitrogen availability is influenced by climate change and human activities. The predicted aridity exacerbation will reduce the nitrogen concentrations in the global drylands; however, it is still not clear how aridity change will impact the nitrogen content in America drylands. Legume shrubs expand markedly in the dryland crop system of the Northern Plains and the Pacific Northwest United States during the cool season. One significant reason is due to their strong capacity as soil nitrogen fixers (Arash et al. 2018). Nitrogen fixation by biocrusts, which covers a large proportion of soil surface in low-nutrient drylands, also contributes significantly to ecosystem nitrogen fixation (Baldarelli et al. 2021; Weber et al. 2015). On the other hand, nitrogen deposition in the temperate N-limited dryland ecosystem set the stage for more possible invasion by nitrophilic grasses (Vallano et al. 2012). Increasing nitrogen pollution is also found to be the primary factor causing 78 listed or candidate species as threatened or endangered in serpentine grasslands of California Bay (Hernández et al. 2016).

10.4 Managing Drylands in Americas: Challenges and Opportunities

10.4.1 Major Issues in Managing Drylands in Americas

Desertification is the most threatening ecosystem change that affects the livelihoods of local people. Due to its close linkage with land degradation, persistent desertification may further lead to the loss of human well-beings. After desertification, woody encroachment and soil erosion also pose serious threat to Americas' drylands by lowering diversity and undermining ecosystem services.

Woody encroachment, perhaps the most dramatic form of dryland vegetation cover change, continues to expand over extensive drylands of the United States and South America (Rosan et al. 2019). The invasive distribution of buffelgrass, which are highly productive in drylands, has expanded to 53% of Sonora State and

12% of semi-arid and arid ecosystems in the Sonoran Desert of Mexico (Arriaga et al. 2004). Sequentially, plant–plant and plant-soil interactions are adjusted and landscape structure and functions are modified (Franklin and Molina-Freaner 2010).

For Americas' drylands, another widespread environmental issue is the exacerbating soil erosion caused by the land cover transition from grassland to shrubland. During the second half of the nineteenth century, large-scale commercial stock-breeding quickly spread to the North and South America semiarid drylands, which caused severe ecosystem degradation on drylands and affected millions of people as happened in the other developing countries (Reynolds et al. 2007). The most recent climatic projections predict that the global dryland area will expand 11–23% by the end of this century (Huang et al. 2016b). Then soil erosion affected areas are likely to further expand under climate change and population growth (Safriel et al. 2005). Some countries in South America are faced with especially severe soil erosion issue. The national assessments conducted in 1979 revealed that soil erosion severely affected 36% of Chile's territory and the affected areas are still expanding.

The intensified land use practices and rapid land-use change pose a rapid growing threat to both plant and soil diversity (Kobayashi et al. 2019). The living organisms in the top soil layer, such as mosses, lichens and other microorganisms, are normally used to reflect the soil diversity. Soil diversity contributes significantly to vegetation growth by maintaining soil fertility, while soil erosion causes the decrease of soil diversity. Soil with lower soil diversity is incapable of supporting the mismatching high vegetation diversity, which in turn decreases soil carbon. In semi-arid grassland, adding nutrients to the soil can slow down the loss of plant diversity (Harpole et al. 2016). The high-intensity grazing can also lead to the loss of the native plant diversity, particularly in combination with extreme climatic events, such as drought (Souther et al. 2020). According to recent studies and assessments of current and anticipated climate changes in the Great Plains, it is also suggested that rural people and ecosystems are more and more sensitive to changes brought on by warming, droughts, and increased variability in precipitation (Ojima et al. 2021).

Water resource scarcity is typical for drylands in South America. Numerous rivers or catchment are fed by melting snow and glaciers, and their flows or runoff have been significantly affected by global warming. Glaciers are served as the water resource buffer for ecosystems, locking up precipitation during the rainy season and releasing water slowly during the dry season. The glacier retreat or shrinkage, and early snow melt will change the seasonal accessibility to water resources (Young et al. 2010) and exacerbate the vulnerability of the dryland ecosystems. Construction of small reservoirs that could be tapped in the dry season could just be "part of the answer". This also raises up the importance of adapting to the present land and resource management styles in the face of the unprepared changes.

10.4.2 Sustainable Managing Drylands: Conservation Agriculture, Husbandry, and National Park System

Biodiversity conservation is one of the most important goals for sustainable drylands management. The overarching government regulations are needed to guide the sustainable management in drylands by various stakeholders to gain multifunctional use of drylands. The US government has announced millions in rewards for conservation partners each year for agriculture and husbandry innovations, supporting improvements in managing land efficiency and environment protection. The natural resources conservation programs of the Natural Resources Conservation Service (NRCS) encourage reducing soil erosion, improving wildlife habitat, and providing financial supports to private rangelands and farmlands.

The stable foundations for ecosystem services for agriculture are provided by the health and fertility of the soils in the Americas. The intensification and diversity of cropping systems, on the other hand, are crucial for maximizing farming's short-term earnings, but they also constitute a serious threat to the sustainable management of the land. Thus, adopting sustainable land management practices, such as the use of Conservation Agriculture (CA) is growing in dryland agriculture (Shrestha et al. 2020). CA is characterized by minimum soil disturbance, crop rotation, and maintaining a certain degree of permanent soil cover. According to updated figures published by FAO, the U.S. is leading the list of countries with more absolute areas under CA. In South America, the adoption of CA has been especially quick. The MERCOSUR countries (Argentina, Brazil, Paraguay, and Uruguay) in South Americas are amongst the top five countries in terms of surface area protection using CA in the world (Shrestha et al. 2020).

For husbandry, many management practices try to keep the disturbance-driven heterogeneity characteristics of rangelands for maintaining forage diversity in rangelands. For example, cross-fencing and winter-patch graze are included in the conservation plans with the NRCS, proved efficient in improving soil carbon levels and ranch profitability (Buckley et al. 2021; Derner et al. 2018). To sustain wildlife and ecosystems in balance with human livelihoods, the patch-burn grazing has been extensively promoted in North American (Scasta et al. 2016). It can be an alternative management approach in fire-prone ecosystems to optimize both livestock production, ecosystem functioning, and biodiversity conservation (Ricketts and Sandercock 2016). On the other hand, the adaptive capacity of rangelands and grassland communities to support the local diversity is also highly variable. A comprehensive socio-ecological system (SES) framework, with indicators and links to key outcomes related to livelihood and ecosystem process running, is critical in improving evaluation of climate and land use effects changes on husbandry (Ojima et al. 2020), thereby facilitating management actions during husbandry.

Studies have shown that, biodiversity is substantially higher within the well-managed reserves as compared to the public lands (Gray et al. 2016). At the country level, to achieve the ultimate goal for protecting biodiversity and sustaining ecosystem services the drylands provide, American governments designate high

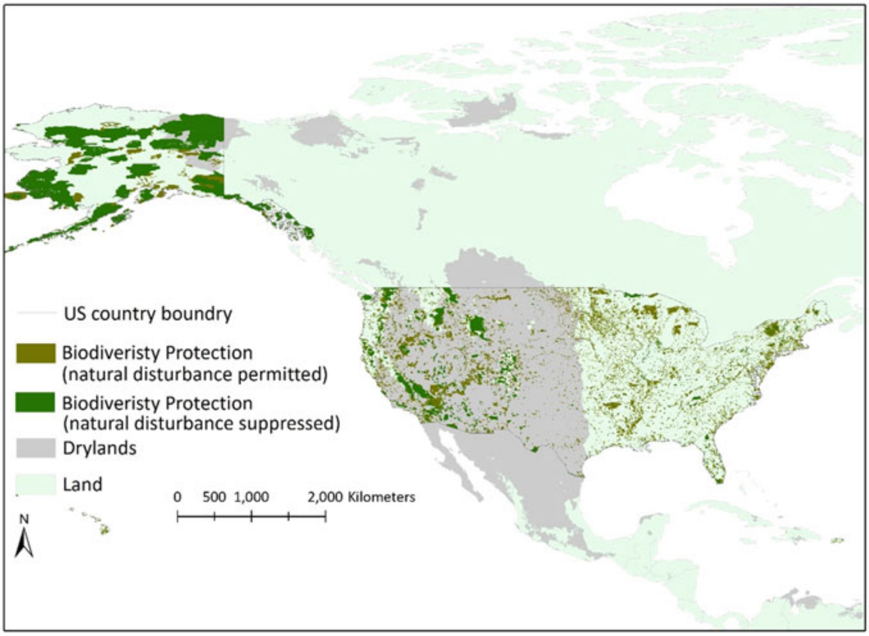

Fig. 10.13 Distribution of biodiversity protection lands in USA (US Geological Survey GAP Analysis Program)

percentage of lands as protected by national parks system. In the USA, drylands make up more than one-third of the natural disturbance permitted biodiversity protection lands, and one-fourth of the naturally disturbed biodiversity protection areas are also scattered there (Fig. 10.13). Among those protected areas, biodiversity conservation is always listed as the top priority goal. Those ecoregion-based managements provide further aid to safeguarding critical species and their diverse habitats. Research also suggests that considerable investments should be directed to private land conservation and encourage the engagement with local stakeholders, consequently increasing the success of endangered species protection (Clancy et al. 2020). As most drylands in western North America and Southern Latin America are exposed to slow climate velocity and located in high land-use instability areas, prioritizing protection, restoration and maintaining the connectivity among protected area networks will be highly beneficial as compared to other drylands such as in European Union (Asamoah et al. 2021). Therefore, policy makers and multiple stakeholder groups, such as scientists, the public, and other private sectors, should cooperate effectively to achieve the restoration and protection goals in America drylands.

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Chapter 11
Dryland Social-Ecological Systems in Australia

Xiaoming Feng, Yongzhe Chen, Fangli Wei, Zhihong Xu, Nan Lu, and Yihe Lu

Abstract Dryland social-ecological systems in Australia are characterized by a water-limited climate, vulnerable terrestrial ecosystems, advanced ecosystem management, and the highest average wealth. Dryland social-ecological systems in Australia have been facing the accelerated warming and rapid socioeconomic developments since the twenty-first century, including GDP increases and urban development, but with great diversity. Ecosystem structures and ecosystem services are highly influenced by extreme climate events. According to the number of extreme high daily precipitation events, droughts and floods have increased rapidly since the 1970s. Australia has achieved successful grazing, fire, biodiversity, and water resource management; climate change mitigation; and ecosystem management methods of community engagement. Non-indigenous population ageing is a social threat of dryland social-ecological systems in Australia in recent decades. The integration of policy makers, funding agencies, and the general public is essential for Australia's dryland social-ecological systems.

Keywords Advanced ecosystem management · Australia · Extreme climate events · High wealth · Population ageing

11.1 Introduction

Australia (113°08′E–153°38′E, 10°41′S–43°38′S), has a terrestrial land area of almost 7.7 million km^2, which includes the Australian continent mainland, the island of Tasmania, and numerous small islands. Australia is the driest inhabited continent in the world (Commonwealth of Australia 2012). The arid Australian climate can

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be attributed to the subtropical anticyclonic zone which covers the center of the continent; the Great Dividing Range which blocks water vapour from the east coast; and the West Australia Current, a cold current that significantly reduces precipitation in Western Australia. The climate in Australia is highly variable, with frequent drought events throughout the country. The climatic fluctuation and extreme climates in Australia are mainly driven by ocean currents, including the Indian Ocean Dipole and the El Niño–Southern Oscillation. Australia is dominated by drylands (aridity index < 0.65; 733.9 × 10^4 km^2; 95.4% of terrestrial Australia), with water-sufficient areas existing only on Tasmania Island, in the eastern and northern coastal areas, and in the southwest corner. More than half of the country (65.2%) is composed of arid regions, followed by semiarid (25.5%), dry subhumid (4.7%), and humid (4.6%) regions, while hyper-arid areas are 0.007% of Australia area (Fig. 11.1). The arid Australian climate gives rise to a specialized and quite vulnerable terrestrial ecosystem which can be characterized by pervasive deserts and sparse grasslands. Approximately 80% of terrestrial Australia is classified as rangelands, where land use is dominated by extensive grazing of sheep and cattle (Feng et al. 2020; Foran et al. 2019).

In spite of the water-limited climate conditions and vulnerable terrestrial ecosystems, Australia has the highest average wealth, and the GDP per capita was approximately 5.74 × 10^4 US dollars in 2018. Australia has a population of nearly 26 million, equalling an average population density of 3.4 per km^2. The population is highly concentrated in cities on the eastern seaboard. Population distribution pattern outside the main cities are of a few medium size towns and then many very small

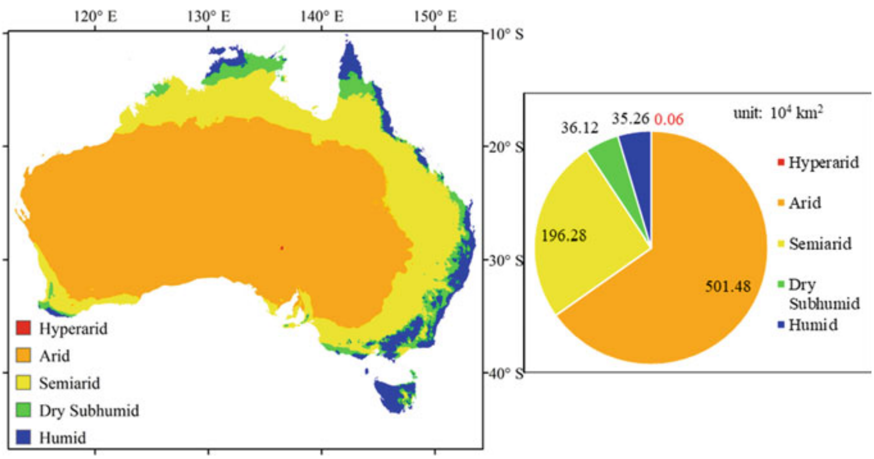

Fig. 11.1 Distribution of drylands in Australia. The standard global aridity index (AI, the ratio of long-term average annual precipitation to average annual potential evapotranspiration) map is from the Global Aridity and Potential Evapotranspiration (PET) Database v2 developed by the Consultative Group for International Agriculture Research-Consortium for Spatial Information (CGIAR-CSI) (Trabucco and Zomer 2019). AI ranges are 0–0.03, 0.03–0.2, 0.2–0.5, 0.65–0.65 and >0.65 for hyper-arid, arid, semiarid, dry subhumid, and humid regions, respectively

communities. Dryland social-ecological systems (SES) in Australia are threatened by the degradation of rangelands due to more arid climates and excessive grazing. Moreover, agricultural expansion, especially poor irrigation activities in areas with high potential evapotranspiration but limited rainfall, has led to dryland salinity, which is a key problem contributing to land degradation in southern Australia (Clarke et al. 2002; Lambers 2003). Human society in terms of population distribution, economic development, and the livelihoods of local communities, is greatly affected by water deficits and drought-induced ecosystem degradation. Therefore, in this chapter, we would like to provide an overview of the spatiotemporal dynamics of climate, ecosystems, and human society in Australia, especially during recent decades, and explore the relationships among these three key components. Multiple datasets on environmental conditions, vegetation cover, and human society or activities are analysed, and published studies are referenced.

11.2 Major Characteristics of Dryland Social-Ecological Systems in Australia

11.2.1 Climate Conditions

The mean annual temperature (MAT) is the lowest in the southeastern part of Australia, e.g., Tasmania Island. In tropical areas in northern Australia, the weather is perennially hot, whereas in the interior of the continent, which is covered by the arid anticyclone, summers are extremely hot, and winters are cool. Extreme temperatures influence vegetation, animals, and even humans (Cheng et al. 2018; Ebi et al. 2021; Hoffmann et al. 2019). Annual precipitation is the lowest in central Australia and high in northern Australia and some coastal regions. The precipitation seasonality declines from north to south in Australia and is higher on the southwestern coast than in the southeastern regions. For most parts of Australia, precipitation is the highest in summers and the lowest in winters. However, western and southern Australia showed the opposite pattern of seasonal precipitation variation. Therefore, over the southwest corner and Spencer Bay (including Kangaroo Island) located in southern Australia, where precipitation seasonality is large, typical Mediterranean climates are present. Mean annual solar radiation is generally higher in the north than in the south, but the seasonality of radiation increases with latitude. Desert areas receive higher radiation than relatively humid places (Fig. 11.2).

The dryland climates in Australia (hereinafter ‘DRY AUS’) can be classified into 10 types (Fig. 11.3) according to the world map of the Köppen-Geiger climate classification (Kottek et al. 2006; Rubel et al. 2017). The central and western parts of Australia are dominated by a tropical desert climate (BWk), the northern coasts are hot year-round and dry in winter (Aw), most parts of the eastern coasts have warm and humid weather (Cfa and Cfb), and Mediterranean climates (Csa and Csb, warm and dry summers) dominate the southwest corner and some areas in southern

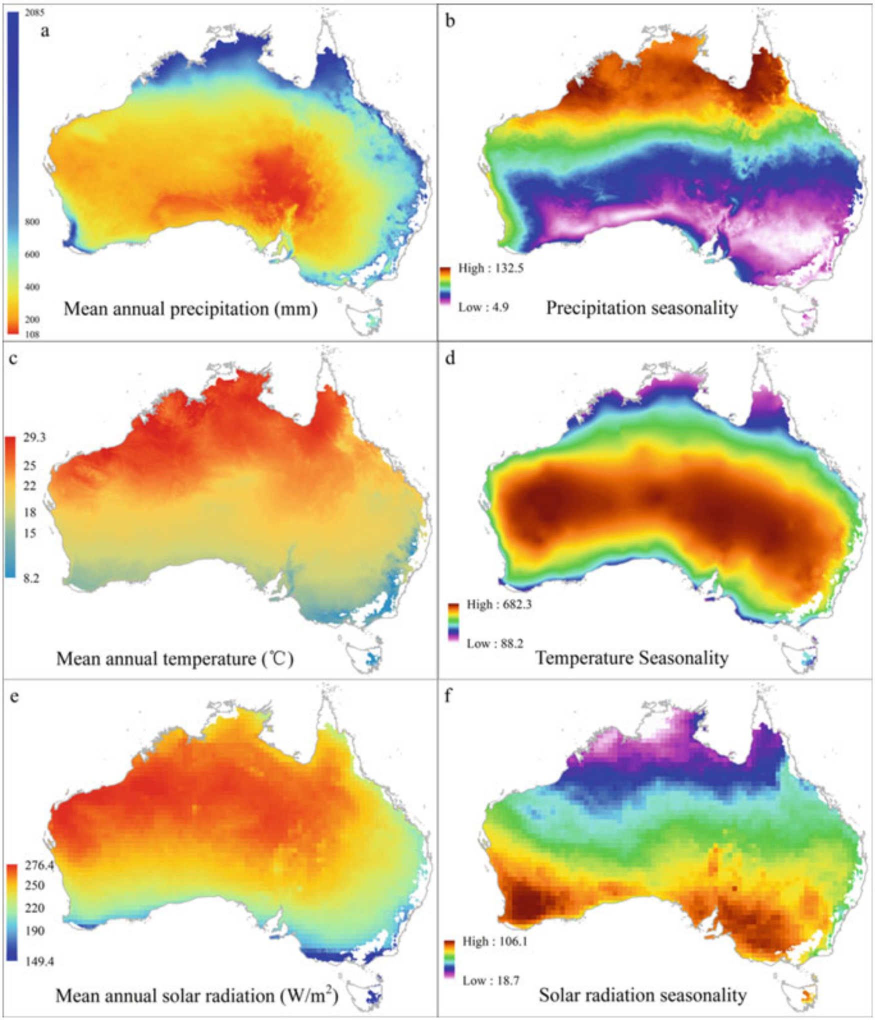

Fig. 11.2 Spatial patterns of temperature, precipitation and solar radiation and the seasonality in the drylands of Australia. The data are from the Australian Gridded Climate Data (AGCD), which are newly published by the Bureau of Meteorology Australia at the national computational infrastructure (NCI), as the successor of the Australian Water Availability Project (AWAP). The monthly precipitation data are available from 1900 until 2019 (Australian Bureau of Meteorology 2020b), whereas the monthly means of daily minimum and maximum air temperature data are available from 1910 to 2019 (Australian Bureau of Meteorology 2020a). Both temperature and precipitation data have a spatial resolution of 0.05°. Solar radiation data are from the updated Breathing Earth System Simulator (BESS)-RAD dataset from 2000 to 2019 and have a high accuracy (Ryu et al. 2018)

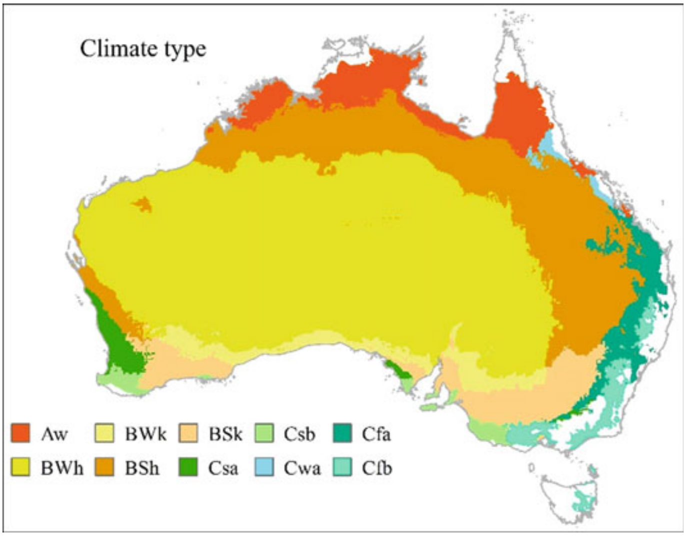

Fig. 11.3 Köppen-Geiger climate classification in the drylands of Australia (In the symbol in the legend, the first letter indicates the *main climates*: A: equatorial; B: arid; C: warm temperate; the second letter indicates *precipitation*: W: desert; S: steppe; f: fully humid; s: summer dry; w: winter dry; and the third letter represents *temperature*: h: hot arid; k: cold arid; a: hot summer; b: warm summer)

Australia. The mean annual air temperature increases with the aridity level, but precipitation declines with the aridity level (Fig. 11.6a). The interrelationships among the interannual variations in solar radiation, temperature, and precipitation are all stronger in more arid regions (according to data from 2000 to 2019). Precipitation is negatively correlated with solar radiation ($p < 0.01$) in all regions. Temperature is positively correlated with radiation and negatively correlated with precipitation in all drylands in Australia, but these relationships are not significant in dry subhumid areas.

