

A review of environmental droughts: Increased risk under global warming?

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Abstract This article reviews current drought effects on environmental systems. It stresses the need for considering environmental drought as a relevant type to be included in drought classifications. Here we illustrate that drought has complex environmental effects and affects many different systems (e.g., soils, air, vegetation, and forests, aquatic systems and wildlife). Droughts can affect the quality, structure, and diversity of these systems. However, we find that most environmental systems show strong resistance and resilience to drought events, and the effects of drought are usually temporary. Structural effects of environmental droughts tend to only occur in areas that are perturbed or in communities near their distribution limits. There are few long-term experimental studies that quantify possible trends in drought effects on environmental systems. Nevertheless, existing studies of forests that are based on tree-ring chronologies or forest inventories indicate increased drought-related effects on environmental systems. Future climate change scenarios suggest increased drought severity worldwide, which could alter the vulnerability of different environmental systems and increase the number of structural drought effects.

Key-words soil, aquatic systems, wildlife, desertification, forest fires.

1. Introduction

Drought is one of the most complex hydroclimatic hazards affecting the world since it is very difficult to quantify its severity (Wilhite and Pulwarty, 2017), and to assess its effects given the large number of systems affected (Wilhite, 2000) and academic disciplines involved (Dracup et al., 1980). The challenge of defining and measuring drought severity (Wilhite and Glantz, 1985) makes it difficult to determine the beginning and end of a drought and the spatial extent of the event (Burton et al., 1978; Mishra and Singh, 2010). It is also challenging to define because of the complexity of the spatial and temporal drought propagation processes (Bonsal et al., 2011; Parry et al., 2016; Quintana-Seguí et al., 2018; Ahmadi and Moradkhani, 2019). Drought is an effect-dependent phenomenon and the diversity of drought effects makes it practically impossible to establish a universal drought definition (Lloyd-Hughes, 2014). Therefore, it is necessary to define different drought types based on the sector and system affected. Classical drought definitions are divided into four types: (i) meteorological, (ii) agricultural, (iii) hydrological, and (iv) socioeconomic (Wilhite and Glantz, 1985). Usually these four drought types are closely related to each other given how drought effects propagate through the hydrological cycle (Changnon and Easterling, 1989).

Drought effects are commonly related to the agriculture and water resource sectors. They may cause substantial economic losses in the agriculture sector of developed countries through reductions in crop yield or total failure of crops (Sweet et al., 2017; Tian et al., 2018). In extreme cases, they can also cause human migration and famine in developing countries (Gray and Mueller, 2012; Grolle, 2015). Hydrological droughts may also cause significant effects to irrigated agricultural systems (Maestro et al., 2016; Vidal-Macua et al., 2018) and problems for urban water supply, industrial needs, reductions of hydropower production, etc. (Balling and Gober, 2007; Jerez et al., 2013).

As noted by Crausbay et al. (2017), the traditional classification of drought into four types does not adequately address the ecological dimensions of drought and tends to be too

human-centric. They stressed the importance of also considering the ecological dimensions of drought. Droughts can have a profound effect on ecosystems; in some cases the effects are temporary, but in others they are permanent or structural (Figure 1). The term permanent is really a matter of timescale. Here we refer to those situations in which systems do not recover to the initial state in a reasonable period of time after climatic conditions recover to normal. The term “permanent” refers to very long recovery periods or cases when systems reach a tipping point from which recovery to the initial conditions is not possible.

Crausbay et al. (2017) define ecological drought “as an episodic deficit in water availability that drives ecosystems beyond thresholds of vulnerability, impacts ecosystem services, and triggers feedbacks in natural and/or human systems”. Since the deficit in available water is defined based on the existing demand in a specific environmental system and region, it is relative. This deficit is usually driven by climate variability processes such as the occurrence of periods with precipitation amounts below the normal or enhanced atmospheric evaporative demand (AED). We adopt this definition in our paper, but prefer the term *environmental drought* since we believe it better represents the coupled nature of human-environment interactions that are central to this type of drought. The environment therefore reflects the many interactions between microbial fauna, animals, plants, soil characteristics, atmosphere, water, among others, but also human influences. These human influences are a key aspect of understanding many of the effects of drought on a variety of ecosystems (e.g. human influences can strongly affect desertification processes; they can also affect water and air quality and other processes like forest mortality, given that the human management of these systems strongly modifies natural systems). This is the main reason why we prefer to use “environmental” instead “ecological” droughts since we believe this term provides a more comprehensive and complete representation of the wide variety of effects and interactions, including the human use (and misuse) of the environment.

Droughts have noticeable effects on vegetation, air, soil and freshwater quality, and the fauna of terrestrial and aquatic ecosystems. In addition, drought conditions enhance the likelihood of forest fires and land degradation processes (e.g., soil erosion). Environmental droughts may trigger episodes of forest mortality (Allen et al., 2010), cause declines in tree growth (Linares et al., 2010), reduce net primary production (Zhao and Running, 2010), cause pasture loss (Zhao et al., 2018), or even alter the biological diversity (e.g., composition and structure) of vegetation communities (Hanke et al., 2016). Droughts can also alter soil composition (van der Molen et al., 2011) and edaphic biodiversity (von Rein et al., 2016), since drier soils may reduce the soil biota, diminish the soil carbon storage and degrade the soil structure (Robinson et al., 2016).

If drought is accompanied by wildfires, they may also contribute to further soil degradation. The degradation of vegetation and soil favors erosion processes (Dardel et al., 2014), which may trigger desertification, especially when droughts are persistent. Obviously, a decrease in vegetation diversity, habitat degradation and the lack of water and food that is caused by droughts can result in a reduction in the density and diversity of the wildlife due to the death or migration (Sinclair et al., 2007).

Environmental droughts are closely related to other drought types. For example, it is not possible to consider the hydrological and environmental dimensions of drought independently since soil hydrology strongly affects vegetation establishment and growth (Vicente-Serrano et al., 2015b). If vegetation is affected by drought (e.g., triggering forest mortality or reducing biomass), it can alter hydrological processes (e.g., rainfall interception, percolation, soil infiltration, runoff) (Brown et al., 2005) and influence the availability of surface and subsurface water resources. Environmental droughts are also strongly linked with agriculture and socioeconomic droughts in a variety of different ways. Ecosystems directly generate products with economic value (e.g., wood, mushrooms, pasture) and so losses/reductions directly cause economic losses. Ecosystems also provide services (Costanza

et al., 1997), which are essential for the development of tourism activities, the preservation of water quality, and human health. Therefore, environmental drought is connected directly and indirectly to the other four classical drought types (meteorological, hydrological, agricultural, and socioeconomic). The addition of environmental drought provides a more holistic means of categorizing drought effects that are specifically related to ecosystems and human-ecological interactions.

Environmental droughts are particularly relevant given the changing climate which is likely to increase the severity of drought events given changes in precipitation and/or increases in the atmospheric evaporative demand (Hoerling et al., 2012; Vicente-Serrano et al., 2014b). Thus, in the past two decades extreme drought events have occurred more frequently in some regions, with no previous precedents in the long-term observations (Barriopedro et al., 2011; van Dijk et al., 2013). Although some uncertainties exist, future climate projections tend to agree on increased drought severity (Naumann et al., 2018); a fact that could increase the frequency and severity of environmental drought effects (see section 4).

This study reviews: (i) environmental effects of droughts, including different systems and the interactions among them, stressing the relevance of the term of environmental drought to refer to a specific drought type, and (ii) the possible signs of change in environmental droughts due to climate change.

2. Environmental effects of drought

2.1 Soil quality

One of the primary effects of drought is a reduction in soil water content (Holsten et al., 2009; Otkin et al., 2016). The rate of soil moisture depletion varies as a function of the climate characteristics, soil type and depth, vegetation coverage, and drought severity (McColl et al., 2017; Tang and Piechota, 2009). Reductions in soil moisture can significantly alter the soil carbon cycle, reduce carbon storage, and affect soil biota (Evans and Burke, 2013; Sardans

and Penuelas, 2013). If droughts occur concurrently with other processes (e.g., overgrazing, poor land management practices, wind erosion), they can cause land degradation and geochemical alteration of the soils (Lal, 2003). Nevertheless, these extreme responses are usually restricted to overexploited areas (see below), and the common response of soil quality to drought in undisturbed soils is usually temporary. Once soil moisture levels return to normal, soil biochemical processes, carbon storage, and soil biota will recover (Goransson et al., 2013; Rousk et al., 2013). This drought recovery process has been observed in soils in both arid and humid environments, including soils with different biochemical properties and depths.