11.2.2 Soil and Topography

Australia has the lowest and flattest topography among all continents. However, eastern Australia is marked by the Great Dividing Range, which stretches more than 3500 km and has widths from 160 km to more than 300 km. The heights of the range are typically 300–1,600 m. The southern Great Dividing Range contains the highest place in mainland Australia: Mount Kosciuszko (2228 m above sea level). Except for eastern Australia, where the silt or clay fraction is relatively high in soils, sand dominates the surface soil in the drylands of Australia (Fig. 11.3). Soil organic

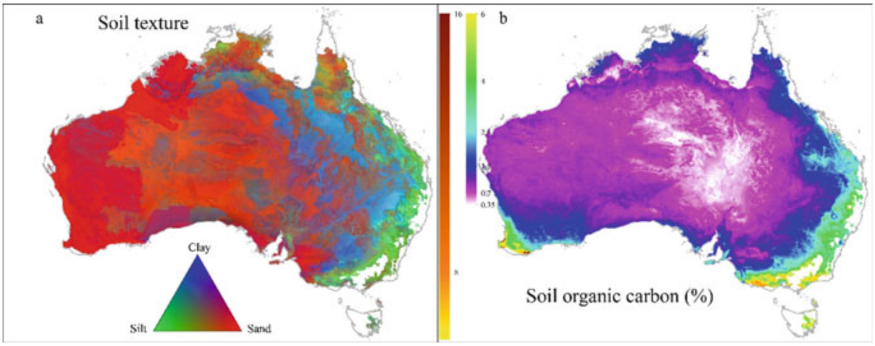

Fig. 11.4 Soil properties of drylands in Australia. **a** Soil texture and the fractions of sand, silt and clay; **b** spatial pattern of soil organic carbon. Soil properties are from the national soil attribute maps produced by the Soils and Landscape Grid Facility of Commonwealth Scientific and Industrial Research Organization (CSIRO) and soil mapping (Odgers et al. 2015a, b; Viscarra Rossel et al. 2015). Bulk density, pH, available water capacity, total nitrogen, and total phosphorus of the soil layer (0–5 cm) are provided in this dataset (Viscarra Rossel et al. 2014)

carbon is high in the eastern coast and southwest corner of Australia but is quite low in the interior parts of the continent, especially in the desert areas (Fig. 11.4).

11.2.3 Land Use/Cover in Dryland Regions in Australia

Drylands in Australia are dominated by sparse and scattered grasses and shrubs (37.1%), followed by open shrublands (10.5%) and sparse trees (9.8%), all of which are typical ecosystem types in arid climates. Shrublands and grasslands are representatives of arid and semiarid regions in Australia and rarely exist in more humid places. On the other hand, closed forests mainly exist in humid and dry subhumid regions, while open forests can be found in semiarid, dry subhumid, and humid regions (see Fig. 11.5). Vegetation cover is much denser in more humid coastal areas. From the humid coasts to the dry interior lands, the ground cover changes from forest to grass and finally to bare ground (Fig. 11.6).

11.2.4 Socioeconomic Factors

Australia's population was 25,704,340 on 31 March 2021 according to the Australian Bureau of Statistics. The population in Australia is concentrated in humid regions, especially urban areas located on the southeast and southwest coasts (Fig. 11.7a). The average population densities in the arid, semiarid, dry subhumid, and humid regions of Australia were 0.12, 4.05, 15.98, and 21.81 individuals per km^2, respectively. The

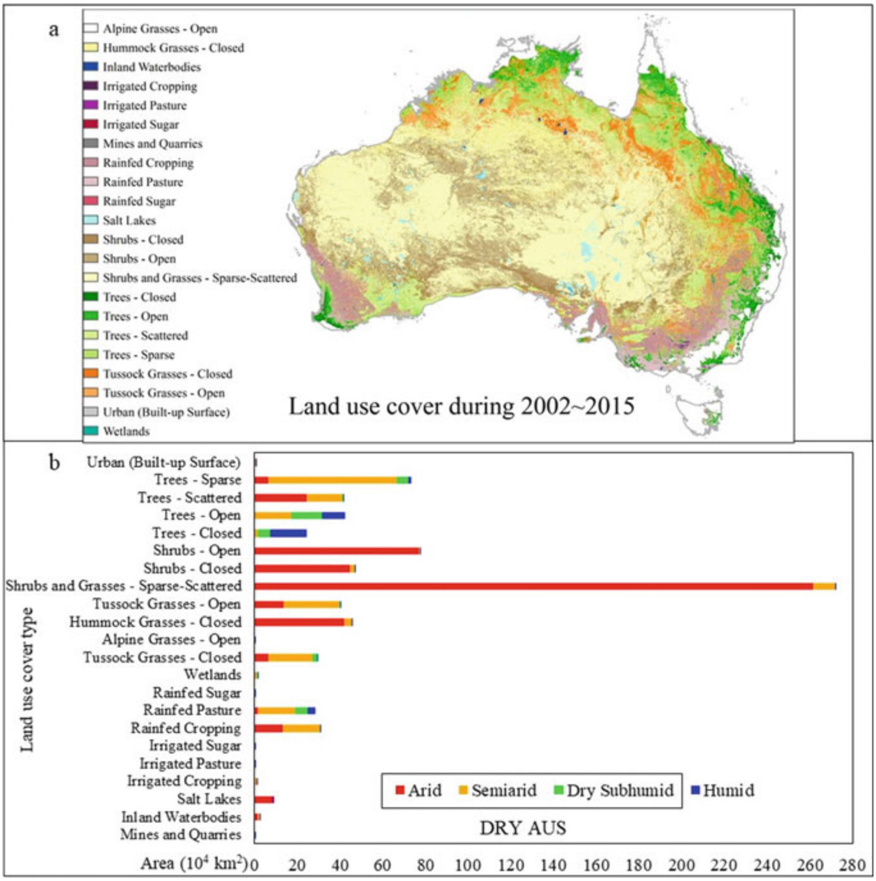

Fig. 11.5 Land cover types in drylands in Australia: **a** most of the land cover in the drylands of Australia during 2002–2015; **b** annual mean total area of 22 land use cover types in different regions distinguished by aridity levels in Australia. Please note that the drylands of Australia, denoted by DRY AUS, consist of arid, semiarid, and dry subhumid regions. The data are from the National Dynamic Land Cover Dataset (NLCD) v2.1, published by the Australian Bureau of Agricultural and Resource Economics and Science (Lymburner et al. 2015)

population in the drylands (arid, semiarid, and dry subhumid regions) of Australia increased from 12.33 million in 2000 to 16.52 million in approximately 2020. The mean gross domestic production (GDP) values per area were 0.41, 14.40, 52.93, and 66.05 × 10^4 US\$/km^2 in the arid, semiarid, dry subhumid, and humid regions of Australia, respectively (Fig. 11.7b).

Grazing (beef cattle/sheep) is a key industry in Australia. Meat and wool production contributed almost 30% of the gross agricultural production value in 2009–2010, and the total number of grazing businesses (farmers) during that period was 88,945 (Commonwealth of Australia 2011). According to National Scale Land Use version 5, in 2010–2011, the total grazing land area was 415.1 × 10^4 km^2, including

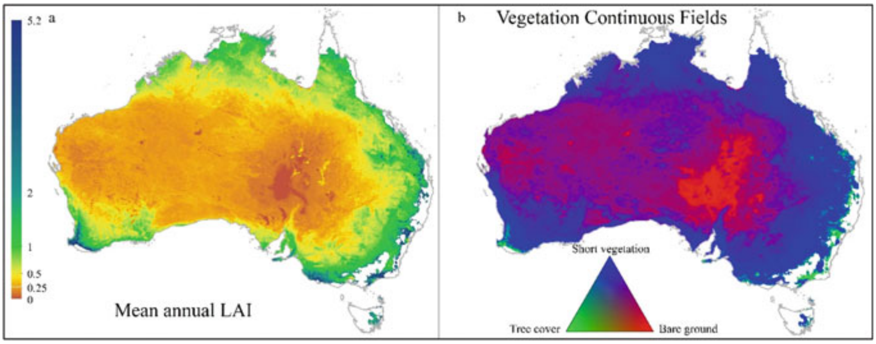

Fig. 11.6 Vegetation cover in drylands in Australia. **a** Map of the mean annual leaf area index (LAI) during 2001–2018; **b** mean vegetation continuous fields in 1982–2016. The LAI data are from the Global Land Surface Satellite (GLASS) product suite (Liang et al. 2021; Tang et al. 2013; Xiao et al. 2014; Xu et al. 2018). Vegetation continuous field (VCF) datasets are from the National Aeronautics and Space Administration (NASA) Making Earth System Data Records for Use in Research Environments Project (Hansen and Song 2018; Song et al. 2018)

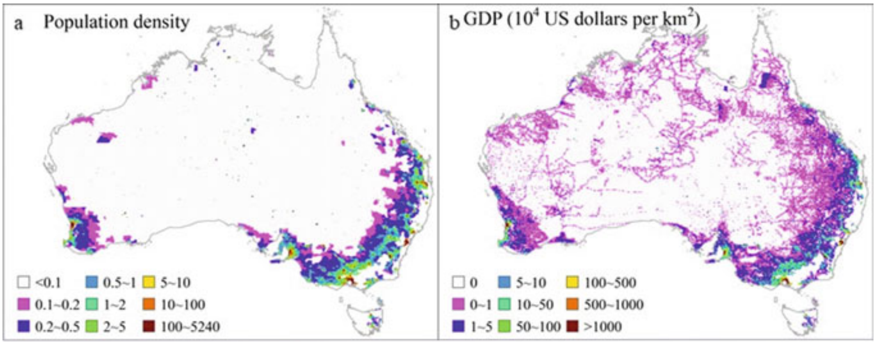

Fig. 11.7 Socioeconomic factors in the drylands of Australia: **a** population and **b** GDP. Gridded population maps are from the Gridded Population of the World (GPW) v4 from the Socioeconomic Data and Applications Center (SEDAC). The Gridded GDP is from Kummu et al. (2018)

344.9×10^4 km^2 of grazing native vegetation land and 70.2×10^4 km^2 grazing-modified pastures, which accounted for 44.8 and 9.1% of terrestrial Australia (Commonwealth of Australia 2016). Grazing native vegetation land is mainly located in the arid region (71.5%), and grazing-modified pastures are mostly distributed in the semiarid region (66.5%) (Fig. 11.21a, b). Thus, the fraction of grazing area to the total area in the semiarid region is the highest, reaching 66.4%, followed by 51.8%, 42.7% and 26.6% for the arid, dry subhumid, and humid regions, respectively.

The indigenous lands, the next largest land use in Australia, is albeit diverse and with its own economy. Land use for mining and tourism is also important as a major part of economic activity besides the extensive grazing in the indigenous lands.

11.3 Changes in Ecosystem Structures

The sparse tree areas have decreased, while the areas with open and closed trees have increased, indicating that forests in the drylands of Australia have probably become denser since 2002. However, for shrublands, the closed areas have declined, while the areas with open shrublands and sparse or scattered shrubs and grasses have increased, implying that the shrubland canopy density in the drylands of Australia may have decreased. Grasslands have expanded, especially in arid areas, which may have also resulted from the degradation of closed shrublands. In addition, some rainfed crop areas in semiarid regions may have been replaced by rainfed pastures (Fig. 11.8).

Leaf area index (LAI) increased significantly in the relatively humid coastal forest regions but decreased in many arid areas that were dominated by shrubs or grasses during 2001–2018 (Fig. 11.9a). This finding is consistent with the land use cover changes in Australia, namely, expanded closed forests and degraded shrublands, and can be attributed to the 'drier drylands, wetter wet areas' climate change pattern. Accordingly, significant LAI gains ($p < 0.01$) occurred in the dry subhumid regions, whereas both the arid and semiarid regions experienced no significant LAI changes in recent decades (Fig. 11.9b).

During a longer period of 35 years (i.e., 1982–2016), the Advanced Very High Resolution Radiometer (AVHRR)-based vegetation continuous field (VCF) maps showed an increase in bare ground in most desert areas in Australia. On the north-eastern coasts, tree cover declined, but short vegetation cover increased, whereas in southern Australia, tree cover greatly expanded (Fig. 11.10a). In the dry subhumid regions, bare ground significantly transitioned into short vegetation after 2006, but in drier areas (arid and semiarid regions), VCF changes were not significant throughout the study period (Fig. 11.10b–d).

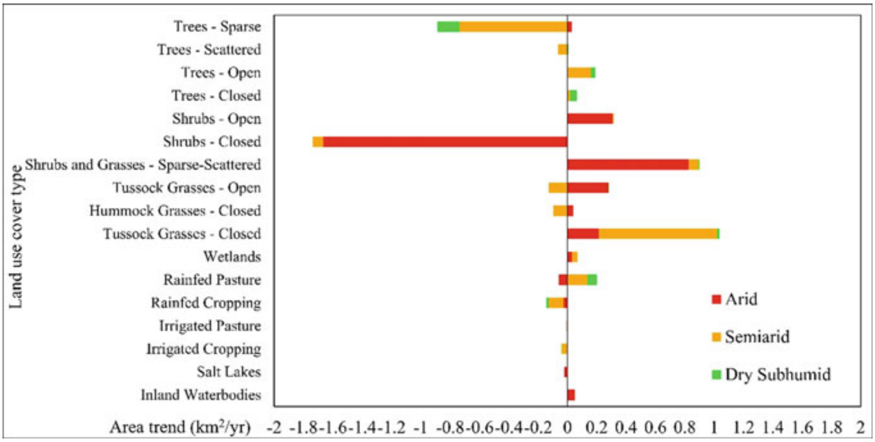

Fig. 11.8 Interannual trends of different land use types in Australia's drylands during 2002–2015

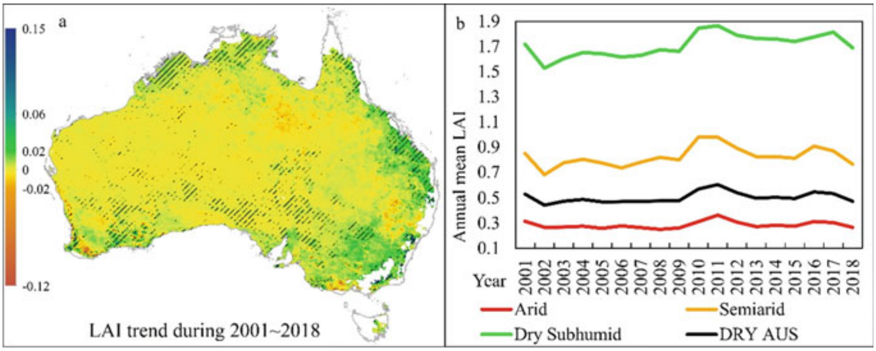

Fig. 11.9 LAI changes in the drylands of Australia. **a** Spatial pattern of the interannual LAI trend during 2001–2018; shaded areas represent statistical significance at the 95% confidence level; **b** interannual variation in the LAI within different regions in the drylands of Australia

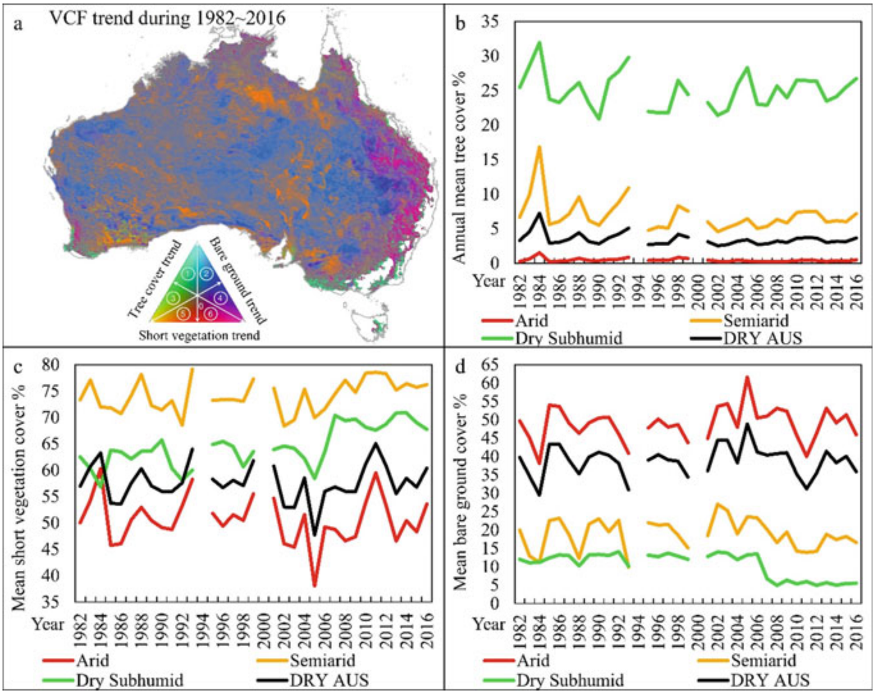

Fig. 11.10 VCF changes in the drylands of Australia during 1982–2016. **a** Map showing the interannual trends of tree cover, short vegetation cover, and bare ground cover; **b–d** interannual variations in **b** tree cover, **c** short vegetation cover, and **d** bare ground cover in different dryland regions in Australia

The drylands of Australia are dominated by bare soil (BS) and nonphotosynthetic vegetation (NPV), especially in the arid interior of the continent (Fig. 11.11a). During 2001–2018, in arid and semiarid regions, NPV, or both photosynthetic vegetation (PV) and NPV generally decreased, while the bare soil area expanded, indicating gradual vegetation degradation that was probably driven by the drier climate in the typical drylands of Australia. Conversely, on the eastern coasts and in southern Australia, where vegetation cover is denser owing to the relatively humid climates, PV, or both PV and NPV, increased, which agreed with the recent precipitation gains in most of those relatively wet areas (Fig. 11.11b). The interannual variations in PV, NPV, and BS are shown in Fig. 11.11c–e. In 2011, due to high precipitation, PV was the highest during the study period, while both NPV and BS were quite low in the drylands of Australia.

Eucalypts—often called gum trees—are iconic Australian flora. According to Australia's State of the Forests Report 2013, ninety-two million hectares of the eucalypt forest type occur in Australia and form three-quarters of the total native forest area. The eucalypt native forest is distributed in all states and territories and across all except the most arid deserts (Fig. 11.12). Eucalypt plants provide a wide variety of resources that include food, shelter, refuge, and breeding sites for animals (Bennett

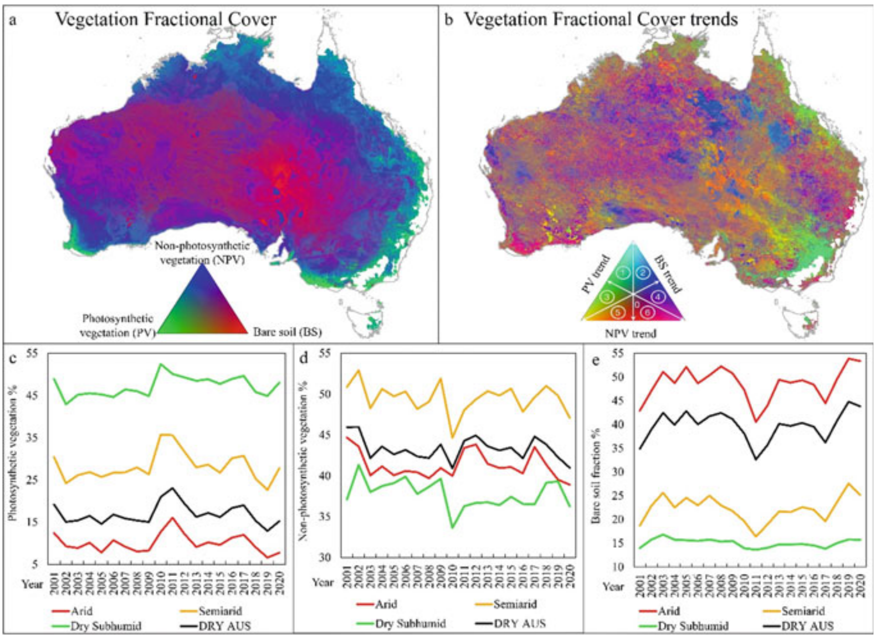

Fig. 11.11 Vegetation fractional cover and its changes in the drylands of Australia. **a** Map showing the spatial distributions of photosynthetic vegetation (PV), nonphotosynthetic vegetation (NPV) and bare soil (BS); **b** spatial pattern of the interannual trends of vegetation fraction cover; **c**–**e** interannual variations in **c** PV, **d** NPV, and **e** BS in regions with different aridity levels in the drylands of Australia

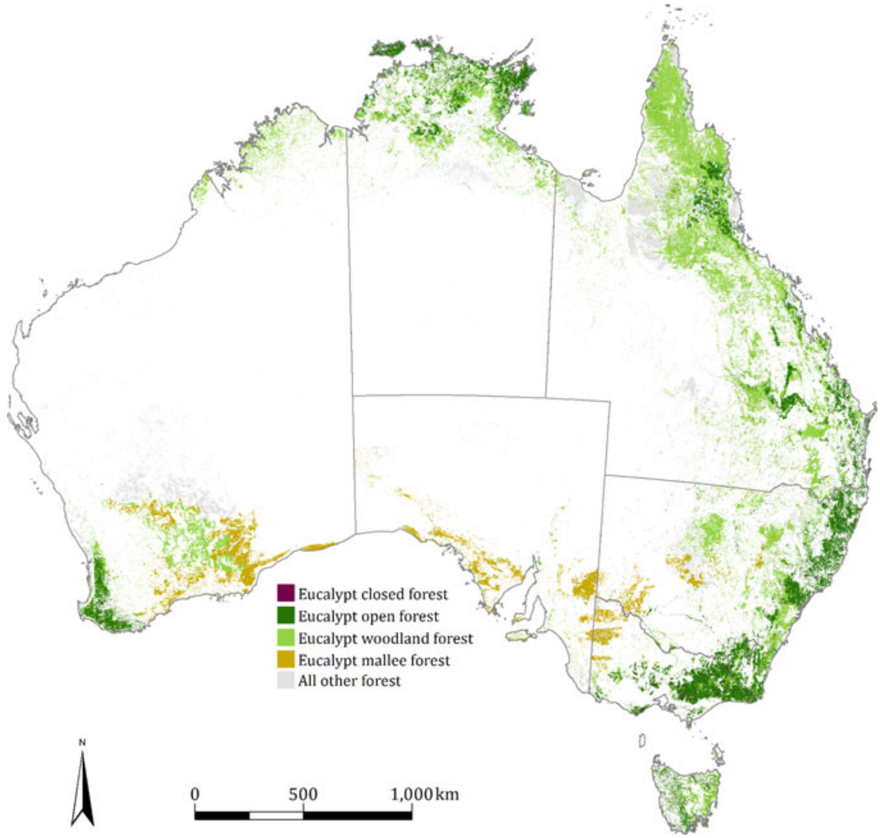

Fig. 11.12 Distribution of native eucalypt forests (Commonwealth of Australia 2016)

2016), as well as medicinal materials and wood for humans. Moreover, native eucalypt forests are important for the conservation of Australia's rich biodiversity because they support many forest-dwelling or forest-dependent species of flora and fauna.

11.4 Changes in Ecosystem Services

Ecosystem services in this chapter include ecosystem carbon sequestration, water yield, soil conservation, and crop production. In particular, ecosystem carbon sequestration is indicated by net primary production (NPP). Water yield is the difference between annual precipitation and ecosystem evapotranspiration. The average annual per-area NPP in the humid region is the largest at 950.1 gC/m^2, followed by that in the dry subhumid area (723.3 gC/m^2) and the semiarid region (405.3 gC/m^2), and finally, it is the lowest in the arid region (142.0 gC/m^2, Fig. 11.13a, b). The total annual NPP

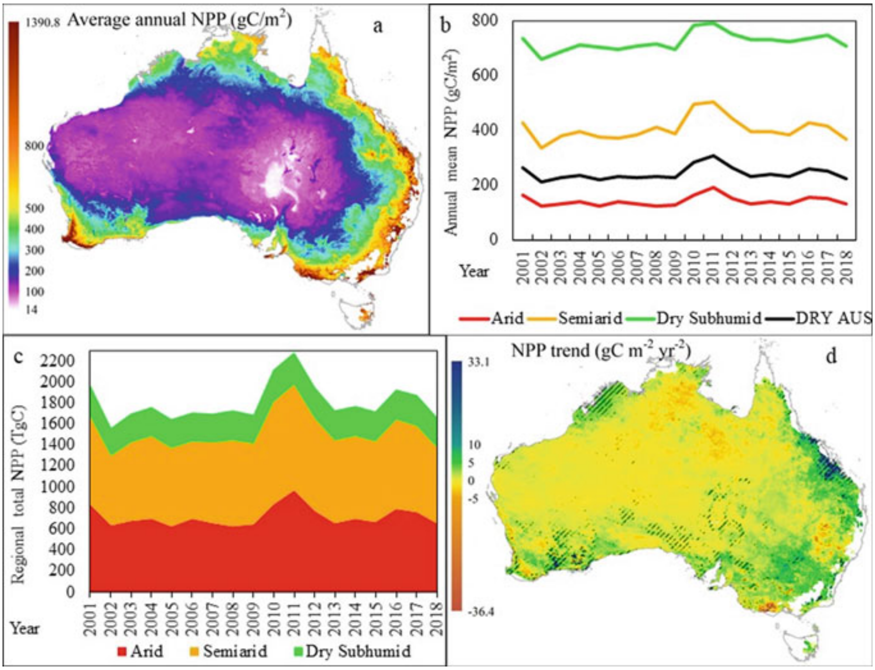

Fig. 11.13 Spatiotemporal pattern of NPP in the drylands of Australia: **a** spatial map of average annual NPP during 2001–2018; **b** mean per-area NPP changes in different regions in Australia distinguished by aridity; **c** interannual variation in total NPP in regions with different aridity levels in the drylands of Australia; and **d** map of the interannual trend of NPP during 2001–2018. Shaded areas indicate that the NPP trends are significant at the 95% confidence level. NPP data comes from the GLASS product suite (Li et al. 2013; Yuan et al. 2007, 2010, 2014)

is, however, the highest in the semiarid region, at 808.7 TgC, followed by the arid region (714.3 TgC). Due to their small areas in Australia, dry subhumid and humid regions produced NPP values of only 279.1 TgC/yr and 367.8 TgC/yr, respectively, during the period of 2001–2018. In the drylands of Australia, no region experienced significant NPP gains over the whole period, but the significance ($p = 0.07$) and rate ($k = 2.66 \pm 2.86$ TgC/yr^2) of NPP increases in the dry subhumid region were larger than those in the semiarid region ($p = 0.57$; $k = 0.49 \pm 1.80$ TgC/yr^2) and arid region ($p = 0.38$; $k = 1.76 \pm 4.11$ TgC/yr^2) in Australia. Figure 11.13c shows that the total NPP in the drylands of Australia reached its highest in 2011, when precipitation was the largest during the study period and was significantly lower in dry years (e.g., 2002 and 2018). As shown in Fig. 11.13d, recent NPP increases are concentrated in the relatively humid coastal areas of Australia, whereas in the arid interior part, the NPP declines in most places, which can also be explained by the climatic temporal pattern.

The annual water yield service was the highest in the tropical region located in northern Australia, mainly due to the high precipitation in that region (Fig. 11.14a).

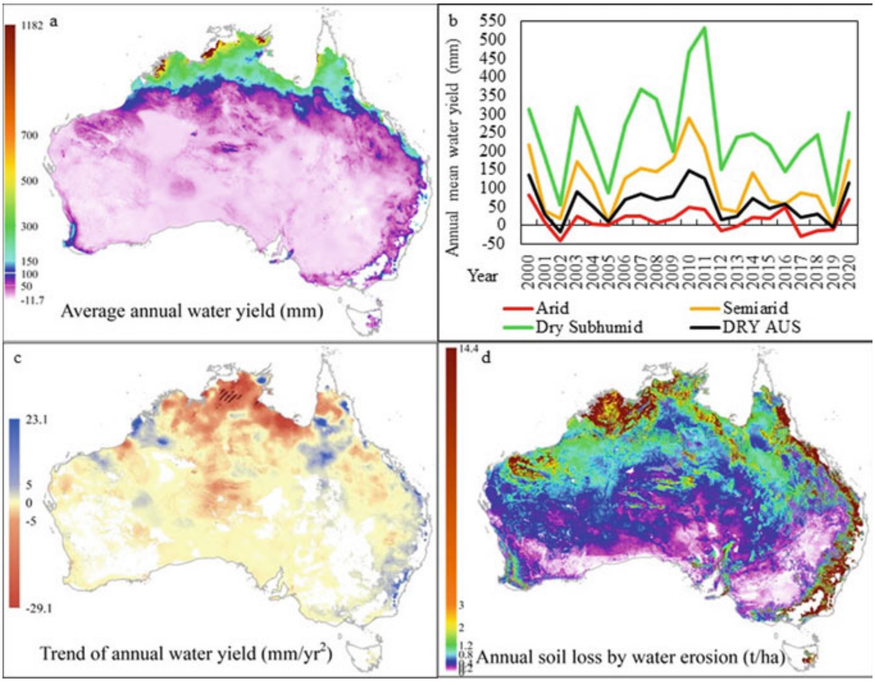

Fig. 11.14 Water yield and water erosion in the drylands of Australia. **a** Spatial pattern of average annual water yield; **b** interannual variation in water yield in regions with different aridity levels in the drylands of Australia; **c** spatial map of the annual water yield trend from 2000 to 2020; and **d** map of annual soil loss induced by water erosion in the drylands of Australia

Evapotranspiration data in Australia are provided by the Australian Landscape Water Balance website (www.bom.gov.au/water/landscape). The average annual water yields in the arid, semiarid, and dry subhumid regions of Australia during 2000–2020 were 15.47 mm, 112.4 mm, and 245.5 mm, while the coefficients of variation (i.e., the ratios of the standard deviation to the average value) in these three regions were 1.93, 0.68, and 0.48, respectively. Accordingly, the water yield in Australia decreased as the aridity increased, but the interannual variability was stronger in more arid areas. In all drylands in Australia, the mean and standard deviation values of water yield were 59.9 and 45.4 mm, respectively. During 2000–2020, the drylands' water yield was the highest in 2000, 2011, and 2020, which were the years with much rainfall, while the values were the lowest and even negative in the dry years of 2002 and 2019 (Fig. 11.14b). Over the whole study period, water yield declined significantly in northern Australia due to the reduced rainfall and decreased slightly and nonsignificantly in most of the interior dry areas of Australia (Fig. 11.14c).

It may be worth noting that the pulse in 2011 related to flooding that stored so much water that it was detectable in sea level globally, indicating the global significance of dryland in Australia. Australia contributed uniquely and substantially to the intensity

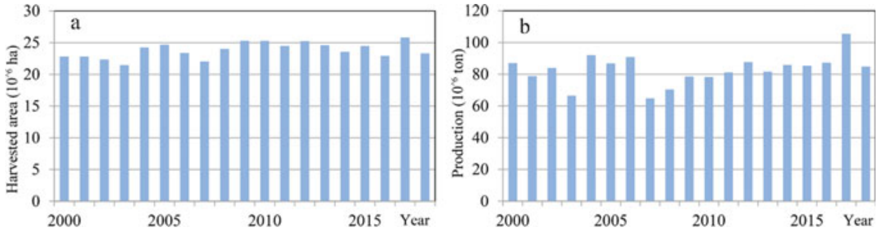

Fig. 11.15 Harvested area and production in the drylands of Australia: time series of harvested area (**a**) and production (**b**) during 2000–2018

and persistence of the global land hydrologic mass increase during the 2011 sea level drop. The persistence of Australia's mass anomaly was attributed to the continent's unique surface hydrology, which includes expansive arheic and endorheic basins that impede runoff to ocean (Fasullo et al. 2013).

By using the Australian soil loss map that shows water erosion calculated through the revised Universal Soil Loss equation (RUSLE) model (Teng et al. 2016; Viscarra Rossel et al. 2016), it is found that the soil erosion is high on the eastern coast and in the northwestern part of Australia and is the lowest in southern Australia (Fig. 11.14d). Moreover, the harvested area and production of primary crops increased at rates of 0.10×10^6 ha/yr and 0.54×10^6 t/yr during 2000–2018, according to food statistics from the Food and Agriculture Organization (FAO) of the United Nations (Fig. 11.15).

11.5 Driving Forces of Change in Dryland Social-Ecological Systems in Australia

11.5.1 Climate Trend

The temperature rose in the drylands of Australia after 1970, but precipitation showed a limited trend during 1900–2019. According to a 20-year period moving window, the variance of temperature followed a cycle of 20–25 years approximately before 1990 in the drylands of Australia, decreasing during 1919–1925, 1948–1965 and 1970–1990 (note: these are the middle times of 20-year periods), while increasing during 1925–1937, 1937–1948 and 1965–1970. However, after 1990, these cycles disappeared and were replaced by a very strong and consistent warming trend. In recent decades, i.e., 2000–2019, the arid region in Australia experienced a warming of 0.053 °C/yr, and for all drylands in Australia, the trend was 0.048 °C/yr, which were the fastest rates since 1910.

The 20-year period precipitation trends increased abruptly after 1940 and slightly declined after 1988. Recently (i.e., 2000–2019), precipitation declined at rates of

−4.9 mm/yr and −5.2 mm/yr in arid regions and in all drylands of Australia, respectively. In Australia, precipitation and temperature are highly correlated, their coupling becoming stronger. Accordingly, the accelerated rise of temperature and reduced precipitation in the twenty-first century suggest that Australia is becoming drier. In addition, the solar radiation in the drylands of Australia has increased somewhat during the twenty-first century (Fig. 11.16a–f).

In Western Australia, the temperature rose significantly during 2000–2019, while in northern Australia, both solar radiation and temperature increased, and precipitation significantly declined in some areas. Moreover, southeastern Australia has recently experienced climate warming. Solar radiation increased significantly in northern Australia and in some parts of the central desert areas (Fig. 11.17a–c). By combining these three basic meteorological variables with the aridity index of the SPEI, most parts of dryland Australia have become drier, especially the northern areas where the drying trends were significant during 2000–2019 (Fig. 11.17d). Compared to the significant coupling between annual mean temperature and precipitation over drylands in Australia, the central-southern parts exhibiting negative correlations between annual temperature and precipitation were fewer in the last 20 years (Fig. 11.17e, f).

Precipitation in drylands in Australia had a significant impact on that year's LAI. Air temperature was negatively correlated with the annual mean LAI/NPP in the drylands of Australia, but the significance of the relationships was lower than that between the LAI and precipitation in all regions (Fig. 11.18a, b). In addition, wetting in the drylands of Australia promoted the LAI in the next year, suggesting a 'lag effect'. For tree cover (TC), only in the arid region did the corresponding year's precipitation exhibit a significant positive effect. However, precipitation rose very significantly ($p < 0.01$), promoting short vegetation (shrubs and grasses) cover in both arid and semiarid regions, indicating that short vegetation in the drylands of Australia is more easily influenced by precipitation changes than are trees. Owing to this strong relationship, bare ground cover in arid and semiarid areas was negatively correlated with precipitation ($p < 0.01$). Precipitation in the previous year also showed some effects, though not significant (Fig. 11.18c). Increases in precipitation can significantly promote photosynthetically active vegetation (PV) in all regions of Australia, especially in more arid areas. However, precipitation in the current year had a significant negative impact on senescent or dead vegetation (NPV) cover in semiarid and dry subhumid regions. Specifically, in arid regions and in all drylands of Australia, although the precipitation in a given year showed no significant effects, higher precipitation in the previous year could significantly ($p < 0.01$) promote the NPV cover, which can be explained by the fact that the drylands of Australia are largely covered by annual herbaceous plants. Finally, both the precipitation in the current year and that in the previous year had a strong negative effect on the bare soil cover in all regions of Australian drylands, especially in arid and semiarid places (Fig. 11.18d).

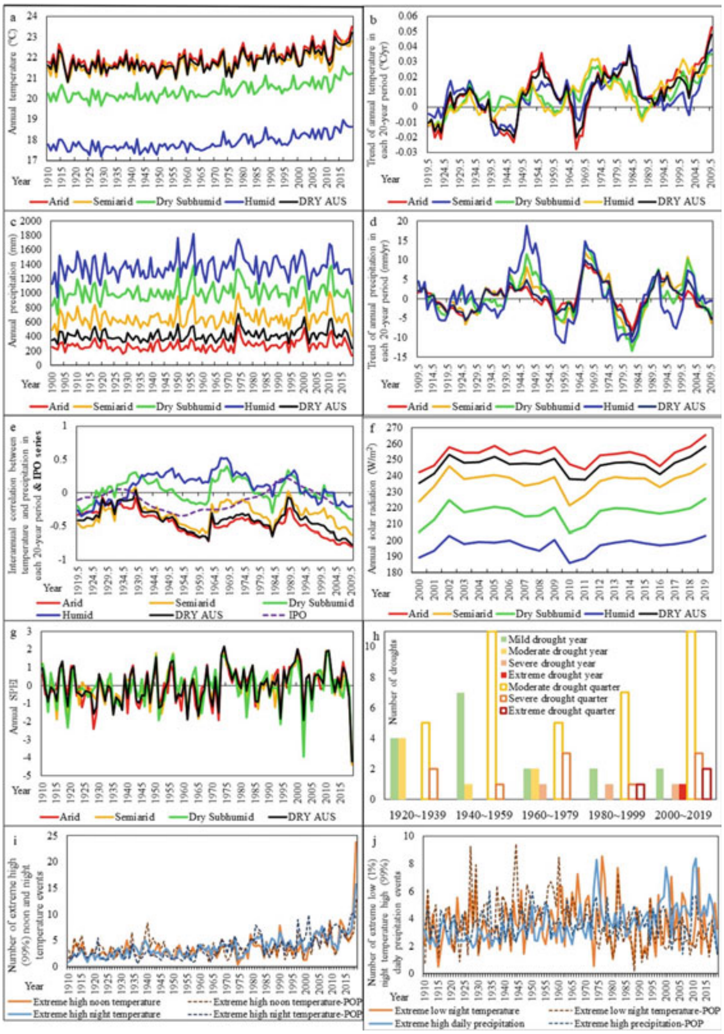

Fig. 11.16 Change in the Australian climate. **a** Interannual variation in mean temperature in different regions; **b** dynamics of the trend of annual temperature in each 20-year moving window in different regions. The x-axis labels are the middle years of the 20-year periods; **c** interannual variation in precipitation in different regions; **d** dynamics of the trend of precipitation in each 20-year moving window; **e** dynamics of the interannual correlation between temperature and precipitation in every 20-year period (note that the interdecadal Pacific oscillation (IPO) curve represents the average filtered interdecadal Pacific oscillation index in each period; see Sect. 11.4); **f** interannual variation in mean solar radiation in different regions; **g** interannual variation in the standardized precipitation evapotranspiration index (SPEI) in arid Australia; **h** number of drought years and drought quarters with different degrees during each 20-year period from 1920 to 2019; **i** average number of extreme hot noontimes and hot nights from 1910 to 2019 in different regions; note that the '-POP' means the values are averaged and weighted by population; and **j** average number of extreme cold nights and extreme high precipitation days from 1910 to 2019 in different regions of Australia

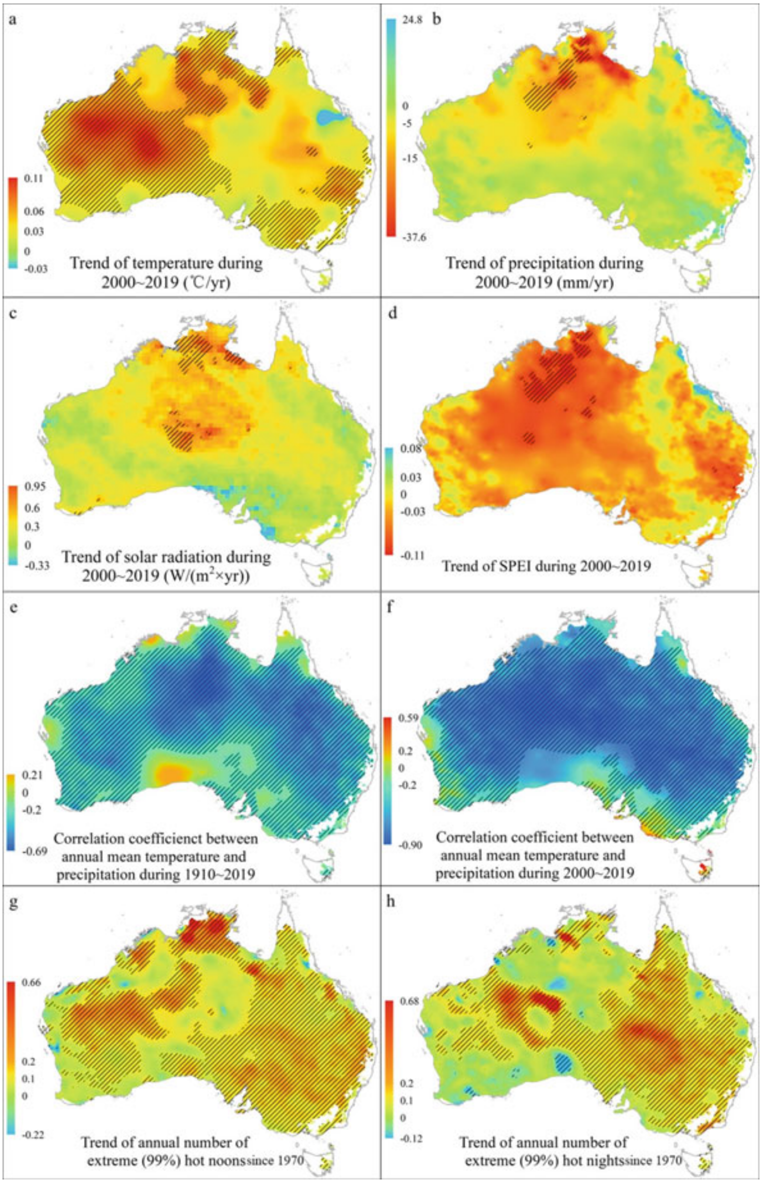

Fig. 11.17 Spatial patterns of the climate changes in the drylands of Australia. **a** Map of temperature trends; **b** map of precipitation trends; **c** map of solar radiation trends; **d** map of standardized precipitation evapotranspiration index (SPEI) trends; **e**, **f** spatial patterns of the correlation between annual mean temperature and precipitation during **e** 1910–2019 and **f** 2000–2019; **g**, **h** trends of annual number of extreme (99%) **g** hot noontimes; and **h** hot nights from 1970 to 2019. The shaded areas indicate significant trends or significant correlations at the 95% confidence level