Droughts can alter soil-atmosphere exchanges. In general, droughts reduce soil respiration since they tend to reduce the bacterial and fungal populations that contribute to carbon cycling and the breakdown of organic matter. Several studies have demonstrated that soil microbial and fungal communities respond rapidly to soil moisture changes (Bell et al., 2014; Cregger et al., 2012; Goransson et al., 2013; Toberman et al., 2008; von Rein et al., 2016), including changes in composition of the microbial communities (Cantarel et al., 2012). Cregger et al. (2012) carried out a large-scale precipitation manipulation experiment in a semiarid coniferous forest and found that changes in soil moisture were the main driver of the composition of soil microbial communities. Bacterial and fungal communities were compositionally different in dry and wet periods and the drought plots had a fungal abundance that was 4.7 times lower than wet plots. This resulted in a decrease in microbial and fungal degradation of the soil carbon. Similar results were obtained by Bastida et al. (2017). They analyzed the responses of the microbial community to a drought induced by means of a rainfall exclusion experiment in a semi-arid *Pinus halepensis* forest. They showed that microbial biomass decreased under drought conditions and the diversity of bacterial types was affected at the genus level. Moreover, the active microbial community was the most sensitive to drought, affecting the cycling of organic matter.

Although drought affects soil microbial populations, several studies have shown that recovery of microbial biomass after severe drought events usually occurs rapidly after soil moisture recovers to normal levels, even in semiarid regions (Hueso et al., 2011; Rousk et al., 2013). Göransson et al. (2013) showed that bacterial growth in soils affected by drought is slower than in moist soils, but in non-overgrazed soils that are not affected by state changes in vegetation and corresponding shifts in microbial communities, dry soils usually recover quickly after precipitation occurs. Thus, in a controlled experiment they did not find differences in the cumulative bacterial growth between dry and control soils one week after rewetting. Hueso et al. (2011) indicated that short- to medium-duration drought events are insufficient to destroy the native microbial biomass. They found that even soils in arid regions show a rapid recovery in microbiological and biochemical parameters after soil rewetting. Therefore, drought effects on soil microbial communities are usually temporary, unless other factors coexist that cause a permanent alteration of soil properties (e.g., erosion, human activities).

Soil microbial activity is closely related to soil respiration and experimental studies have examined how soil moisture affects soil respiration (Davidson et al., 1998; Gaumont-Guay et al., 2006; Harper et al., 2005; Oishi et al., 2013; Selsted et al., 2012; Suseela et al., 2012). The majority of these experimental studies have shown that drought reduces soil respiration. For example, Selsted et al. (2012) conducted a precipitation exclusion experiment in a Danish Calluna-Deschampsia-heathland and showed that soil respiration can be reduced by 50% as a response to drought. Suseela et al. (2012) examined how drought affected respiration at the Boston-Area Climate Experiment and found that heterotrophic respiration decreased when soil water content was below 15%. Nevertheless, drought effects on soil respiration can be complex, since soil respiration is reduced when soils are dry, but increases rapidly after soil rewetting (Borken and Matzner, 2009; Davidson et al., 1993). This occurs because carbon inputs (production) decrease more than outputs (respiration) (Hoover et al., 2016).

Davidson et al. (1993) analyzed soil emissions of NO and N₂O in dry tropical forests of Mexico and, as expected, they found a reduction during dry periods. They also showed that precipitation in dry soils caused pulses in NO and N₂O emissions driven by denitrification processes that are controlled by soil moisture. Similar results have been observed by Hoover et al. (2016) in a mesic grassland. They showed that after two years of extreme drought conditions, soil respiration was reduced by 25%; however, when precipitation returned, the drought plots showed higher rates of soil respiration than the control plots (up to +17%). Drought effects on soil respiration are also dependent on a variety of other factors, including climate. For example, Jensen et al. (2003) analyzed the effects of drought on the soil biochemical processes in two heathland ecosystems at contrasting water inputs (750 mm and 1700 mm per year) and showed that drought conditions at the drier site led to reduced microbial activity, soil carbon, and nitrogen levels. While at the wetter site there was an increase in microbial activity during the drought and below-ground CO₂ emissions increased by 22%.