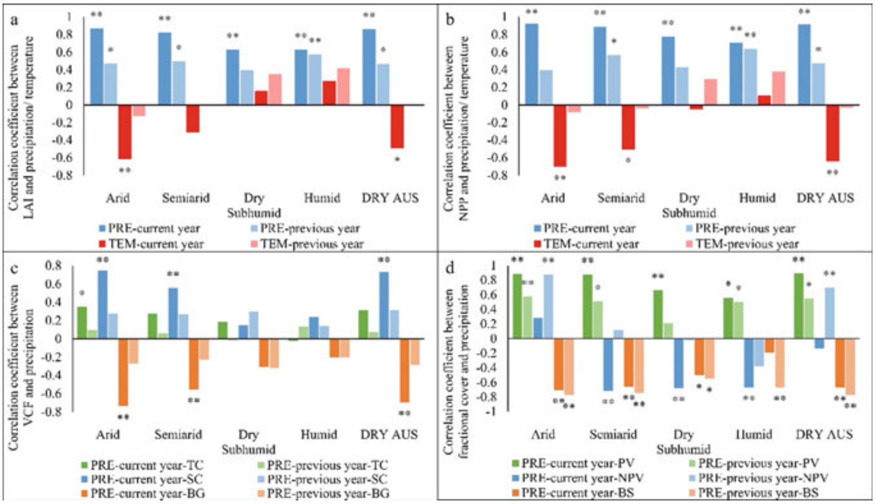

Fig. 11.18 Climatic impacts on the ecosystem in Australia: **a** interannual correlation coefficients between LAI and precipitation or air temperature in different regions distinguished by the aridity level; **b** correlation between NPP and precipitation or temperature; **c** correlation between VCFs (tree cover, short vegetation cover, and bare ground indicated by TC, SC, and BG, respectively) and precipitation in the current year and that in the previous year; and **d** temporal correlation between fractional cover (PV, NPV, and BS) and precipitation

11.5.2 Changes in Extreme Climate Events

Not only extreme droughts but also the numbers of extreme (99%) hot noontimes and hot nights have significantly increased in the drylands of Australia since the late 1970s (Fig. 11.16i). Moreover, the number of extreme cold nights with the lowest 1% of all night temperatures did not change significantly throughout the 110 years, but in highly populated areas, there were fewer extreme cold nights after 1960 (Fig. 11.16j). The number of extreme high daily precipitation events in all drylands of Australia also increased rapidly, especially after the 1970s, indicating increases in both droughts and floods. However, in urban areas with high populations, extreme high precipitation events decreased after the 1970s. Eastern Australia experienced significant increases in annual extreme (99%) hot noontimes and nights over the past 50 years (1970–2019, Fig. 11.17g, h).

11.5.3 Dynamics of Fire Disturbance

Fire is a frequent and prevalent and usually the most important natural disturbance to ecosystems in the drylands of Australia (Bradstock et al. 2002). As the consequences of very different seasonalities in north and south Australia (shown in Fig. 11.2b),

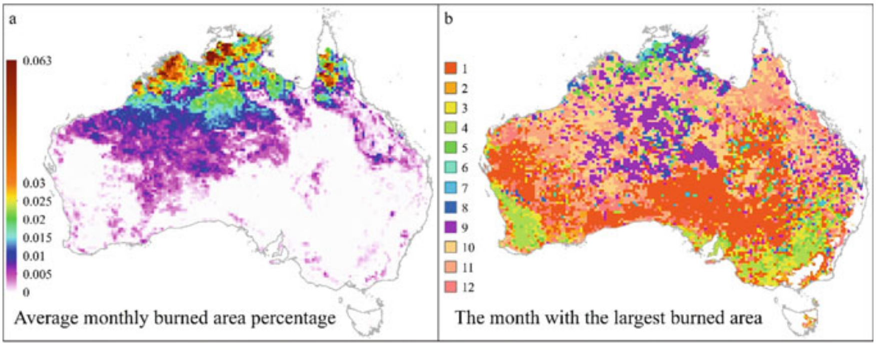

Fig. 11.19 Fire distribution in the drylands of Australia. **a** Spatial pattern of the average monthly burned area percentage; **b** spatial pattern of the month with the largest burned area in a year averaged during 1996–2016. The monthly burned area is averaged during 1996–2016 using the Global Fire Emissions Database (GFED) version 4.1 (Randerson et al. 2018) and is further converted into the burned area fraction (i.e., the ratio of burned area to the total grid cell area)

fire in northern tropical grasslands is certainly the most extensive (Fig. 11.19a), but the nature of fire is totally different in the north (annual grass fires) from the south (dominated by longer return time forest fires). Fires in the drylands of Australia have clear seasonal variation patterns. In 27% of the total dryland area in Australia, the largest fire events occur in January, when air temperature reaches the maximum in a year, while for 81% of the total dryland area, the largest area of burning occurs during spring to summer, i.e., September to January of the following year (Fig. 11.19b). The burned area percentage is the highest in the dry subhumid region of Australia, with an average monthly value of 1.21%, followed by the semiarid region (0.84%), the humid region (0.60%), and finally, the arid region (0.32%), indicating that burning in Australia relies on vegetation, the raw material for combustion, and the relatively arid weather, which are environmental requirements for fires.

Burned area in the drylands of Australia showed a nonsignificant decrease during 1996–2016, yet large burned areas were observed in 2000–2002 and 2011–2012. It was also reported that between September 2019 and early January 2020, when droughts occurred, Australian mega forest fires led to unprecedented burned areas (Boer et al. 2020; Ward et al. 2020). The temporal variation in fire in the drylands of Australia was attributed to climate change, especially moisture conditions (Clarke et al. 2019; Kelley et al. 2019; Phillips and Nogrady 2020) and human controls (Liu et al. 2021). Precipitation can significantly promote burned areas in the following year within arid and semiarid regions, probably by increasing the fuel for combustion, whereas precipitation significantly reduces fires in humid regions and nonsignificantly inhibits fires in dry subhumid regions within the same year. In contrast, a nonsignificant positive impact of precipitation on fire occurred in arid regions in the same year (Fig. 11.20).

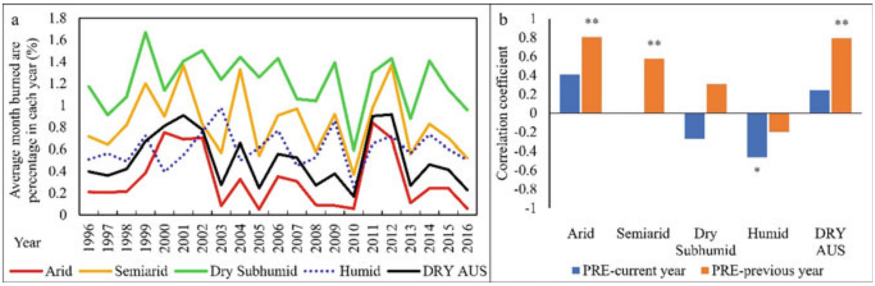

Fig. 11.20 Fire dynamics in the drylands of Australia. **a** Interannual variation in average monthly burned areas in regions with different aridity levels in Australia; **b** correlation coefficients between burned areas and the precipitation in the current year and that in the previous year in different regions in Australia

11.5.4 Ecosystem Management

The Australian government has invested abundant funds in pasture management (Barson et al. 2011) with advanced grazing and fire management systems, water resource management, systematic mechanisms of carbon farming and agroecological approaches to simultaneously support agricultural biodiversity and promote sustainable livelihoods. Moreover, these investments have achieved successes in ecosystem management methods of community engagement.

Different grazing modes were used to improve productivity, maintain desirable pasture species, and reduce land degradation. Native pastures can be managed through a number of grazing strategies, including continuous grazing, rotational grazing, cell grazing, time control grazing, and spell grazing (O'Reagain et al. 2014). Continuous grazing requires minimal labour and can deliver good production but is often accompanied by overgrazing, with livestock habitually revisiting preferred areas. Rotations are often organized around the plant growth cycles with the aim to optimize pasture utilization, which prevents uneven grazing and allows perennials to replenish their root reserves and better withstand dry periods, thereby benefiting both soil structure and land conditions (McDonald et al. 2019). Cell grazing and time-controlled grazing are similar to rotational grazing but are more intensive and involve more paddocks or 'cells' (McCosker 2000). Spell grazing involves locking up pastures at critical times in their growth cycle to allow plants to replenish root reserves and set seeds. This reduces the risk of overgrazing and encourages pasture plant recruitment through seed setting. Additionally, some feed additives are used to inhibit the microorganisms that produce methane in the rumen and subsequently reduce methane emissions. Overall, a successful grazing system should manage pasture utilization effectively (carrying capacity and timing of spelling), reduce uneven grazing, and match the stocking rate to the diet quality required by animal production targets (Hunt 2008).

To prevent soil degradation, soil and water erosion and plant/animal biodiversity losses caused by overgrazing (Hansen et al. 2019), the Australian government invested more than US$ 442 million to improve soil and biodiversity management practices on farms by November 2011 (Barson et al. 2011) and has supported the PROGRAZE™ educational systems (Martindale and Marriott 2004). These education and management practices (e.g., controlling the stocking rate, rotational grazing, preventing invasive plants and controlling pests (Hacker et al. 2019)) are intended to prevent soil acidification, maintain soil nutrients and protect ground vegetation cover, etc. (Waters et al. 2017). At least 50–70% ground cover, depending on the location, is recommended (Barson et al. 2011). Under grazing controls, the total grazing area in Australia has declined over the past 10 years, which was approximately 341×10^4 km^2 in 2016–2017 but only 325×10^4 km^2 in 2020 according to the Australian Bureau of Statistics. According to Fig. 11.21c, the head of livestock, especially sheep, has significantly decreased since 1991, indicating strong management of grassland grazing in the drylands of Australia.

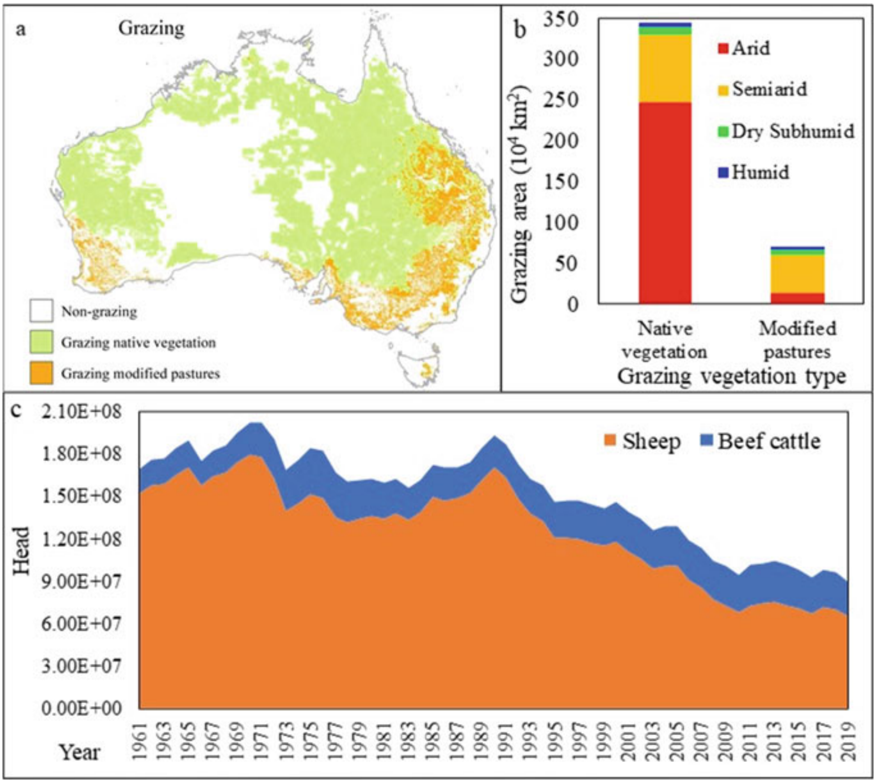

Fig. 11.21 Grazing extent in the drylands of Australia: **a** spatial distribution of grazed native vegetation and grazing-modified pastures; **b** areas of grazed native vegetation and grazing-modified pastures in different regions of Australia; and **c** heads of beef cattle and sheep in Australia from 1961 to 2019, according to official statistics

Grazing and wildfires play important roles in tropical savanna management and industries in Australia, yet they both contribute to greenhouse gases in Australia. Fires consume vegetation and, in doing so, produce CO_2 and other greenhouse gases. Greenhouse gases emitted from savanna fires average 3% of Australia's emissions. Due to the large area percentage of rangelands in Australia (~80%), direct livestock emissions account for approximately 70% of greenhouse gas emissions in the agricultural sector and 11% of total national greenhouse gas emissions. These values make Australia's livestock the third largest source of greenhouse gas emissions after the energy and transport sectors. Livestock are the dominant sources of methane (CH_4) and nitrous oxide (N_2O), accounting for 56% and 73% of Australia's emissions, respectively.

Australia government has implemented climate change mitigation policies to reduce emission caused by the changing agricultural practices. Carbon farming is one of the most important policies contributing to the climate change mitigation, which aims to increase the amount of carbon stored in the soil and vegetation (sequestration) and to reduce greenhouse gas emissions from livestock, soil, or vegetation by changing agricultural practices or land use (Evans 2018). Carbon farming is important in Australia because agriculture accounts for 13.5% of Australia's greenhouse gas emissions. To help reduce these emissions, the Carbon Farming Initiative in Australia was implemented in December 2011 to encourage land projects. Carbon farming potentially offers landholders financial incentives to reduce carbon pollution, which often generates economic and environmental ancillary (co)benefits (Tang 2016; Tang et al. 2018), such as improved soil quality, erosion prevention, better protection for stock, improved livestock production, native habitat creation for threatened species (Dumbrell et al. 2016; Kragt et al. 2016). Moreover, there still exist some barriers for carbon farming due to the lack of information and the government policy uncertainties (Kragt et al. 2017).

Prospective projects in Australia include savanna fire management and rangeland management (Kelly and Brotons 2017; Rolfe et al. 2021). The timing and intensity of fire, and consequently its various ecological impacts, can be significantly influenced by fire management activities. Effective fire management can reduce greenhouse gas emissions, protect fodder and infrastructure, and potentially attract payment for stewardship activity (Steffensen 2020). For thousands of years, indigenous Australians expertly managed the tropical savannas of northern Australia. They use fire to shape the landscape and to achieve social, economic, and spiritual well-being for the country. The fire management practices help protect the biodiversity, balance trees and grasses, mitigate emission, reduce infrastructure damage, and improve livelihoods (Bradstock 2010; Kelly and Brotons 2017; McKemey et al. 2022; Nikolakis and Roberts 2020; Steffensen 2020).

The Murray-Darling Basin (MDB) is an example of a complex river system undergoing substantial water reform to balance the needs of human and the environment. The basin extends across 4 states in south-eastern Australia, occupying 14% of Australia's total surface area. Much of the basin is semiarid and contains 50% of Australia's irrigated agriculture. Multiple efforts, such as the 2007 Water Act and 2012 Murray–Darling Basin Plan (MDBP), were issued to sustainably optimize social and environmental outcomes in relation to water use in the basin (Bischoff-Mattson and Lynch 2017). The basin plan includes five parts (i.e., balancing/sharing, monitoring, review, revision, and adaptation) and guides all stakeholders to use water in a sustainable way (Productivity-Commission 2018). Most importantly, the MDBP manages the basin as one system, enabling the river systems to adapt to climate changes and continue to support all water stakeholders in the long term. Although crisis-driven management to some extent prevents ‘economic and environmental decline’, the Plan makes no direct allowance for climate change, setting the scene for a future crisis that will trigger further reform (Colloff and Pittock 2022; Pittock 2019).

Australia claims to be pursuing a ‘green growth’ model in response to the global economic crisis and climate change. An agroecological approach supports agricultural biodiversity while promoting sustainable livelihoods (Lanka et al. 2017). Indigenous people living in Australia’s tropical savanna landscapes are increasingly searching for income opportunities from environmental services or ecosystem services as an avenue for economic development and improvement of socioeconomic conditions (Greiner 2010). The sustainable livelihood in Australia can better cope with and recover from stresses and shocks and promote its capabilities and assets both now and in the future (Davies et al. 2008; Moran et al. 2018).

11.5.5 Social and Economic Development

Population ageing is probably the most important social problem for Australia in recent decades, since the proportion of the elderly population nearly doubled from 8.24% in 1970 to 16.21% in 2020 (Fig. 11.22a), and was still increasing at a high rate (0.29%/yr during 2010–2020), bringing about challenges to social and economic development (Kendig et al. 2016; O’Loughlin et al. 2017), 2.23 million women (17.2% of all women) for the ages over 65 years, more than men in the same age (1.96 million, which is 15.4% of all men), were now living in Australia. Spatially, the percentage of elderly people was generally higher in southern Australia than in northern Australia and is the highest on the southeast and southwest coasts (Fig. 11.22b). The elderly proportion was similar among regions with different aridity levels, but the proportions of children and juveniles in arid regions were lower, resulting in fewer youth labourers, especially 20–29-year-old people, than in other regions. Rapid growth of the Indigenous Population is expected, with population momentum, identification change, and mixed partnering and childbearing shown to contribute more to growth than above-replacement fertility and increasing life expectancy. Since 1971 the indigenous population of Australia has trebled. From 1991 to 1996 numbers grew by 33%. The future growth of Australia’s Indigenous Population is thus intimately connected to its interaction with the Non-Indigenous Population (Wilson 2016).

In Australia, the population is now generally moving out of Western Australia (WA) and the Northern Territory (NT) into eastern and southeastern Australia,

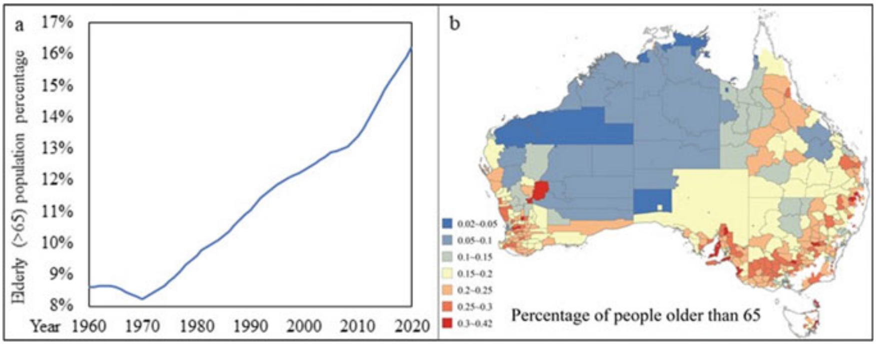

Fig. 11.22 Change in the population structure in the drylands of Australia: **a** temporal change in the proportion of the elderly (>65 years old) population in Australia; and **b** the percentage of people older than 65 in all local government areas in Australia in gridded 2020 population maps are from the Gridded Population of the World (GPW) v4 dataset from the Socioeconomic Data and Applications Center (SEDAC). The land area per pixel values were resampled from ~1 km resolution to 0.05°

including New South Wales (NSW), Victoria (Vic.), and Queensland (Qld). Therefore, the general direction of internal migration in Australia is from relatively arid regions to more humid regions.

The GDP in the drylands of Australia has more than doubled since 1990, increasing from US$ 3.3 × 10^{11} to US$ 7.0 × 10^{11} in 2015 (Fig. 11.23). The GDP increase rates were somewhat higher in the semiarid and dry subhumid regions (3.3% and 3.2%/yr) than in the arid regions (2.5%/yr).

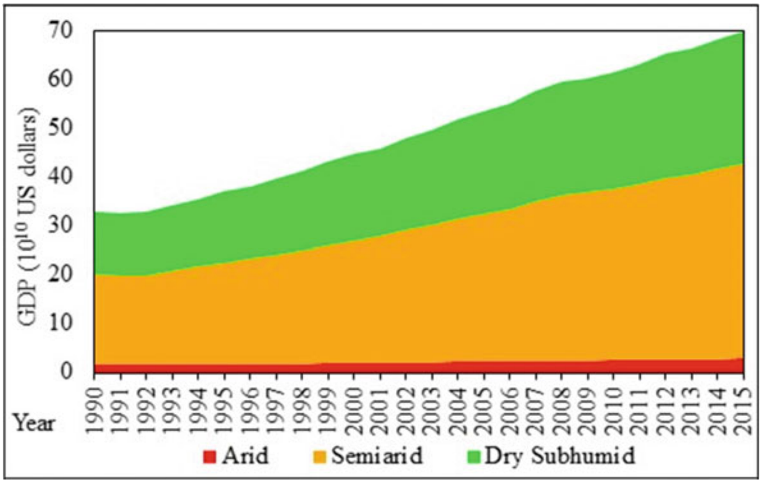

Fig. 11.23 GDP changes from 1990 to 2015 in regions with different aridity levels in the drylands of Australia. The gridded GDP is from Kummu et al. (2018)

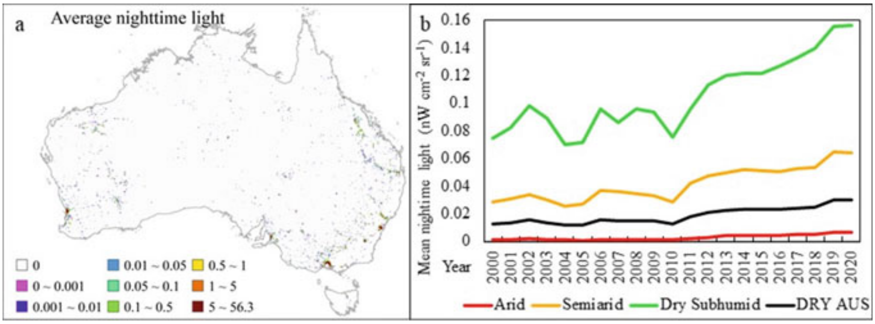

Fig. 11.24 Nighttime lights (NTL) in the drylands of Australia: **a** spatial pattern of NTL averaged during 2000–2020; and **b** mean NTL changes from 2000 to 2020 in regions with different aridity levels in the drylands of Australia

According to nighttime lights (NTL), socioeconomic activities in dry humid areas in Australia remained relatively stable during 2000–2010 but rose rapidly later, indicating that urban development in the drylands of Australia significantly accelerated after 2010 (Fig. 11.24b) but remained unchanged in drier regions.

11.6 Summary and Conclusion

Dryland social-ecological systems in Australia are facing the accelerated warming and rapid socioeconomic developments since the twenty-first century, including population and GDP increases and urban development since the twenty-first century, but with great diversity in space. In terms of spatial variance, forests in the drylands of Australia have become denser since the twenty-first century, but shrubs may have degraded. This result is consistent with the NPV, or both the PV and NPV, which generally decreased in arid and semiarid regions and vice versa in dry subhumid areas. Increases in the LAI/NPP were concentrated in the relatively humid coastal areas of Australia, whereas in the arid interior part, the LAI/NPP generally declined. Precipitation changes dominated the variation in vegetation in the drylands of Australia (legacy effects exist), where short vegetation is more easily influenced by precipitation changes than trees. Reductions in fire have significant impacts on emission mitigation and air purification but may have adverse effects on endemic biodiversity. Fire management (i.e., proactive burning) is necessary both to conserve biodiversity and to reduce the negative impacts on socioeconomic systems (fight fire with fire). The roles of livestock grazing/fencing in biodiversity are heterogeneous. Both grazing and fencing can be useful management tools to achieve conservation objectives and can also be threats to biodiversity conservation. Australia has invested considerably in improving biodiversity since the late 1980s. Integration of policy makers, funding agencies, and the general public are essential for the next step of dryland social-ecological system conservation in Australia.

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Chapter 12
Structure and Functioning of China's Dryland Ecosystems in a Changing Environment

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Abstract China has one of the largest dryland areas worldwide, covering 6.6 million km^2 and supporting approximately 580 million people. Conflicting findings showing a drier China's drylands with increasing aridity and observed greenness indicate the complexity of environmental processes, highlighting a pressing research need to improve understanding of how active dryland processes, ecosystem structure and functioning will alter. This chapter synthesizes the changes, impacts, and their drivers in China's dryland ecosystems. Results from analyses covering the period 2000–2015 showed that 58.69% of the vegetated area exhibited an increase in vegetation greenness, cover, and productivity, while 4.29% of those showed a decrease in all three aspects. However, 37.02% of the vegetated area showed inconsistent trends in vegetation greenness, cover, and productivity, suggesting high uncertainty in estimations of vegetation dynamics in drylands. China's drylands are nevertheless at risk of expansion and could pass an irreversible tipping point with increasing aridity, particularly in the country's semi-arid regions. Nitrogen enrichment and overgrazing generally reduce plant species diversity. Wind erosion, water erosion, salinization, and freeze–thaw erosion are typical processes of desertification in China's drylands. Large-scale ecological restoration projects enhance greening and ecosystem services of China's drylands, but also impose substantial pressure on these water-limited environments. Future research is needed to examine interactions among different drivers of environmental change (e.g., the relationships between CO_2 fertilization and increased

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aridity). Such research could usefully include complex systems approaches to link patterns and processes across spatial and time scales, and long-term experiments on physical-chemical-biological process interactions.

Keywords Climate change · Land use change · Desertification · Ecological conservation and restoration · China's drylands

12.1 Background

Drylands are regions where the Aridity Index (AI, determined by dividing mean annual precipitation by potential evapotranspiration) is below 0.65 (Huang et al. 2017a). Globally, drylands occupy ~41% of terrestrial land surface, supporting more than 38% of the world's population, of which approximately 90% are in developing countries (Berdugo et al. 2017; Reynolds et al. 2007). Drylands are characterized by scarce and highly variable annual precipitation, high potential evapotranspiration, low fertility of soils, and sparse vegetation (Huang et al. 2017a; Smith et al. 2019). Dryland ecosystems play an important role in providing numerous services such as water, food, fiber, habitat, biodiversity, and carbon sequestration (Ahlström et al. 2015; Bestelmeyer et al. 2015; Poulter et al. 2014). However, the sustainability of these ecosystem services is a concern, as drylands are considered to be fragile ecosystems and extremely sensitive to land degradation induced by climate change and human activities (Costanza et al. 2014; D'Odorico et al. 2013; Huang et al. 2017a; Maestre et al. 2016; Middleton and Sternberg 2013). Studies have reported that global aridity is increasing, and that the world will be drier in the future due to climate change (Huang et al. 2016; Li et al. 2023; Lian et al. 2021; Park et al. 2018). For example, it is estimated that drylands will expand up to 56% of the Earth's surface under the RCP8.5 scenario, or up to 50% under the RCP4.5 scenario, respectively (Huang et al. 2016). One recent aridity database analysis (covering 1950–2000) showed that global drylands have expanded by almost 4% in this time period, with major expansion in the arid (+3.4%) and semi-arid (+0.9%) regions (Prăvălie et al. 2019). Aridity is increasing in almost all continents except for Europe and South America, focusing on low and middle latitudes (Prăvălie et al. 2019). However, these findings are inconsistent with observed increases in greenness over drylands (He et al. 2019). Several studies have found that the AI is not an accurate proxy for defining drylands, as it fails to consider the key role of varying atmospheric CO_2 concentrations that drive climate change and its impacts on vegetation (Berg and McColl 2021; Stringer et al. 2021). The expansion of global drylands with a decreasing AI contrast with findings from other studies that use variables such as precipitation and soil moisture to identify drylands (Berg and McColl 2021; Lian et al. 2021; Roderick et al. 2015; Zhang et al. 2020a).

Of all the countries with drylands in the world, China ranks second in its extent of dryland areas after Australia (Prăvălie 2016). More than half of China's land surface (56.48%) is defined as dryland (Prăvălie 2016). China's drylands are home to approximately 580 million people, accounting for 20% of the world's population living in drylands (van der Esch et al. 2017). China alone accounts for almost one third of increases in dryland expansion worldwide (Prăvălie et al. 2019). Desertification is land degradation in drylands and is prevalent in China's drylands, challenging water supply, food security, and carbon sequestration (Wang et al. 2008). China leads globally in large-scale land conservation and restoration programs to combat desertification, greening the country's drylands (Bryan et al. 2018). However, large-scale ecological restoration projects also impose substantial pressure on these water-limited environments (Cao 2008; Wang et al. 2010).

During the last two decades, increasing research effort has been devoted to understanding China's dryland ecosystems and their responses to ongoing global change (Ci and Yang 2010; Huang et al. 2017a; Wang et al. 2008; Yang et al. 2011). This chapter aims to provide a comprehensive understanding of the basic characteristics, changes, and drivers of China's drylands. It reviews the key fronts on which progress has been made, suggests research priorities in both the near- and long-term, and proposes possible strategies to address the main remaining research gaps. It is essential to advance understanding and develop appropriate strategies to cope with continued climate changes and ecosystem dynamics. Such efforts can help inform actions to advance towards the sustainable development goals (SDGs) in China's drylands, offering insights for other global drylands.

This chapter is structured to provide the following:

(1) A review of the major characteristics of China's drylands, including their distribution, climate, soil, land uses, land degradation, eco-hydrological processes, and social and economic development;
(2) A synthesis of current understanding of the changes in China's drylands, covering dryland dynamics, structure and functions, ecosystem services, and human well-being, and considering the livelihoods of local communities;
(3) A discussion of the factors affecting ecosystem structure and functioning of dryland ecosystems under environmental change;
(4) A synthesis of major research priorities and potential approaches to address them.

12.2 Major Characteristics of Drylands in China

12.2.1 Distribution and Landforms

Based on the AI, drylands can be further classified as hyper-arid (AI < 0.05), arid (0.05 <= AI < 0.20), semi-arid (0.20 <= AI < 0.50), and dry sub-humid (0.50 <= AI < 0. 65) areas (Huang et al. 2017a). China's drylands cover an area of approximately 657.52×10^4 km^2, accounting for about 66% of the terrestrial surface (Fig. 12.1a), among which the hyper-arid, arid, semi-arid, and dry sub-humid areas are 84.20×10^4 km^2 (8.55%), 208.64×10^4 km^2 (21.17%), 256.46×10^4 km^2 (25.99%), and 108.23

$\times 10^4$ km^2 (10.96%), respectively. China's drylands are mainly located in latitudes between 30° and 50° N, and in longitudes between 75° and 135° E (Fig. 12.1a).

Drylands are mainly located in north China, covering 17 provinces, municipalities, and autonomous regions, including Xinjiang, Tibet, Qinghai, Inner Mongolia, Gansu, Ningxia, Shaanxi, Shanxi, Hebei, Henan, Shandong, Jiangsu, Heilongjiang, Jilin, Liaoning, Beijing, and Tianjin. Among all the provinces that have drylands, Xinjiang and Inner Mongolia have the largest dryland areas, with 173.46 $\times 10^4$ km^2 and 114.25 $\times 10^4$ km^2, respectively (Table 12.1). Drylands in Xinjiang are dominated by hyper-arid and arid regions, covering 65.29 $\times 10^4$ km^2 and 87.84 $\times 10^4$ km^2, respectively.

The topography of China's drylands varies greatly, and is mainly composed of inland basins (e.g., Tarim Basin, Junggar Basin), high plateaus (i.e., Qinghai-Tibet plateau, Loess Plateau, Inner Mongolia Plateau), and high mountain systems (e.g., Himalayas Mountains, Tianshan Mountains, Kunlun Mountains and Qilian Mountains). Deserts and Gobi landforms are widely distributed in China's drylands (Fig. 12.1b). Deserts cover an area of 56.34 $\times 10^4$ km^2, among which Taklimakan Desert and Gurbantunggut Desert are the largest. Gobi is mainly distributed around the Tarim Basin, Junggar Basin, the foothills of the Kunlun Mountain, Tianshan Mountains, and Hexi Corridor. Gobi is mainly formed by external forces such as wind and water power, and its surface is mainly gravel, which is different from sandy land. Loess landforms are mainly distributed on the Loess Plateau, showing large hilly and gully areas with loose soils that are easily eroded and transported by running water.

12.2.2 Climate, Soil, Land Uses, and Land Degradation

The formation and evolution of landforms in drylands result from the combined effects of abiotic factors (e.g., rainfall, temperature, soil), biotic attributes (e.g., vegetation), and land degradation.

Climate

Most of the drylands in China are located in the central Eurasian continent surrounded by high mountains and plateaus, where the moist summer monsoon from the Pacific Ocean cannot penetrate deep into the northwest hinterland; and nor can the wet summer monsoon from the Indian Ocean due to the barrier of the Himalayas Mountains and the high Qinghai-Tibet Plateau (Li and Ling 1992). These areas are consequently characterized by water scarcity and drought. Water resources are limited as precipitation (mean: 304.0 mm; Std: 22.6 mm) is typically much lower than potential evapotranspiration (mean: 814.9 mm; Std: 25.5 mm) (Fig. 12.2a, b). Precipitation is both temporally and spatially highly variable. Rainfall during the year usually occurs as short-duration and high-intensity rainstorms during a relatively short rainy season from June to September. Multiple precipitation pulses occur alternately with dry

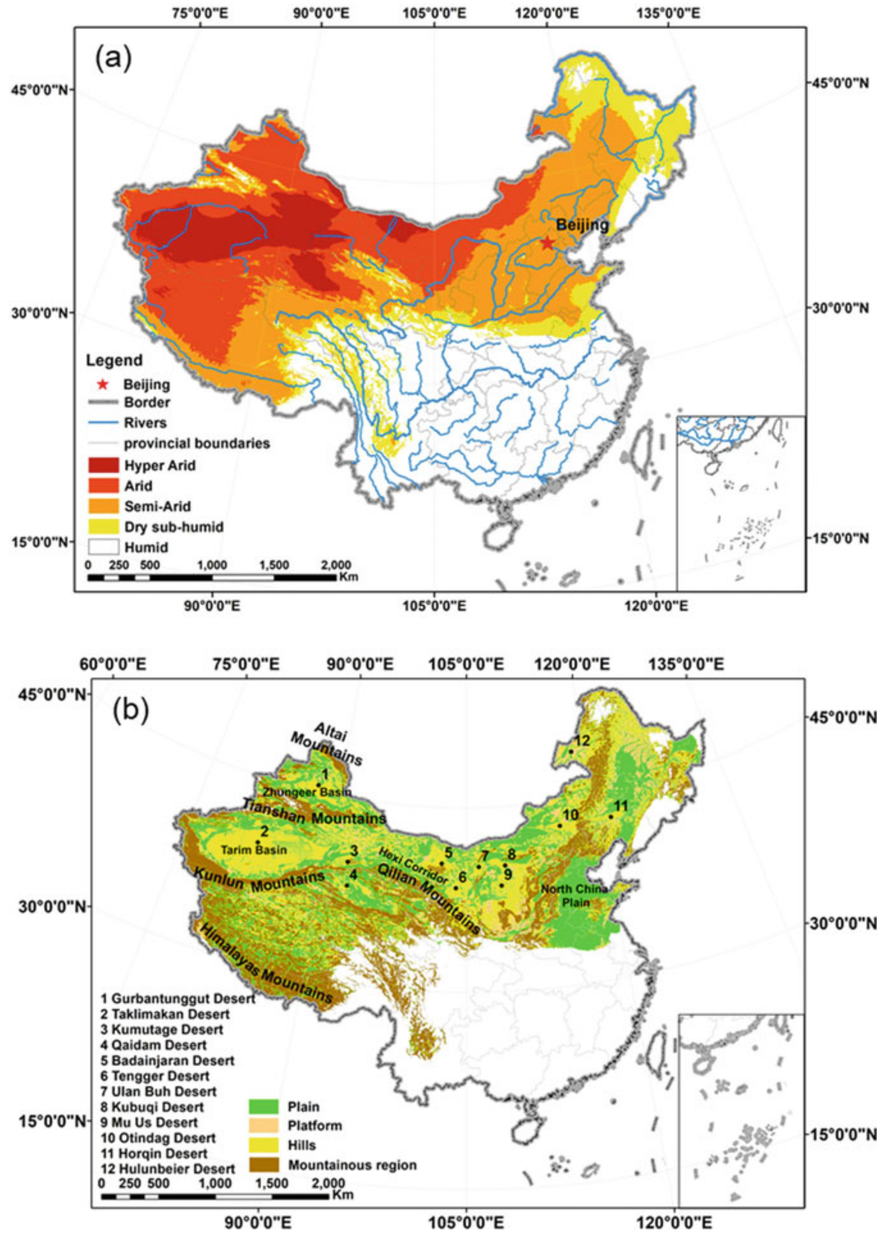

Fig. 12.1 **a** Spatial distribution of drylands and four subtypes in China. Data is derived from the Global Aridity Index database (Trabucco and Zomer 2009). **b** Landforms and location of deserts in China's drylands. Data is derived from data sets provided by Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (RESDC) (http://www.resdc.cn)

Table 12.1 Area and proportion of drylands and sub-types in different provinces and cities of China

Province	Area and proportion of arid areas at all levels (10^4 km^2/%)				
	Hyper arid	Arid	Semi-arid	Dry sub-humid	Sum
Anhui	0	0	0.88	3.06	3.94
	0.00%	0.00%	6.23%	21.74%	27.97%
Beijing	0	0	1.67	0	1.67
	0.00%	0.00%	100.00%	0.00%	100.00%
Gansu	4.09	15.45	10.65	5.52	35.71
	10.11%	38.22%	26.34%	13.65%	88.30%
Hainan	0	0	0	0.03	0.03
	0.00%	0.00%	0.00%	0.00%	0.00%
Hebei	0	0	19.03	0.02	19.06
	0.00%	0.00%	99.71%	0.17%	99.88%
Heinan	0	0	6.63	4.84	11.47
	0.00%	0.00%	40.17%	29.32%	69.49%
Heilongjiang	0	0	10.40	30.11	40.51
	0.00%	0.00%	23.12%	66.92%	90.03%
Hubei	0	0	0	0.01	0.01
	0.00%	0.00%	0.00%	0.08%	0.08%
Jilin	0	0	7.24	4.92	12.16
	0.00%	0.00%	37.76%	25.64%	63.40%
Jiangsu	0	0	0.94	2.67	3.61
	0.00%	0.00%	9.37%	26.57%	35.94%
Liaoning	0	0	7.64	3.25	10.89
	0.00%	0.00%	52.70%	22.40%	75.11%
Inner Mongolia	6.68	43.47	52.05	12.06	114.26
	5.82%	37.89%	45.37%	10.51%	99.59%
Ningxia	0	2.50	2.37	0.16	5.03
	0.00%	49.54%	46.93%	3.10%	99.57%
Qinghai	7.60	11.62	33.38	10.81	63.41
	10.60%	16.20%	46.54%	15.07%	88.42%
Shandong	0	0	10.72	5.08	15.80
	0.00%	0.00%	68.02%	31.92%	99.95%
Shanxi	0	0	15.42	0.42	15.83
	0.00%	0.00%	97.18%	2.63%	99.82%
Shaanxi	0	0	10.63	2.80	13.43
	0.00%	0.00%	51.57%	13.59%	65.16%
Sichuan	0	0	0.40	5.40	5.81

(continued)

Table 12.1 (continued)

Province	Area and proportion of arid areas at all levels (10^4 km^2/%)				
	Hyper arid	Arid	Semi-arid	Dry sub-humid	Sum
	0.00%	0.00%	0.71%	9.59%	10.30%
Tianjing	0	0	1.09	0.00	1.09
	0.00%	0.00%	93.47%	0.00%	93.47%
Tibet	0.55	47.76	47.02	10.14	105.47
	0.45%	39.63%	39.02%	8.41%	87.52%
Xinjiang	65.29	87.84	17.83	2.53	173.49
	37.29%	49.84%	10.09%	1.43%	98.65%
Yunnan	0	0	0.35	4.41	4.76
	0.00%	0.00%	0.92%	11.49%	12.40%
Total	84.20	208.64	256.36	108.23	657.42
	8.55%	21.17%	25.98%	10.96%	66.50%

periods. The interannual variation of rainfall is typically high, in particular in hyper-arid and arid regions (Li et al. 2021). Rainfall varies greatly over short geographical distances, with high rainfall in mountainous regions but scarce rainfall in the surrounding plains. Rainfall differs across the gradient of hyper-arid, arid to semi-arid and dry sub-humid areas, increasing gradually from the northwest towards the east, south, and southeast (Fig. 12.2a). Runoff in response to rainfall events in China's drylands is dominated by infiltration-excess overland flow (runoff production when the rainfall intensity is greater than the soil infiltration capacity), while saturation-excess overland flow (runoff production when the unsaturated zone and saturated portion of the soil profile is saturated by long periods of rainfall) is seldom observed. High intensity events during the rainy season frequently lead to flashy runoff and low infiltration, while rainfall with low intensity seldom produces runoff due to the high temperatures and the associated rapid and high rates of water loss to evaporation and transpiration. The spatial pattern of rainfall and evapotranspiration leads to runoff in upland drylands with altitude greater than 1,000 m, while the lowland drylands have no runoff production at all (Chen et al. 2015). Consequently, river networks across China's drylands landscapes are poorly dissected and dominated by ephemeral streams (existing only for a short period following rainfall or snowmelt) and intermittent streams (streams that exist for longer periods than an ephemeral stream but not all year round).