Drought also affects the carbon and nitrogen mineralization and nutrient cycling (Fierer et al., 2003; Larsen et al., 2011; Sardans et al., 2008; van der Molen et al., 2011). Larsen et al. (2011) exposed a seminatural Danish heathland ecosystem to extended summer drought and showed that dry soils reduced belowground gross nitrogen mineralization, affecting the nitrogen turnover and reducing the nutrient availability for plant growth. Bell et al. (2014) conducted a 7-year experiment in a Chihuahuan desert grassland to determine the influence of precipitation variability on decomposition and nutrient cycling. They showed a two-year delay in the microbial response to increased precipitation. This demonstrates that the recovery processes in dryland areas can be much slower than in humid environments. Complete recovery (e.g., high microbial biomass and soil enzyme C and P acquisition) did not occur until 3-5 years after the drought ended. Nevertheless, if given enough time, soil biota and biochemical processes are quite resilient after drought events, even in harsh environments.

Given the resilience of soil processes to drought, structural effects on soil quality usually only occur if there are other stressors to the system. For example, these stressors include the loss of vegetation cover, affecting nutrient cycling and the accumulation of organic matter (Reynolds et al., 1999), the occurrence of intense forest fires (Mataix-Solera et al., 2011; Shakesby et al., 1993), and soil erosion (Garcia-Ruiz et al., 2017). The interactions among all these processes (Allen, 2007; Sardans and Penuelas, 2013) and land management (e.g., livestock pressure (Illius and O'Connor, 1999)) may also act as an additional stressor (Ravi et al., 2010). For example, Mangan et al. (2004) showed in the Nebraska Sand Hills that fire and grazing alone have little effect on vegetation cover, but combined with drought, they caused biomass to decrease and erosion rates to significantly increase in these low organic matter soils. Wei et al. (2010) also showed in a loess area of China that major erosion episodes are associated with dry years since drought reduces vegetation cover. Soil is then more affected by the occurrence of intense rainfall events, since the majority of annual soil loss is due to a few intense rainfall events (Gonzalez-Hidalgo et al., 2007). Allen et al. (2011) studied sediment production during a drought that affected the south-central U.S. between 1947 and 1956. They showed that rainfall intensity during the drought was greater than during the non-drought period. Therefore, sediment yields were larger during the drought. This illustrates that drought effects on soil quality are complex and depend on many environmental and human factors, as well as precipitation intensity and rainfall-runoff processes.

In summary, existing studies of drought effects on soil quality suggest that droughts have a significant influence on soil respiration and geochemical conditions. Nevertheless, there is a high capacity for recovery once climate conditions return to normal, although the rates of recovery are different between humid and dry regions. Permanent drought effects on soil quality are not expected unless other stressors co-exist with drought conditions.

2.2. Air quality

Drought is also considered an important driver of air quality, affecting both atmospheric dust concentrations and atmospheric chemistry. This can occur directly by affecting atmospheric chemistry, or indirectly given the strong relationship between drought and forest fires. Here we must make a distinction between the role of short-term atmospheric stability and the occurrence of long-term drought episodes. Several studies have demonstrated that urban air quality issues are often associated with a persistent high-pressure system that leads to weak atmospheric circulation and descending air (e.g., Beer (2001); Juneng et al. (2011)). The presence of high-pressure systems is also associated with dry weather, but it does not necessarily result in drought conditions since these conditions must persist for an extended period of time to cause a drought. Therefore, the focus of this section is not on the linkages between high pressure and air quality, but on the effect of persistent periods of low precipitation and/or high atmospheric evaporative demand that cause drought events.