Mean temperature in the drylands of China ranges from −30 to 30 °C (Li and Ling 1992), and varies greatly between southern and northern parts (Fig. 12.2c). The maximum recorded temperature (49.6 °C) in China was documented in the famous "Fire Prefecture" of Turfan in Xinjiang, while nearby Fuyun was recorded one of the lowest minimum temperatures (−51.5 °C). The diurnal temperature range is substantial, with the temperature difference in the Tarim Basin as much as 15–20 °C. This big difference in temperature between night and day is observed in

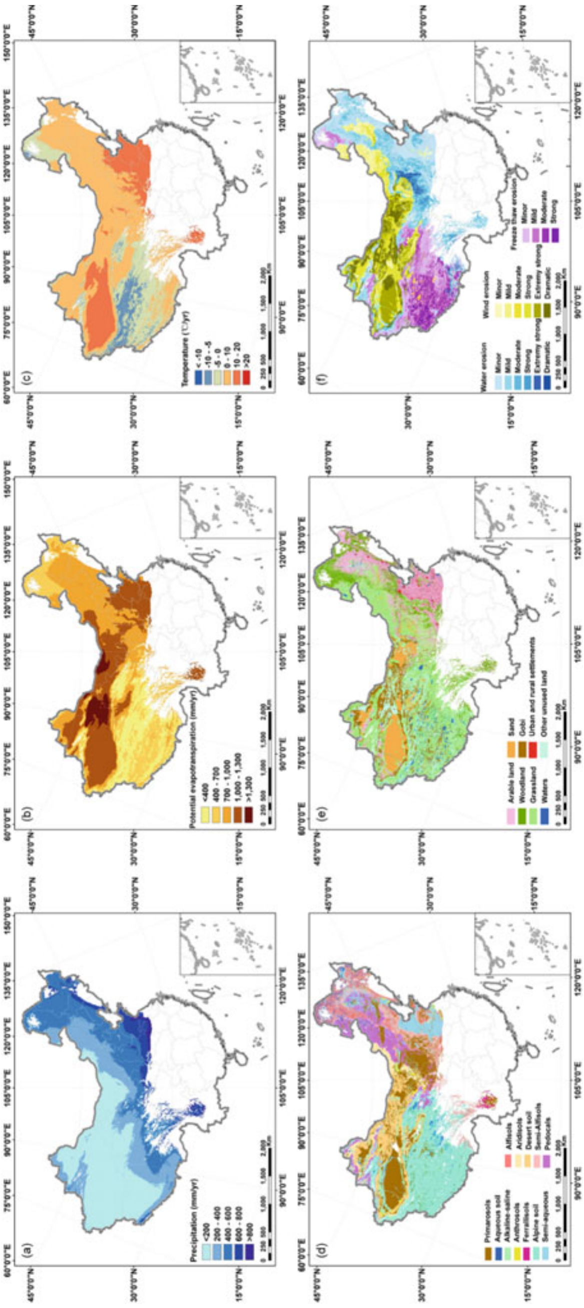

Fig. 12.2 Spatial distribution of basic characteristics in the drylands of China, including **a** precipitation, **b** potential evapotranspiration, **c** temperature, **d** soil types, **e** land uses, and **f** land degradation processes

an old saying: "Cotton-padded jacket in the morning, T-shirt at noon, and enjoy watermelons around the stove". Both altitude and terrain influence temperatures. The temperature decreases by 5–6 °C with an increase of 1,000 m in altitude. In the northern Qinghai-Tibet Plateau where the altitude is approximately 4,000–5,000 m above sea level, the mean temperature in July stays below 10 °C.

Drylands in China are rich in solar and wind energy. The total annual duration of sunshine varies from 2,500 to 3,000 h, with annual solar radiation from 136 to 160 kcal/cm^2 (Li and Ling 1992). High solar radiation usually occurs in the Qinghai-Tibet Plateau, the Tarim Basin, and Hexi Corridor in Northwest China.

Soil

There are many kinds of soil both in horizontal and vertical zonality which has resulted from the varied patterns of climate, rock formation, topography, vegetation, and the long history of agricultural development in China's drylands (Li and Ling 1992). The 1:1,000,000 soil database in the drylands of China was tailored from the 1:1,000,000 soil database in China that was based on the 1:1,000,000 soil maps of China compiled and published by the National Soil Census Office in 1995. The spatial database was based on the soil genetic classification of China, including 12 orders, 61 groups, and 227 subgroups (Shi et al. 2004b). From arid, semi-arid in the northwest to sub-humid in the middle and east of the China's drylands, the major soil type ranges from desert soil to steppe soil, and to the forest-steppe soil sequences (Li and Ling 1992). The largest soil order in China's dryland is Alpine soil (Fig. 12.2d), which is widely distributed on the Qinghai-Tibet Plateau, covering a total dryland area of 168.89×10^4 km^2. Alpine soils are further classified into felty soils, dark felty soils, frigid calcic soils, cold calcic soils, cold brown calcic soils, frigid desert soils, cold desert soils, and frigid frozen soils. Alpine soils are mainly in the high-altitude cold region where soil erosion by freezing and thawing is substantial, so the soil layers generally have frozen layers and permafrost. Soil biological function in Alpine soil areas is weak due to the poor hydrothermal conditions, resulting in sparse vegetation cover and slow accumulation of humus. Primarosols are widely distributed in the arid and semi-arid regions of China, such as the Tarim Basin, Zhungeer Basin, and the Loess Plateau, with a total area of 122.58×10^4 km^2. Primarosols in the Taklimakan Desert is further classified as aeolian sandy soil, and that in the Loess Plateau is Loessial soil. Desert soil and Aridsols are widely distributed in northeastern Xinjiang, northwestern Gansu, and western Inner Mongolia, with the total area of 62.67×10^4 km^2 and 31.76×10^4 km^2, respectively. These two types of soil are vulnerable to erosion, with low nutrients and poor fertility, meaning they are not conducive to vegetation growth and farming. Alkali-saline soils are widely distributed in low altitude areas such as the plains, basins, and valleys of arid and semi-arid inland regions where the groundwater table is high and there is considerable evaporation of surface water, making soluble salts in the subsoil easily drawn up into the topsoil.

Land Uses

China's drylands are dominated by grasslands (2.3 million km^2, 34%), desert (1.4 million km^2, 21%), and croplands (1.1 million km^2, 16%) (Fig. 12.2e). Large amounts

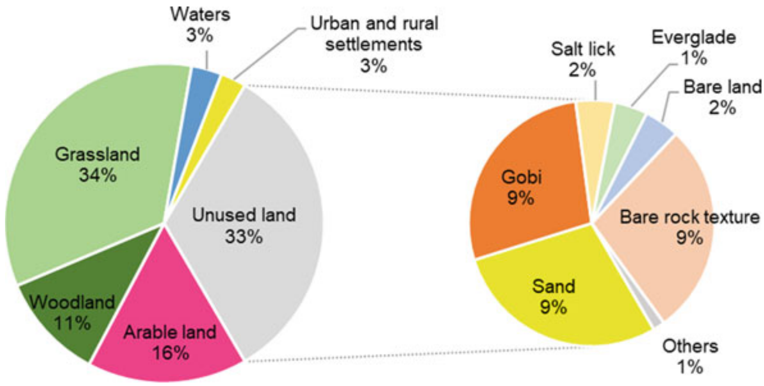

Fig. 12.3 Pie chart showing the proportion of land uses in China's drylands in 2020. Data is derived from data sets provided by Data Center for Resources and Environmental Sciences, Chinese Academy of Sciences (RESDC) (http://www.resdc.cn)

of unused land including Gobi, sandy land, and bare rocky land account for 33% of the total dryland area (Fig. 12.3). Most of these areas are barren land with sparse vegetation due to natural conditions and human activities such as overgrazing (Li and Ling 1992). The spatial pattern of land uses in China's drylands shows that cultivated land, urban and rural settlements are mainly in the North China Plain. Large areas of sandy land and Gobi are concentrated in northern Xinjiang, western Inner Mongolia, and western Gansu.

Ecohydrology

Plant growth is mainly determined by the available soil moisture during the growing season (Bai et al. 2004; Wu et al. 2011). The spatial and temporal patterns of available water strongly govern dryland vegetation (Scott et al. 2014). Hydrological processes influence the distribution, structure, function, and dynamics of biological communities, while feedbacks from biological communities affect the water cycle (Fig. 12.4a). Investigating the two-way interactions between and interdependence of ecological and hydrological processes is essential to better understand ecosystem dynamics in drylands (Brauman et al. 2007; Newman et al. 2006; Turnbull et al. 2008; Scott et al. 2014).

Dryland vegetation is typically patchy and heterogeneous. Studies conducted at multiple spatial scales (e.g., plot, hillslope, catchment) have found that the patchy vegetation affects the temporal and spatial pattern of water, sediment and nutrients, soil microbial biomass, and functional diversity (Hu et al. 2010; Li et al. 2008), and ultimately the functioning of China's drylands (Fu et al. 2003; Wang et al. 2001).

Land Degradation

In ancient China, most drylands were covered by dense forests and grasses, and the soil was fertile for agriculture development and grazing (Li and Ling 1992). However, the once-productive ecosystem has been historically deteriorated by human activities

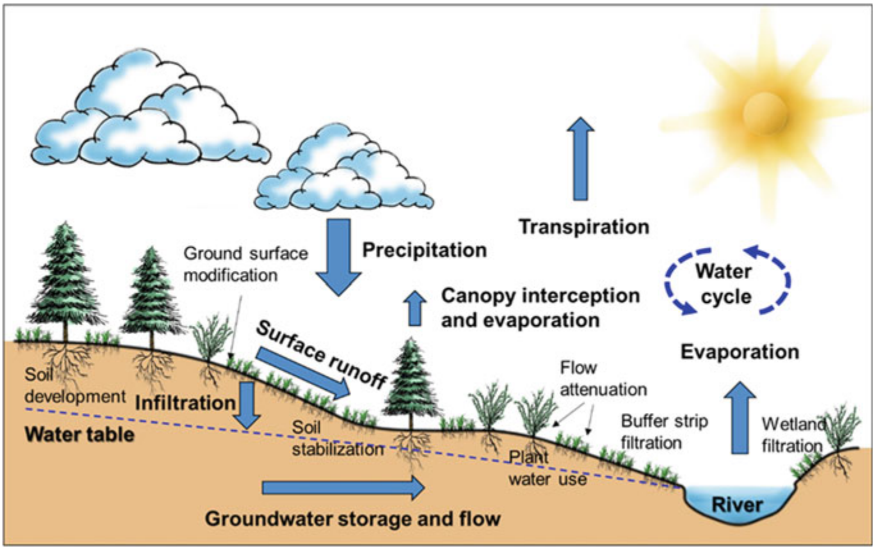

Fig. 12.4 Water cycle and ecosystem interactions in drylands (after Brauman et al. (2007)). At the watershed scale, dryland ecosystems affect water through canopy interception, evaporation, transpiration, water use by plants (i.e., forest, shrub, and grass), flow attenuation, and ground surface modification. The hydrological cycle driven by solar energy includes precipitation, infiltration, surface flow, ground flow, and evaporation. Water fluxes are indicated by arrows. The water balance equation is expressed as precipitation = evapotranspiration (transpiration + evaporation) + discharge (surface + ground water) + change in water storage (surface + ground water)

(e.g., land reclamation, farming) during the last 6,000 years, leading to natural hazards such as widespread drought, soil erosion, and salinization. Erosion by wind, water and freeze–thaw are three key processes of desertification in China's drylands (Shi et al. 2004a), which together affect 95.4% of the country's dryland area (Li et al. 2021). Breaking this down, the dryland regions affected by wind erosion, water erosion, and freeze–thaw erosion cover 2.28 million km^2 (34.5% of China's total drylands), 2.46 million km^2 (37.4% of total drylands), and 1.55 million km^2 (23.5% of total drylands), respectively. More than half (56.2%) of drylands affected by wind erosion (mostly in northwestern and northern arid and hyper arid regions) experience strong (5000–8000 tons km^{-2} yr^{-1}), extremely strong (8,000–15,000 tons km^{-2} yr^{-1}), and dramatic (>15,000 tons km^{-2} yr^{-1}) magnitudes of erosion. However, the drylands affected by water erosion and freeze–thaw erosion are predominantly influenced by minor, mild, or moderate erosion (<5,000 tons km^{-2} yr^{-1}), which accounts for 92.9% and 94.6% of the water erosion and freeze–thaw erosion regions, respectively.

Since the establishment of the People's Republic of China in 1949, a series of important policies and measures (e.g., Three-North Shelterbelt Development Program, Grain for Green Program) have been adopted to develop the drylands (Li and Ling 1992). Land degradation increased thereafter and peaked in the early 1980s (Wang et al. 2008). The widespread land degradation in China's drylands seriously constrained socioeconomic development, especially before the end of the twentieth century (Lü et al. 2012). However, the land degradation trend has been reversed since 1980s as observed by Normalized difference vegetation index (NDVI) (Piao et al. 2005). NDVI has usually been used as a proxy assessment of land degradation or improvement, but it fails to consider other influencing factors such as climate which could be represented by rain-use efficiency (RUE). Land degradation as measured by RUE-adjusted annual sum NDVI analysis showed that 80% of degrading areas are in the humid and cold-climate zone (i.e., non-dryland areas); while drylands have a much lower proportion of degrading areas, with 10% in the dry sub-humid, 5% in the semi-arid, and 5% in the arid and hyper-arid areas (Bai and Dent 2009).

12.2.3 Social and Economic Development

China's drylands have a population of 580 million, representing 41.6% of the total population of the country. The areas with the highest population densities in China's drylands are in the southeast provinces of Beijing, Tianjin, Shandong, Hebei, and Henan. Coastal areas of Northeast China are also densely populated (Fig. 12.5a). The western region is sparsely populated, and there are large areas devoid of human populations.

Nighttime lights offer an important indicator to measure the degree of regional economic and social development. Lighting information based on satellite sensors is closely related to urban development and human activities. The nighttime light map of China's drylands (Fig. 12.5b) shows a distribution highly consistent with the distribution of GDP and population. The eastern part of the drylands, especially the North China Plain, is densely populated with towns and high levels of economic development, and these areas have larger brightness values of remote sensing image pixels. There are also several bright spots in the Northeast parts of the drylands, in the provincial capitals.

Gross Domestic Product (GDP) is usually used to measure the total value of all final products and services produced by a country or region in a one-year cycle. The size of GDP is closely related to population distribution and urban development (Fig. 12.5c). In 2015, the annual GDP of China's drylands was about 26,095 billion yuan (Fig. 12.5d), accounting for 41.9% of national GDP. The GDP of the semi-arid and dry sub-humid regions accounted for 39.5% of the national GDP. The GDP of China's drylands is mainly provided by the two dryland sub-types in the eastern part of China's drylands. The GDP contribution of the hyper-arid area is maintained at a low level with no significant growth over the past 20 years. The GDP of China's drylands decreases from southeast to northwest. The Qinghai-Tibet Plateau and southern Xinjiang have extremely low GDP levels due to their sparse populations.

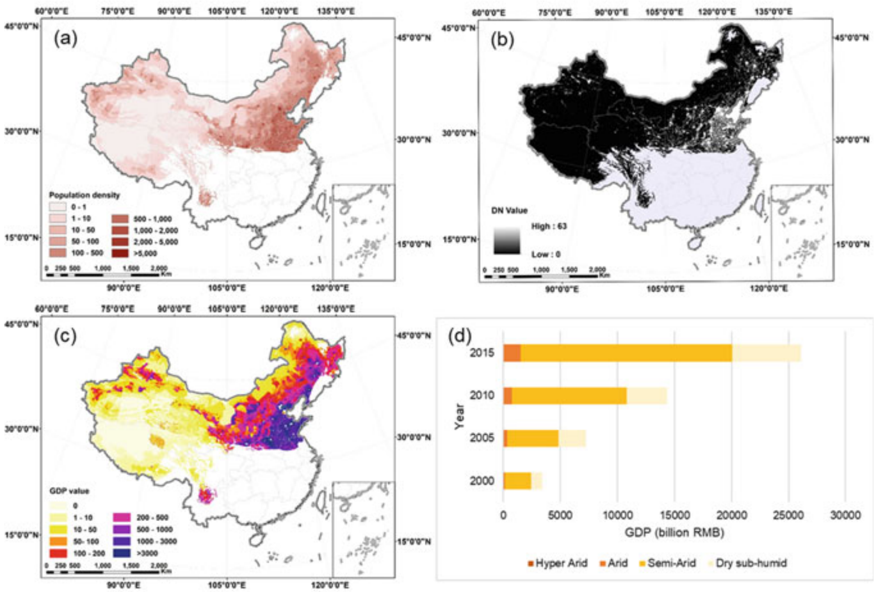

Fig. 12.5 **a** Spatial distribution of population density, **b** Gross Domestic Productivity (GDP), **c** Night Light Map (NLM), and **d** trends of GDP in the drylands of China. The DN value represents the brightness value of remote sensing image pixels, and records the gray scale of ground objects in the range of 0–63. The larger the DN value, the brighter it is

12.3 Changes to Drylands in China

12.3.1 Structure and Functions

Climate Change

China's drylands have experienced temperature increases of 4.12 °C during 1980–1997 and 4.93 °C during 1997–2015, with an average annual increase of 0.013 °C (Fig. 12.6a). However, precipitation showed non-significant trends ($p > 0.1$) in general (Fig. 12.6b). Potential evapotranspiration increased from 798.74 mm during 1980–1997 to 831.09 mm during 1997–2015, with average annual increases of 1.30 mm (Fig. 12.6c). The changing trends of temperature, precipitation, wind speed, and potential evapotranspiration followed a cycle of 2–4 years (Fig. 12.7).

Except in a few areas in the northwest (hyper arid areas) and northeast, temperature increased in the most of drylands of China (Fig. 12.8a). Precipitation declined in northeastern China, but increased in the northern arid and semi-arid areas (Fig. 12.8b). Wind speed declined in some of northeastern and northwestern China, but increased in the northern and southwestern arid and semi-arid areas (Fig. 12.8c). Potential evapotranspiration increased in most of the drylands except the western Tibetan Plateau (Fig. 12.8d).

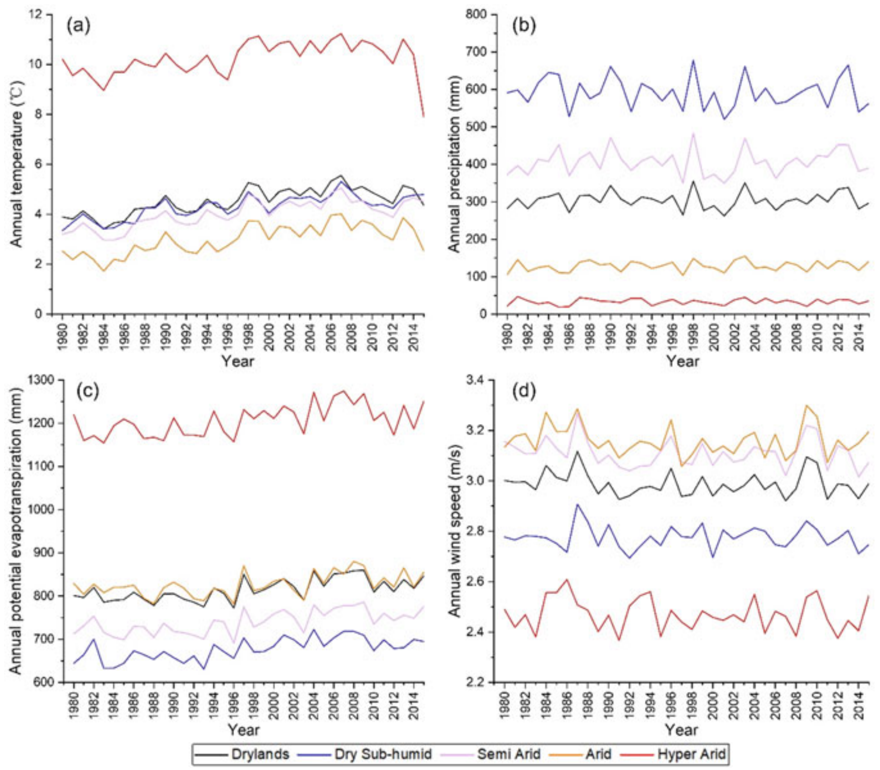

Fig. 12.6 Annual climate change in China's drylands and four subtypes during 1980–2015. **a**, **b**, **c**, **d** show the interannual variation of mean temperature, precipitation, potential evapotranspiration, and wind speed, respectively

Land Use/Cover Change

Land use/cover change in China's drylands showed a decrease in forest from 1970 to 2000, and an increase from 2000 to 2015 (Fig. 12.9). Grassland showed a continued decrease by 5.4×10^4 km^2 from 1970 to 2015, with reductions in all high, moderate and low coverage grassland. Cropland and construction land increased during the 1970–2015 by 5.8×10^4 km^2 and 3.4×10^4 km^2, respectively. The area of unutilized land reduced by 1.5×10^4 km^2 from 1970 to 2015 mainly in the sub-types of Gobi and Sandy land (Fig. 12.9).