First, there is a clear connection between dust concentrations and drought since dust storms are related to both drought severity and poor land management practices (Goudie and Middleton, 1992; McTainsh et al., 1998; Reheis, 2006). The dust bowl that affected the United States in the 1930s is an excellent example of how drought can affect air quality (Cook et al., 2014a; McLeman et al., 2014; Schubert et al., 2004). Poor land management practices coupled with a prolonged period of drought from 1932 to 1939, resulted in exceptional wind erosion of agricultural lands, frequent dust storms, and high concentrations of soil particles suspended in the air. Nevertheless, dust storms have also affected other regions. For example, the largest dust storm to ever occur in Australia took place on the 23rd October 2002 and coincided with the Millennium drought (McTainsh et al., 2005).

Drought can also affect air quality by influencing atmospheric chemistry through direct and indirect natural processes. One example of how drought influences air quality biogeochemical processes is tropospheric ozone (Abeleira and Farmer, 2017; Huang et al.,

2016; Ramsey et al., 2014), which is a pollutant that results from a chain of reactions involving carbon monoxide and other hydrocarbons and which has substantial health effects.. Lin et al. (2017) analyzed the concentrations of surface ozone over the United States between 1980 and 2014 and showed that tropospheric ozone concentrations were higher than normal during the severe drought event of 1988, and Wang et al. (2017) found increases in surface ozone (8%) and the concentration of particles smaller than 2.5 microns (PM_{2.5}) (17%) associated with drought events in the growing season (March-October). In general, these pollutants are related to the joint effects of drought on deposition, natural emissions (wildfires, volatile organic compounds and dust), and on atmospheric chemistry. In the summer of 2003, Europe experienced one of the most extreme drought events ever recorded. This drought led to significant decreases in vegetation respiration and activity (Ciais et al., 2005) and it suppressed soil biochemical flows to the atmosphere (Granier et al., 2007). Solberg et al. (2008) showed that during this drought, surface ozone in central Europe reached its highest level since the end of the 1980s. Record levels for the maximum hourly ozone concentrations were set at many sites in central Europe. A number of positive feedbacks contributed to these elevated ozone concentrations such as persistence in the atmospheric boundary layer, and forest fires in the Iberian Peninsula. Moreover, a reduction in dry deposition and elevated air temperatures also contributed to the increased ozone concentrations.

Drought also indirectly affects air quality because droughts are associated with more forest fire activity (Field et al., 2004; Tressol et al., 2008; Vicente et al., 2017). Forest fires emit large quantities of particulate matter into the atmosphere. This degrades air quality and affects human health. There are several examples of the connection between drought-related fires and increased air pollution. Khandekar et al. (2000) illustrated how the 1998 El Niño caused extreme drought over Indonesia characterized by forest fires in Borneo that affected over one million acres. Wang et al. (2015) analyzed the effect of the 2011 Southern U.S.

drought on fine aerosol (PM_{2.5}) concentrations in summertime. They showed that PM_{2.5} increased an average of 26% because of the drought event, which was related to increased organic carbon (120%) from wildfires. Saylor et al. (2015) showed a connection between concentrations of ammonia and PM_{2.5} ammonium due to increases in wildfire activity across the southeastern United States caused by elevated temperatures and widespread drought. In Europe, there are multiple examples of this behavior. Martins et al. (2012) analyzed the effect of forest fires on aerosols and ozone in Portugal during the severe fires that affected the country between 2003 and 2005 and found increases of approximately 20% under fire conditions compared to non-fire conditions.

The major drought that occurred in central Russia in the summer 2010 was associated with a strong heat wave (Barriopedro et al., 2011). This drought caused widespread forest fires in the region and resulted in elevated concentrations of CO, ozone, and particulate matter below 10 μm (PM₁₀) (Konovalov et al., 2011). This event also illustrated the complex interactions between drought and forest fires. In late June, severe aerosol pollution was caused by air masses that originated in drought-afflicted regions in southern Russia, but by August it was the advection of smoke from forest fires that caused the elevated surface concentration of pollutants (Kuznetsova, 2012). Thus, interactions of warmer than normal conditions and air pollution caused by wildfires contributed to more than 2000 deaths (Shaposhnikov et al., 2014).

The effect of drought on air quality can be spatially complex since drought conditions in one region can influence air quality in other regions. Peppler et al. (2000) showed, for example, that in 1998, large areas of Central America and Mexico were affected by wildland fires-driven by drought conditions. Winds transported smoke over the Gulf of Mexico and visibilities were greatly reduced and pollutants increased in the Southeastern U.S. Similarly, Koe et al. (2001) found that the drought that occurred in southeast Asia in 1997 caused large forest fires. The smoke from these fires caused visibility and health problems in a number of

major cities. Although the fires mostly affected the coasts of southeast Sumatra and southwest Kalimantan, they contributed to transboundary air pollution in Singapore and Kuala Lumpur.

In summary, drought influences wind erosion and dust storms, but land management also plays a role. There are few studies that have linked drought with the chemical composition of the air, but they agree that drought contributes to more air pollution through various geochemical processes and forest fires.

2.3. Forest mortality and wildfire

The influence of drought on forest growth and mortality, and wildfires are some of the most commonly studied environmental drought effects. The reason is that drought effects on vegetation can be more easily measured than other environmental drought effects given the availability of remote sensing information (Byer and Jin, 2017; Ji and Peters, 2003; McDowell et al., 2015; Michaelian et al., 2011), the possibility of measuring forest growth by means of dendrochronological records (Sanchez-Salguero et al., 2017; Truettner et al., 2018), and the collection of forest inventories (Klos et al., 2009) and wildfire statistics (Xiao and Zhuang, 2007). There are several reviews that have summarized the complex response of vegetation and forests to drought (Bonal et al., 2016; Choat et al., 2018; Lindner et al., 2014). Allen et al. (2010) reviewed more than 150 studies that document drought-driven tree mortality since 1970. The drought-induced mortality events that they identified spanned a wide range of forest biomes from dry savannas to humid forests, including boreal forests (Chen et al., 2017) and equatorial rainforests (Bonal et al., 2016; Phillips et al., 2009). Some of these events were relatively localized and only resulted in modest increases in mortality rates over the normal background conditions, while other events were associated with regional-scale forest die-off that also involved insect outbreaks. There are many examples of recent forest mortality episodes triggered by drought episodes in different world regions. Michaelian et al. (2011) analyzed forest drought mortality during the episode of 2000-2001 in western North America.

They found large areas affected by tree mortality and the loss of biomass represented about 20% of the total aboveground biomass. Moreover, the spatial patterns of tree mortality closely resembled the patterns of drought severity. Gu et al. (2015) also illustrated widespread forest mortality in Southwest USA as a response to severe drought conditions in 2002.

The response of forest growth and mortality can be complex. It not only depends on forest resistance and adaptation strategies, but it is also highly dependent on the severity of the drought events (Taeger et al., 2013), and the time scale at which drought is measured (Pasho et al., 2011; Vicente-Serrano et al., 2015b). Thus, all biomes, regardless of whether they are located in arid or humid climates, tend to respond to short drought time scales as a consequence of different mechanisms and drought adaptation conditions (Vicente-Serrano et al., 2013). These mechanisms include morphological, phenological, and physiological strategies, which strongly diverge among species (Chaves et al., 2003), but also inter-specific and even individual genetic adaptations (Gaspar et al., 2013). Drought effects also seem to differ as a function of the observed tree growth that occurs before drought onset (Peltier et al., 2016).

The response of physiological and growth processes to drought can also be complex. Gazol et al. (2018) showed in Spain that forest growth is more sensitive than photosynthetic activity to drought. This pattern has been experimentally observed in Scots pines by Aaltonen et al. (2017). They showed that the photosynthetic rate in the drought-affected trees was not different than the control seedlings, but the effects on forest growth were clearly identified in the drought-affected trees in comparison to the control. This pattern reflects a strategy by trees to cope with drought conditions and establish physiological preferences to cope with water-stress conditions. The more spatially extensive mortality events are associated with prolonged water deficits, often lasting multiple years, and were often accompanied by other stressors, such as insects or disease outbreaks. Within the forest stand, the spatial patterns of drought effects and tree mortality are strongly determined by life-history traits and drought