Vegetation Indices Change

Numerous vegetation indices have been developed to investigate vegetation growth dynamics, including vegetation productivity, vegetation greenness, and vegetation cover (Ding et al. 2020). Due to the various indices to depict vegetation growth dynamics and great uncertainty in the estimation of vegetation change (Piao et al. 2020), it is essential to determine the consistency of vegetation growth dynamics

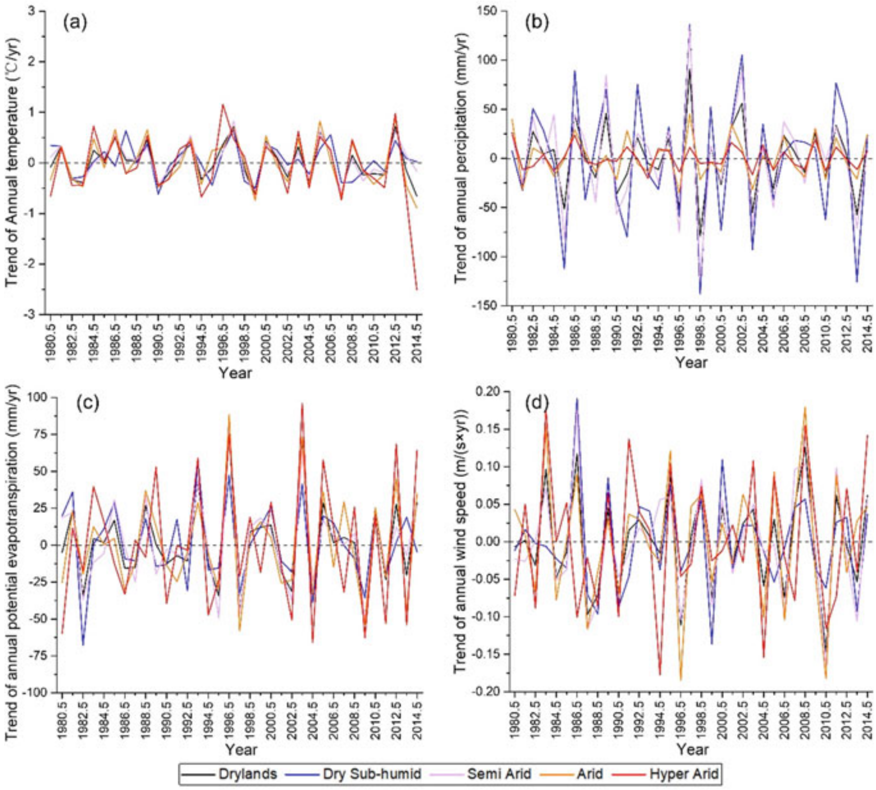

Fig. 12.7 Trend of annual climate change in China's drylands and four subtypes during 1980–2015. **a**, **b**, **c**, **d** show the dynamics of the trend of annual temperature, precipitation, potential evapotranspiration, and wind speed, respectively

using multiple indices (Ding et al. 2020). In this section, three widely used satellite-derived vegetation indices were applied to assess 2000–2015 vegetation growth trends in China's drylands. Specifically, net primary productivity (NPP), Normalized Difference Vegetation Index (NDVI), and the leaf area index (LAI) were used to characterize vegetation greenness, vegetation cover and productivity, respectively. Results showed that NPP, NDVI, and LAI increased in the drylands of China during 2000–2015 (Fig. 12.10a–c). The spatial distribution of the vegetation growth trends showed that there was a combination of vegetation improvement and degradation (Fig. 12.10d–f). Generally, vegetation indices increased in the central and eastern semi-arid and dry sub-humid regions, and decreased in the northwestern drylands. The area over which vegetation growth was enhanced was generally greater than the area with degraded vegetation. Overall, the distribution of vegetation growth trends was similar among NPP, NDVI, and LAI, but there are areas where distinct differences existed among different vegetation indices.

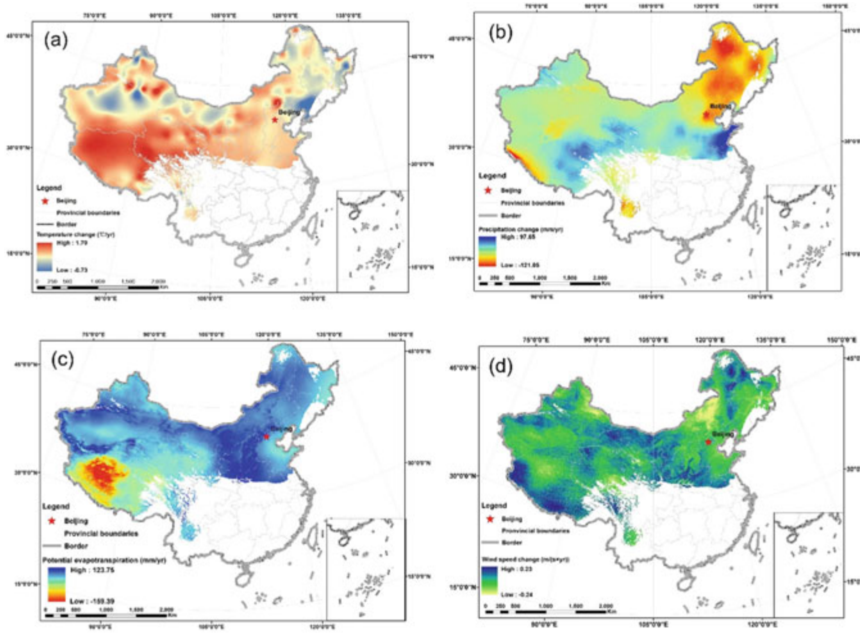

Fig. 12.8 Spatial trend of climate change in the drylands of China during 1980–2015. **a**, **b**, **c**, **d** show the trend of temperature, precipitation, wind speed and potential evapotranspiration respectively

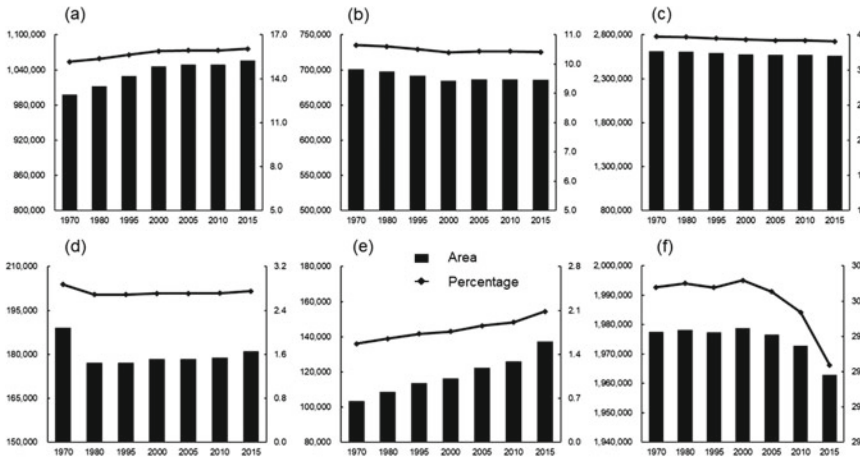

Fig. 12.9 Changes in land use distribution during the period 1970–2015 in the drylands of China. The primary Y-axis shows the area and the secondary Y-axis shows the percentage. Land uses considered include: **a** cropland; **b** forest; **c** grassland; **d** water body; **e** construction land; **f** unutilized land

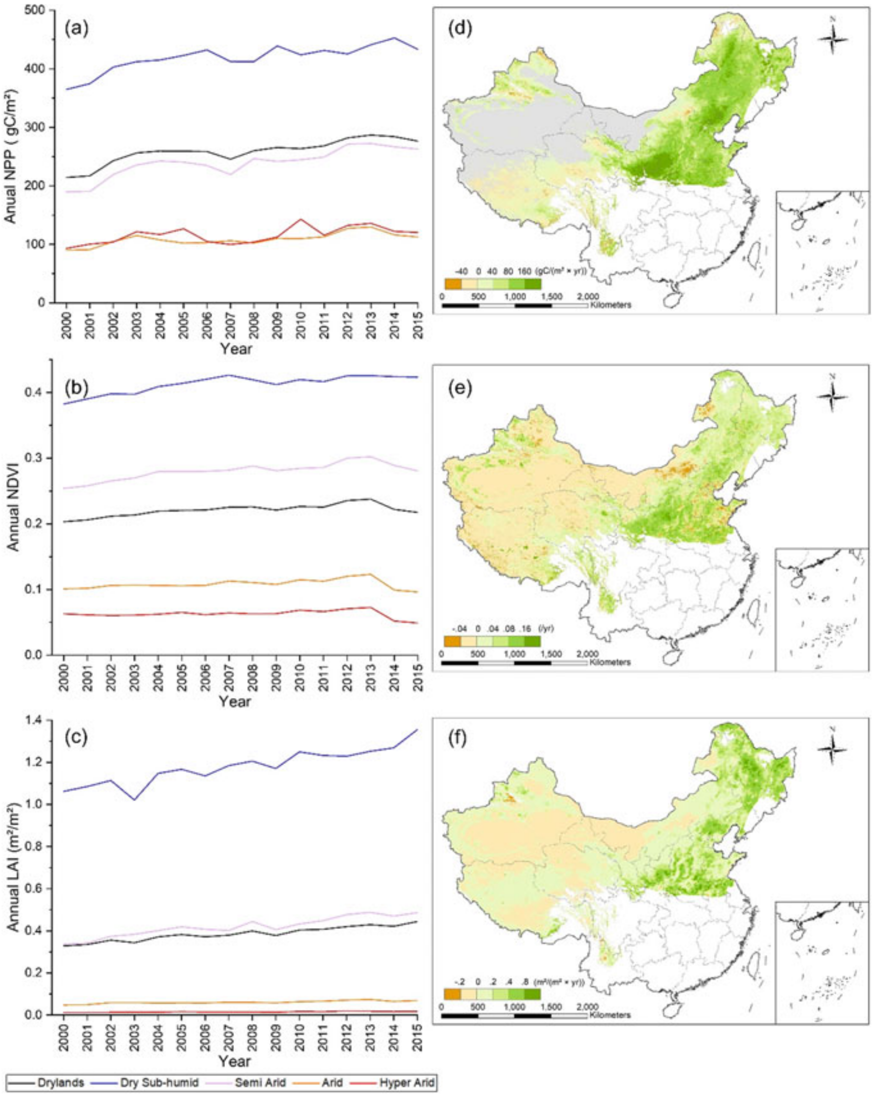

Fig. 12.10 Temporal and spatial patterns of NPP, NDVI, and LAI changes in the drylands of China during 2000–2015. Panels **a**, **b**, and **c** show the trends of annual NPP, NDVI, and LAI, respectively, in China's drylands and sub-types. Panels **d**, **e**, and **f** show the spatial pattern of NPP, NDVI, and LAI, respectively. The gray area represents no data

The combination of changes in vegetation productivity, vegetation greenness, and vegetation cover in China's drylands showed a diversity of vegetation growth dynamics (Fig. 12.11). 62.98% of the vegetated area exhibited an increase or decrease in all three aspects. In most of the eastern dry sub-humid and semi-arid areas, NPP, NDVI, and LAI all increased, but in some semi-arid areas of the Qinghai-Tibet plateau, all three vegetation indices decreased. 37.02% of the vegetated area experienced inconsistent trends in vegetation productivity, vegetation greenness, and vegetation cover. 15.98% of the vegetation area experienced enhanced vegetation productivity and cover, with degraded greenness, especially on the edges of semi-arid and arid areas (Fig. 12.11). Regions with increased greenness (NDVI) but decreased productivity (NPP), and vegetation cover (LAI) accounted for 1.85% of the vegetated area in drylands. Another noteworthy vegetation growth pattern is found in the regions where only vegetation productivity increased while greenness and cover decreased. Those areas accounted for 4.77% of the vegetated area and were concentrated in the northeastern Inner Mongolia region's drylands.

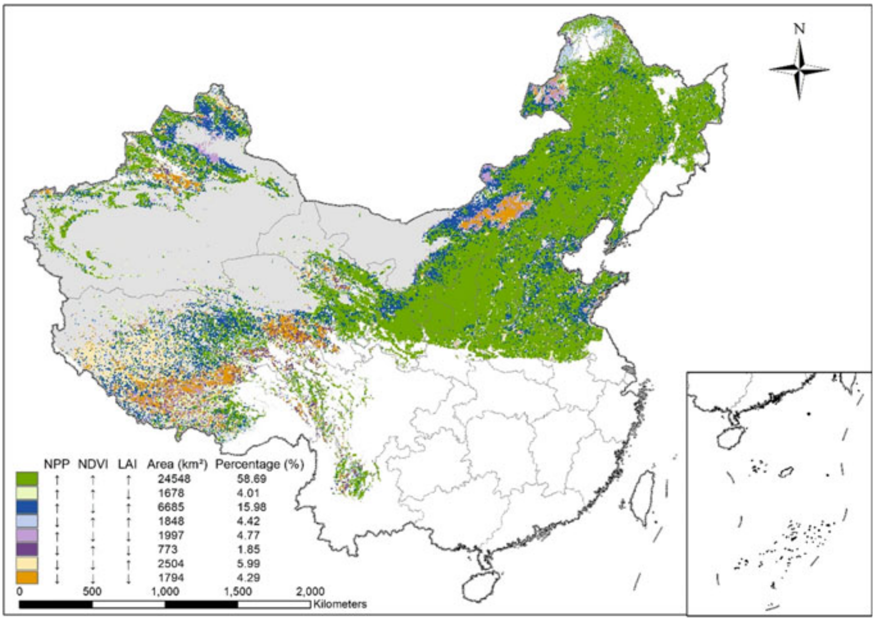

Fig. 12.11 Combination of trends in NPP, NDVI, and LAI of the drylands in China during 2000–2015. The gray area represents no data

12.3.2 Ecosystem Services

Water is the principal driver of ecological processes. Among the various services provided by ecosystems (Fig. 12.12), hydrological services (e.g., water supply) are the basis for realizing other services such as soil generation, carbon sequestration, and recreation (Brauman et al. 2007). To better understand and quantify water-related ecosystem services, it is essential to link ecohydrological processes (e.g., water, carbon, energy, and nutrient cycling) to ecosystem services (water and food security, and climate moderation) (Brauman et al. 2007; Sun et al. 2017). Water scarcity drastically limits dryland ecosystem services, particularly supporting and regulating services which are of great importance for soil formation, nutrient cycling, and water and climate regulation (Prăvălie 2016). The low freshwater availability of dryland ecosystems implies that water is insufficient to accommodate China's dryland population of 580 million while also ensuring optimal ecosystem functionality.

Dryland ecosystems in China are important in providing a wide range of ecosystem services including water yield, soil conservation, carbon sequestration, and habitat quality (Lü et al. 2012). Based on the datasets of ecosystem services evaluated by Xu

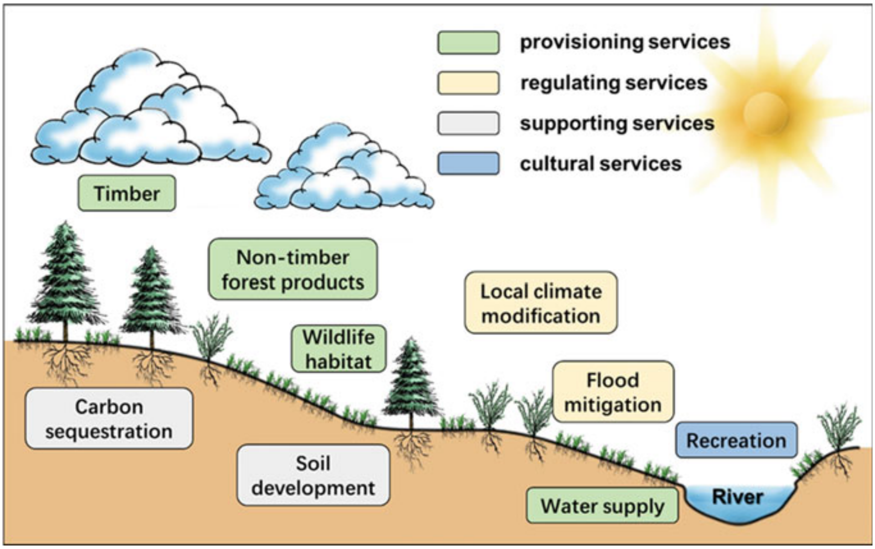

Fig. 12.12 Examples of hydrological and other services that a watershed produces, such as water supply, timber and non-timber forest products, soil development, carbon sequestration, and local climate modification and recreation. Based on the categories used by The Millennium Ecosystem Assessment (2005), provisioning services refer to the products obtained directly from ecosystems such as water, food, and timber; regulating services indicate that ecosystems have the ability to regulate processes such as climate, and the water cycle; supporting services are indirect services which are important for soil formation, nutrient cycling and so on; and cultural services refer to benefits that ecosystems provide to people including tourism, education, recreation, and aesthetic values

et al. (2020), the changes in the four major ecosystem services in China's drylands (e.g., water yield, soil conservation, carbon sequestration, and habitat quality) during 2000–2015 were examined. Results showed a strong correlation between the studied ecosystem services and aridity, indicating that the values for all four ecosystem services followed the order: dry-sub-humid > semi-arid > arid > hyper-arid.

Water yield, soil conservation, carbon sequestration, and habitat quality in dry sub-humid regions are the highest among the four dryland sub-types (Fig. 12.13). Significant conversions of farmland to woodland and grassland have resulted in enhanced soil conservation and carbon sequestration, but decreased regional water yield under a warming and drying climate trend. Water yield generally increased from 2000 to 2010, and then decreased from 2010 to 2015 (Fig. 12.13a). The spatial pattern showed that water yield increased in general but declined in southeastern and southwestern drylands. Soil conservation showed a non-significant trend during 2000–2015 (Fig. 12.13b). Spatially, soil conservation declined in southeastern and southwestern drylands, where water erosion and freeze–thaw erosion are serious, respectively. Carbon sequestration generally increased during 2000–2015, especially in eastern dry sub-humid and semi-arid areas, but decreased in some of the southwestern semi-arid and arid areas (Fig. 12.13c). The finding is consistent with the vegetation change, showing that NPP, NDVI, and LAI increased in most of eastern dry sub-humid and semi-arid areas, but decreased in some semi-arid areas of the Tibetan plateau. Habitat quality is highest in the northeastern semi-arid area and southwestern semi-arid and arid areas such as the Qinghai-Tibet Plateau. Habitat quality increased in arid and hyper arid areas, but decreased in dry sub-humid and semi-arid areas, especially in the east and northeast drylands (Fig. 12.13d).

12.4 Driving Forces of Dryland Change

The ecosystem structure, functioning and delivery of ecosystem services by drylands are substantially affected by multiple drivers, including climate change, dryland conservation practices, livestock grazing and fencing, and nitrogen deposition (Fu et al. 2021; Maestre et al. 2016). The following parts of this section give an overview of the drivers of change in the drylands of China.

12.4.1 Climate Change

Climate projections indicate that hotter, drier conditions and extreme rainstorms will continue to intensify over the twenty-first century (Feng and Fu 2013; Fu and Feng 2014), and are assumed to result in dryland expansion and further desiccation and degradation (Huang et al. 2016, 2017c, 2020). Ecosystems in the transitional regions (e.g., semi-arid regions) are fragile and highly sensitive to warming and drying, and are generally agricultural districts with large populations, leading to great challenges

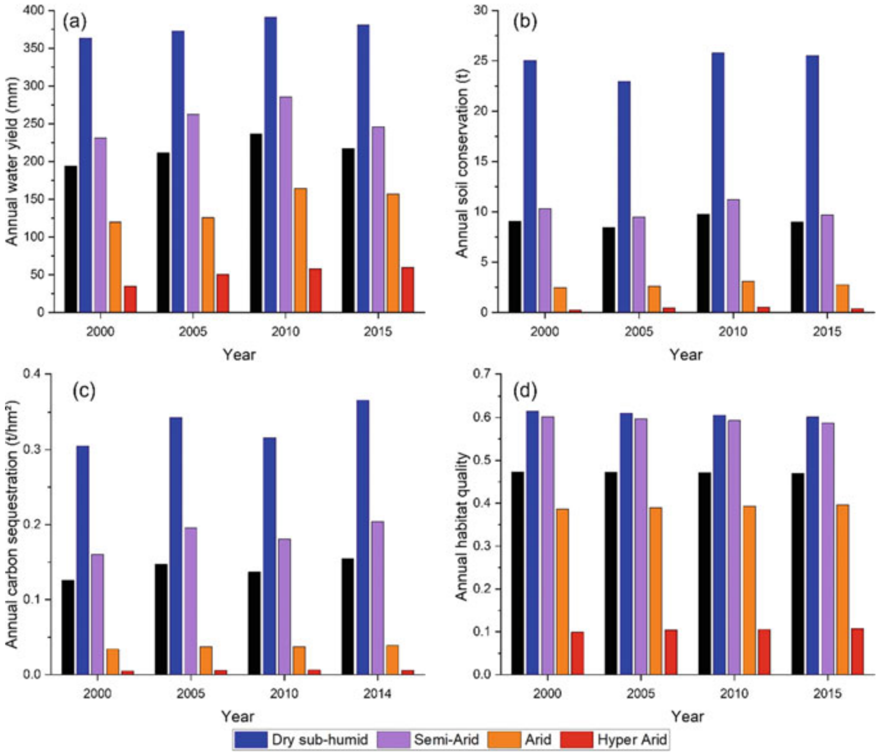

Fig. 12.13 Ecosystem service change in China's drylands and sub-types. **a**, **b**, **c**, and **d** are the water yield, soil conservation, carbon sequestration, and habitat quality, respectively

for both the ecosystem and human wellbeing (Huang et al. 2017a). Semi-arid drylands are highly sensitive to climate change (Huang et al. 2017b; Poulter et al. 2014), and the largest contributor to land-based carbon sink interannual variability, vital in regulating the climate (Ahlström et al. 2015). An expansion of 33% in China's semi-arid regions from 1948 to 2008 (Li et al. 2015b) will have reduced soil organic carbon storage and emitted CO_2 into the atmosphere (Maestre et al. 2016).