tolerance of each species (Johnson et al., 2018; Pasho et al., 2011; Peltier et al., 2016; Truettner et al., 2018; Vicente-Serrano et al., 2014a). Thus, since each species has a different sensitivity to drought, the occurrence of a drought may cause the selective replacement of some species with more drought tolerant ones (Galiano et al., 2010). Klos et al. (2009) found that coniferous forests showed a noticeable reduction in tree growth during drought events based on 15 years of forest inventories in southeastern U.S. However, they did not find differences in the growth of oak species. Moreover, these authors showed that tree species richness reduced the overall mortality at the stand level. Larger and older trees seem to be more prone to drought-induced mortality (Bennett et al., 2015; McDowell and Allen, 2015), but there is substantial species-to-species variation. In some cases, smaller sub-dominant trees and saplings have been found to have the greatest susceptibility to drought-induced mortality (Camarero et al., 2015).

McDowell et al. (2008) identified different mechanisms, which can be related, to explain drought effects on forest mortality: (1) hydraulic failure, (2) carbon starvation, and (3) increased abundance of biotic agents. The first mechanism seems to have been the most important for explaining recent episodes of forest mortality (Adams et al., 2017b; Anderegg et al., 2015; Anderegg et al., 2016; Choat et al., 2018). However, it remains difficult to diagnose and predict the likelihood of tree survival to a drought event because mortality commonly involves many interacting factors. Drought can trigger mortality in trees that are under stress because of pre-existing conditions such as age, environmental stressors (e.g., poor soils, limiting climate) and the effect of insects and pathogens (Haavik et al., 2015). In addition, forest resilience and recovery rates can be highly variable in different forest conditions and species (Gazol et al., 2018; Taeger et al., 2013). For example, the level of tree defoliation is a good predictor of the capacity for recovery (Guada et al., 2016).

Obviously, drought-induced mortality is an extreme outcome and often heat and water stress cause declines in growth and productivity rather than mortality. For example, Zhao and

Running (2010) showed that there is a strong and direct relationship between interannual variations in total global terrestrial net primary production and growing-season precipitation and the Palmer Drought Severity Index ($r = 0.96$ and 0.76 , respectively). In recent decades, spatially-expansive droughts and the decline in precipitation in the Southern Hemisphere have reduced global terrestrial net primary production (Zhao and Running, 2010). When droughts are associated with other ecosystem disturbances like disease and insects, this also will release carbon to the atmosphere. Nevertheless, the response of vegetation and activity and growth to drought is strongly variable and depends on vegetation types, forest species, but also on the general bioclimatic conditions of each region (Gouveia et al., 2009; Vicente-Serrano, 2007). Thus, different biomes tend to have a distinctive sensitivity and response to drought events (Vicente-Serrano et al., 2013) and the patterns of forest resilience to drought events also strongly vary among forest types (Gazol et al., 2018).

Changes in forest structure and composition have also been observed. For example, after severe and long-lasting droughts, such as those that affected the Sahel, Hänke et al. (2016) showed that the species composition changed once recovery and natural re-forestation began to take place. They demonstrated that despite a return to wetter conditions, the species composition had shifted so that a higher proportion of the species were drought-resistant. This is but one example of how drought-induced mortality and drought stress can cause shifts in the structure and composition of an ecosystem.

There is a complex relationship between human land management and fire history with periods in which wildfire can be determined by fuel limitations whereas others are climate-driven (Pausas and Fernandez-Munoz, 2012). Nevertheless, there is a strong and direct connection between drought and wildfire activity (Crockett and Westerling, 2018; Turco et al., 2017a; Xiao and Zhuang, 2007). Forest fire activity is strongly controlled by drought and aridity during the fire-season. Drought conditions during the fire season result in drier, more flammable fuels due to reduced precipitation, increased temperatures, wind speeds, and

vapor pressure deficits. Therefore, wildfire activity responds strongly to climate variability and drought conditions (Ruffault et al., 2018). Of course, there are confounding factors such as land management and human activity that influence the amount of available fuels and the probability of wildfire ignition. In the western United States, decades of fire suppression have contributed to increased biomass and therefore increased sensitivity to drought conditions. There are also strong positive feedbacks among droughts, heat waves, insect and disease outbreaks and wildfire activity. Severe drought events increase the probability of tree mortality. This produces additional dead fuels and the high temperatures and low relative humidity dry out the existing fuels and provide an environment that is conducive for wildfire ignition.