Global climate change is likely to produce higher aridity (Berdugo et al. 2020), which will cause negative ecological consequences by limiting soil moisture and disrupting vital C, N, and P biogeochemical cycles (Delgado-Baquerizo et al. 2013). Key ecosystem structures and the functional properties in drylands showed a strong nonlinear change with increasing aridity, indicating that dryland ecosystems could pass an irreversible tipping point as aridity increases (Berdugo et al. 2020; Delgado-Baquerizo et al. 2013; Wang et al. 2014a; Wardle 2013). Different climatic change drivers affect vegetation in different ways. Rising atmospheric CO_2 enhances water-use efficiency and plant growth (Li et al. 2013), while an increase in aridity negatively affects water availability and plant productivity (Berdugo et al. 2020; Maestre et al.

2016). However, it is still not known whether the positive effects of CO_2 fertilization can buffer the negative effects of increased aridity.

12.4.2 Livestock Grazing and Fencing

Due to increasing demand for meat, milk, and other livestock products, many dryland regions in China are seeing grazing intensification (Su et al. 2005). Overgrazing is an important driver of widespread declines in biodiversity, ecosystem functioning, and services in the arid and semi-arid grasslands of China (Bai et al. 2007; Deng et al. 2014; Li et al. 2017b; Su et al. 2005). Overgrazing decreases plant species diversity and productivity (Bai et al. 2007), reduces the C, N and P pools in above-ground biomass, and alters C:N:P stoichiometry of steppe ecosystems (Bai et al. 2012); results in soil compaction through trampling and reducing soil infiltration rate, and enhances topsoil exposed to water and wind erosion (Li et al. 2015a, 2017b). A synthesis analysis based on 61 studies from 88 independent research sites within the Qinghai-Tibetan Plateau showed that livestock grazing significantly increased plant species diversity, but decreased aboveground biomass by 47.15%, soil organic carbon by 12.41% and soil total nitrogen by 12.75% (Lu et al. 2017). To mitigate the negative impacts of climate change in the arid and semi-arid grasslands of China, reducing the stocking rate is essential, particularly to sustain native steppe biodiversity, and conserve ecosystem functioning (Bai et al. 2012).

Fencing is widely used as a restoration and management practice in grassland ecosystems worldwide (Deng et al. 2014; Wu et al. 2009, 2010). Fencing improves soil quality by increasing soil organic carbon, soil total nitrogen, the soil C:P ratio and N:P ratio within the 0–100 cm soil profile, and increases vegetation coverage, biomass, and plant diversity (Deng et al. 2014; Wang et al. 2014b). Fencing grassland with grazing exclusion decreased bulk density, pH, and forbs (Wang et al. 2014b). 8-year grazing exclusion significantly affected C pools but had no significant influence on the soil N pool (Wang et al. 2014b). More attention should be given to identifying the main soil and plant characteristics that drive C and N dynamics after grazing exclusion (Wang et al. 2014b). The effects of grazing management are influenced by local environmental factors such as climate, elevation, slope, and water availability (Gao et al. 2010; Lu et al. 2017).

12.4.3 Desertification

China's drylands are seriously threatened by desertification (Qi et al. 2012), leading to declines in ecosystem functions and services (Prăvălie 2016). Desertification is the outcome of coupled processes which primarily result from climate variation exacerbated by human activities (Chi et al. 2019; Liu et al. 2008; Wang et al. 2008). This section describes the key processes of desertification including wind erosion, water erosion, salinization, freeze–thaw erosion, and rocky desertification (Fig. 12.14).

Wind erosion is a key driver of desertification in global drylands (Poesen 2018; Shi et al. 2004a). China's drylands affected by wind erosion are mostly in the country's northwestern and northern arid and hyper arid regions, where the majority of wind erosion intensity is characterized by strong, extremely strong, and dramatic magnitudes. Wind erosion results in loss of soil nutrients (Wang et al. 2006b; Yan et al. 2005) and reduction in NPP and the provisioning services of croplands, grasslands, and forests (Zhao et al. 2017). Wind erosion impacts the lives of 200 million people going as far back as half a century (Wang et al. 2010). Sand and dust storms caused by wind erosion have adverse impacts on air quality, public health, safety of transportation, communication, and irrigation infrastructure, and have significant impacts on the economy (Jiang et al. 2018; Shen et al. 2018; Wang et al. 2016b). Wind erosion dynamics are driven by a combination of climatic factors (i.e., global atmospheric circulation, wind speed) (Jiang et al. 2018; Zhang et al. 2018), soil properties (i.e., surface roughness and erodibility) (Chi et al. 2019), and human activities (i.e., land use/cover change) that leave the soil more exposed (Zhao et al. 2017).

Water erosion and alluvial processes are important drivers of desertification in the semi-arid and dry sub-humid regions of China, with 1.39 million km^2 and 0.80

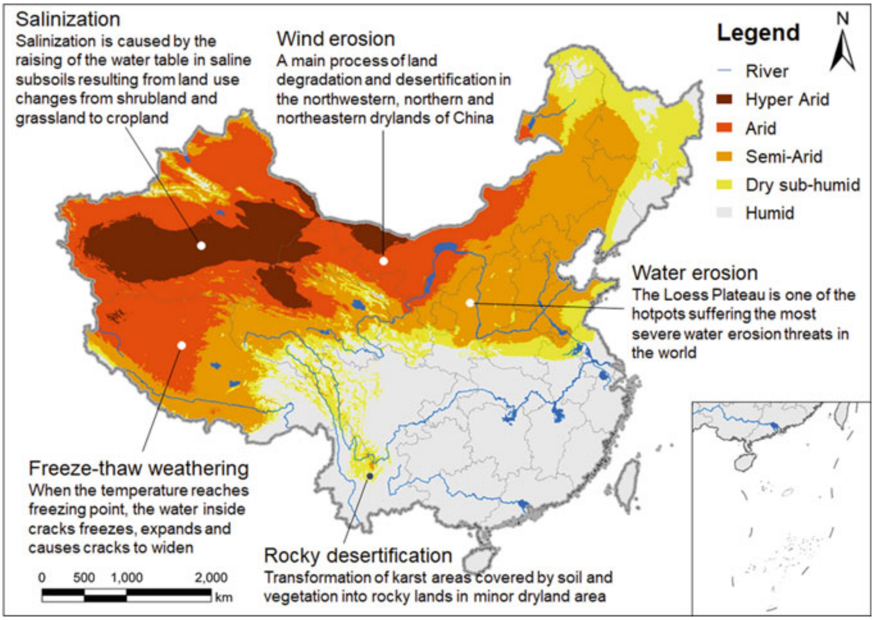

Fig. 12.14 The major external forces that cause desertification in drylands of China. Generally, desertification is caused by wind erosion and aeolian processes; water erosion and alluvial processes; freezing and thawing processes on cold plateaus; soil salinization and alkalization processes; and rocky desertification in the dry-sub-humid karst areas

million km^2 affected by water erosion, respectively. In particular, the Loess Plateau in the arid and semi-arid regions is one of the hotspots suffering the most severe water-erosion-induced soil erosion problems in the world (Fu et al. 2017; Morgan 2009; Shi and Shao 2000; Wang et al. 2016a). Rainsplash, runoff energy and gravity are the three main active agents in water erosion processes such as splash erosion, interrill erosion, rill erosion, and gully erosion (Li et al. 2018b, c; Li and Pan 2020). Piping is a common subsurface erosion process in the semi-arid loess hilly and gullied regions of North China. Pipes are efficient pathways for water, sediment, and carbon transport, and have the potential to initiate or affect development of gullies through roof collapse or channel extension (Li et al. 2018b; Poesen 2018). Both process-based and empirical soil erosion models have been used previously in the arid and semi-arid Loess Plateau of China to understand these processes (see Li et al. (2017a) for a detailed review). However, it is still a challenge to understand how basic water erosion processes (gully erosion, pipe erosion) function and how the various erosion agents (e.g., rainsplash, runoff, gravity) interact.

Frost weathering is commonplace in the cooler high altitude climates of drylands on the Qinghai-Tibet Plateau (Cheng and Wu 2007). Frost weathering is important in producing eroding soil particles (Li et al. 2018a, b), enhancing heat exchange between the atmosphere and the soil surface and influencing the local and regional climate (Cheng and Wu 2007), and affecting surface and subsurface hydrological processes (Li et al. 2018a). Permafrost degradation could result in desertification and ecosystem deterioration on the Qinghai-Tibet Plateau. Changes in the active layer and permafrost conditions under climate warming scenarios are likely to increase emissions of major greenhouse gases (e.g., carbon dioxide and methane) stored in frozen soils (Cheng and Wu 2007; Yang et al. 2010; Zhang et al. 2020b). Despite the important role of frost weathering in changing carbon pools and fluxes on the Qinghai-Tibet Plateau, very little research has attempted to quantify the effects on carbon dynamics and the underlying hydrological processes (Yang et al. 2010).

Other processes such as salinization and alkalization could enhance desertification in drylands. Salinization affects approximately 0.17 million km^2 of the arid and semi-arid regions where the surface soil is rich in sodium chloride and sulfate (>0.3%) (Arndt et al. 2004). Salinization has negative impacts on land productivity since high pH and salinity, and low nutrient levels, restrict plant growth. Rocky desertification is widely distributed in the southwest karst drylands (Jiang et al. 2014; Tong et al. 2018). Rocky desertification is caused by erosion of the thin soil layer (mostly <10 cm) and is induced by increasing human exploitation of natural resources, which has particularly taken place during the past half century (Jiang et al. 2014). However, rocky desertification is not a major land degradation issue in China's drylands due to the relatively minor land area that it affects (<1% of the total drylands).

12.4.4 Interactions Among Different Drivers

Abiotic factors and biotic attributes of the ecosystem modify and are modified by each other, and ultimately change ecosystem multifunctionality (Fig. 12.15). Large rain pulses greater than a threshold of between 10 and 25 mm are capable of improving carbon sequestration capacity in the semi-arid steppe of northern China (Chen et al. 2009). Plant structures modulate abiotic properties through biotic-abiotic feedbacks (e.g., evapotranspiration) and associated hydrological responses (e.g., runoff, infiltration). Vegetated and bare surface patches determine whether and how patches interact, and affect the downslope routing of water, sediments and nutrients (Li et al. 2008). Additionally, vegetation patches affect runoff and erosion processes on a hillslope, and the spatial organization of bare and vegetated surfaces (e.g., size, length and connectivity of bare areas), which determines the operating processes at the hillslope scale.

Wind, water, and freeze–thaw weathering are three major agents of desertification. In addition, the three erosion agents usually occur simultaneously and interact strongly with each other. For example, the effects of rainsplash and overland flow on soil erosion (soil particle detachment and available material transport) largely depend on antecedent conditions, including frost weathering which is important in increasing soil erodibility (Li et al. 2018b). Climate change (e.g., warming, CO_2 elevation), human activities (e.g., cropland and settlement expansion, and overgrazing by livestock), and their interaction are key in initiating desertification in China's drylands (Wang et al. 2006a).

Due to the fast-than-average warming rates and growing human consumption of resources, China's dryland socio-ecological systems may experience systemic and non-linear changes (Fu et al. 2021; Lian et al. 2021), particularly in the semi-arid regions (Berdugo et al. 2020). These changes will negatively affect the key ecosystem services provided by drylands as well as the livelihoods of the substantial human population living in those areas. 2 °C warming has greater negative effects

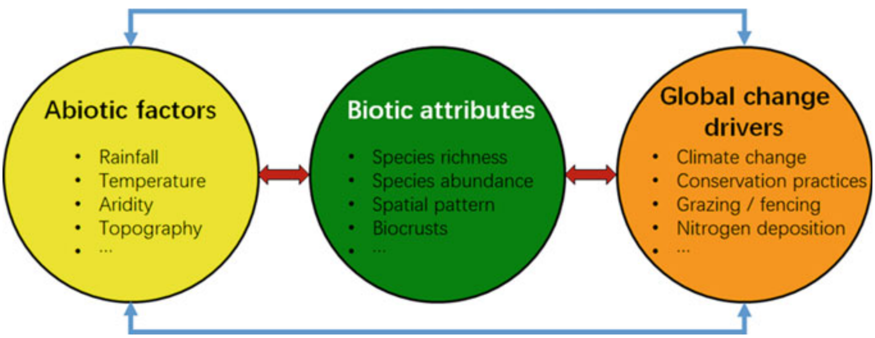

Fig. 12.15 Conceptual framework showing the relationships and feedbacks among abiotic factors, biotic attributes, ecosystem processes, ecosystem functioning, and global environmental change drivers in drylands of China

on ANPP in the arid and semi-arid grasslands than the dry sub-humid grasslands (Cheng et al. 2018). Climate projections point to a greater risk of extreme events (e.g., rainstorms and droughts) and aridification in the arid and semi-arid regions of China (Fu et al. 2008). Decreasing precipitation and increasing temperatures enhance soil drying, making soil suction increase, and available soil moisture for plant root uptake less accessible (Huang et al. 2017a). This soil moisture–temperature positive feedback leads to decreased evapotranspiration and increased sensible heat flux and temperature, a completely dry soil layer and desertification (Seneviratne et al. 2010). Expansion of drylands will increase the risk of water scarcity, land degradation, and declines in human wellbeing (Fu et al. 2021; Li et al. 2015b; Yao et al. 2020). While we see significant expansion in the drylands of northern China (Li et al. 2015b), there is conflicting evidence showing that China's drylands will shrink under future 1.5 and 2.0 °C warming scenarios when using runoff and leaf area index (LAI) to delineate drylands instead of the AI (Zhang et al. 2020a). It is thus unclear that the country's dryland boundaries will expand overall under climate change.

12.5 Ecological Management

12.5.1 Payments for Ecosystem Services

Payments for Ecosystem Services (PES) have been widely used as an effective tool for ecological conservation and restoration without restricting socioeconomic development (Salzman et al. 2018; Yang et al. 2013). China leads in its investment in global government-financed PES programs, implementing PES strategies at a scale and speed simply not possible in other countries (Salzman et al. 2018). During the last four decades, there has been a substantial increase in PES programs in China's drylands (Bryan et al. 2018). Illustrative PES programs include the Grain to Green Program (regarded as the world's largest PES program in terms of investment and area coverage), and the Natural Forest Conservation Program, focusing on logging bans and afforestation (Liu et al. 2008; Salzman et al. 2018). Many previous studies have reported the ecological and socioeconomic outcomes of PES programs. For example, Wu et al. (2019) used a framework that linked the Grain to Green Program, livelihood activities, and socioeconomic outcomes, to investigate how the Grain to Green Program affected the incomes of local households in the Yanhe watershed of the Loess Plateau. Wu et al. (2019) selected five livelihood activities, including crop production, orchard fruit production, non-farm work, labor migration, and greenhouse-grown vegetable production. 'Non-payment income' was selected as an indicator of the socioeconomic outcome, to represent income from sources other than payments from the Grain to Green Program. Several hypothesis was proposed including: (i) all the five livelihood activities are able to increase non-payment incomes; (ii) the Grain to Green Program is to convert steep croplands to forest and grassland, which has a negative impact on agricultural production; but positively affects orchard fruit production due to the increase in area of orchard fruit plantation; (iii) the Grain to Green Program has a positive impact on participation in non-farm work and labor migration in the household (Liu et al. 2008; Yin et al. 2014). In addition, it was hypothesized that different livelihood activities interact with each other. For instance, labor migration has greater earnings than local non-farm work and creates more job vacancies for local non-farm workers. Both labor migration and non-farm work have negative effect on crop production. Due to the limited labor in a household, the five livelihood activities were negatively correlated to each other. Wu et al. (2019) found that the implementation of the Grain to Green Program significantly increased participation in local non-farm jobs and household incomes. They suggested several ways to improve the socioeconomic outcomes by increasing non-farm work benefits and reducing the reliance of households on income from crop production.

12.5.2 Efforts to Combat Desertification

To combat desertification, China has implemented a wide range of large-scale land conservation and restoration programs (Fig. 12.16) in drylands (Bryan et al. 2018; Ouyang et al. 2016). Detailed descriptions were provided in Bryan et al. (2018), Li et al. (2021) and Kong et al. (2021). The Natural Forest Conservation Program and Grain for Green Program are two of the biggest programs offering PES in China and worldwide in terms of scale, payment, and duration (Liu et al. 2008). These ecological restoration projects have changed land-use patterns and exerted a significant influence on dryland ecosystems (Bryan et al. 2018; Cheng et al. 2018; Lu et al. 2018).

Ecological conservation and restoration projects have resulted in vegetation greening (Chen et al. 2019; Piao et al. 2020), reduced soil erosion and land degradation (Piao et al. 2020; Zhu et al. 2016), and enhanced ecosystem services through soil conservation and carbon sequestration (Lu et al. 2018; Tong et al. 2018). However, in many afforestation areas, large-scale plantations have experienced high mortality due to a lack of understanding of the suitability of planted species to the local environment and soil desiccation in the deep soil layer caused by over-planting (Cao 2008; Feng et al. 2016). Although some positive outcomes have been achieved over the last two decades, large uncertainties remain regarding long-term policy effects on the sustainability of the performance of the ecological conservation and restoration programs. Future research is needed to further explore the dynamic interactions between people and their living environments in a changing world (Lü et al. 2012).

China is a developing country that suffers from long-term and large-scale desertification in its drylands, and the country's efforts to combat desertification produce many best management practices. For example, the State Forestry Administration of China has established the national desertification monitoring system and the China Desert Ecosystem Research Network (CDERN), to strengthen monitoring and research in desert regions (Wang et al. 2013). The CDERN network has 43 research stations across the arid, semi-arid, and dry sub-humid areas in North China, providing long-term observations and scientific demonstrations for the prevention and control

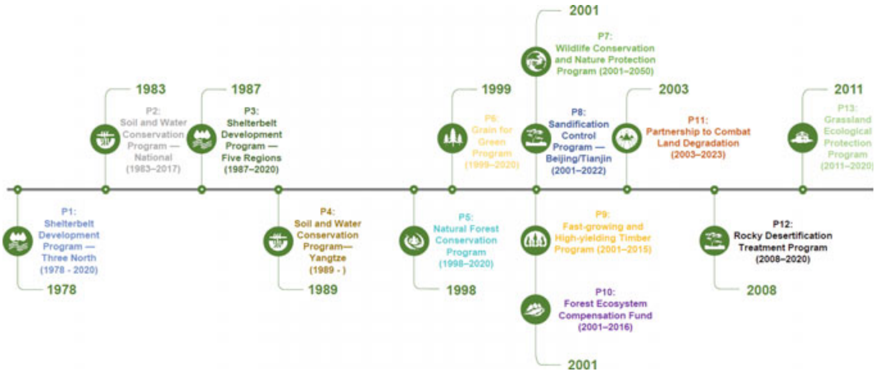

Fig. 12.16 The 13 major programs include: the Shelterbelt Development Program—Three North (China's Great Green Wall) (1978–2050); Soil and Water Conservation Program—National (1983–2017); Shelterbelt Development Program—Five Regions (1987–2020); Soil and Water Conservation Program—Yangtze (1989–indefinite); Natural Forest Conservation Program (1998–2020); Grain for Green Program (1999–2020); Wildlife Conservation and Nature Protection Program (2001–2050); Sandification Control Program—Beijing/Tianjin (2001–2022); Fast-growing and High-yielding Timber Program (2001–2015); Forest Ecosystem Compensation Fund (2001–2016); Partnership to Combat Land Degradation (2003–2023); Rocky Desertification Treatment Program (2008–2020); and Grassland Ecological Protection Program (2011–2020)

of desertification and regional economic development. In addition, a series of standardization of desertification control technology and ecological protection measures have been approved by the National Standardization Technical Committee, including Technical specification for sand control, Closing (sand) technical specification for afforestation, Technical specification for oasis protection forest system construction (Bao et al. 2017). The normally used desertification control technology includes integrating a series of effective sand-stabilizing methods, selecting drought-tolerant sand-fixing plants, and promoting the fast recovery of vegetation through technology. China's experience and lessons could be important for other developing countries in order to combat desertification and to improve livelihood of residents (Ci and Yang 2010).

12.6 Summary and Perspectives

Biotic and abiotic interactions through space and time are vital in determining vegetation dynamics and shaping ecosystem responses in China's drylands. The key processes of desertification, including wind erosion, water erosion, salinization, freeze–thaw erosion, and rocky desertification, hamper the ability of China's drylands to provide ecosystem goods and services. Expected increases in aridity will nevertheless negatively impact ecosystem structure and functioning in the drylands of China, even if there is no clear evidence that the country's dryland boundaries will expand overall under climate change when using runoff and LAI to define drylands. Large-scale ecological restoration projects enhance the greening of China's drylands, but also impose considerable pressure on these water-limited environments. The effectiveness of the restoration projects should be evaluated in a comprehensive way.

To unravel the complex and dynamic mechanisms of dryland structure and functioning, much work remains to be done on understanding the interactions between biotic attributes and abiotic factors, the two-way interactions between and interdependence of ecological and hydrological processes, and key desertification processes. Integrated research is needed based on multiple spatial–temporal scale observations alongside multidisciplinary studies.

This chapter is of major importance in improving our understanding of China's drylands where a large proportion of the human population directly depends on ecosystem services from these environments. Due to their wide distribution and unique features, improved and synthesized knowledge about China's drylands also contributes to the general understanding of how terrestrial ecosystems function and respond to ongoing global environmental changes in drylands around the world.

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