However, connections between drought and wildfires are sometimes difficult to determine since other factors such as invasive species, fire management, and increasing human populations in the wildland-urban interface are believed to have contributed to these trends. Therefore, it is difficult to isolate the relative importance of climate variability and changes in drought frequency and severity for explaining these trends although there is strong evidence that an extremely dry year will increase forest fire activity (Turco et al., 2017a; Turco et al., 2017b; Williams et al., 2014). In any case, forests in the northern latitudes and equatorial regions report a close connection between drought severity and wildfires. For example, Brando et al. (2014) showed in the Amazonian region that during the droughts of 2007 and 2010, 12% and 5% of the southern Amazon forests were affected by forest fires, respectively, in comparison with <1% in non-drought years.

In summary, existing studies conclusively demonstrate that droughts cause a reduction in forest growth and an increase in forest mortality episodes. However, the effects of drought on forests are complex since each species has different resistance and adaptation strategies, and the drought characteristics (severity and time scale) and bioclimatic context all strongly influence how forests respond to drought. The literature also indicates that there is a clear

connection between drought and wildfire activity. However, the nature and strength of this connection is strongly modulated by other factors such as invasive species, fire management, and increasing human settlement in the wildland-urban interface.

2.4. Land degradation and desertification

Land degradation is one of the main environmental challenges in semi-arid regions of the world (Reynolds et al., 2007). Although desertification is sometimes confused with drought, these two phenomena are different since drought is a climatic phenomenon characterized by a temporal anomaly in water availability with respect to the long-term climatology (Wilhite and Pulwarty, 2017). On the contrary, land degradation is characterized by the loss of the productive capacity of the ecosystems and decreases in vegetation growth and vegetation cover (LeHouerou, 1996). There has been substantial scientific debate about the importance of drought as a driver of land degradation in semi-arid regions. Some authors argue that decreases in precipitation cause reductions in vegetation cover and that this leaves the soil more vulnerable to the effects of wind or water erosion (Kassas, 1995; Stringer et al., 2009; Thurow and Taylor, 1999). Under these conditions, soil erosion and reductions in soil fertility prevent vegetation from recovering once precipitation returns to normal. If dry conditions persist for long enough, this scenario could result in irreversible land degradation characterized by sparse vegetation and isolated vegetation patches (Kefi et al., 2007). This theory, that drought can cause land degradation and desertification, is supported by observations from the African Sahel, where a prolonged period of abnormally dry conditions that spanned the 1970s, 1980s and 1990s (Nicholson, 2001), caused a widespread decline in vegetation coverage in the region (Eklundh and Olsson, 2003).

However, the connection between drought and desertification has been reevaluated recently, again due to observations from the Sahel. A widespread greening of the Sahel is occurring due to the return of wetter conditions (Anyamba et al., 2014; Herrmann et al., 2005;

Olsson et al., 2005). Therefore, it has been suggested that desertification may not be as widespread or permanent as previously thought (Nicholson, 2001) and that drought is not an important cause of desertification, as compared with the past and present impact of humans and their livestock (LeHouerou, 1996; Rishmawi and Prince, 2016).

A more moderate view suggests that drought is one of the factors of land degradation, but it cannot be considered as the sole explanatory or triggering factor. Drought must coexist with other factors, such as erosion, poor land management, or overgrazing, to cause desertification to occur. Thus, even under the most limiting environmental conditions, it is difficult to identify degradation processes that are only due to drought. An example of this phenomenon was illustrated by Vicente-Serrano et al. (2012b) in a semi-arid region of southern Europe characterized by limited soil quality (gypsums) for water retention. In this region, vegetation cover has been used by grazing livestock. In addition, drought frequency and severity have increased during the past fifty years because of the increasing atmospheric evaporative demand. Despite the recent drought conditions, since livestock and human pressure in this region is currently limited, vegetation recovery is occurring in most locations. Only in a few small areas, characterized by limited vegetation cover and harsh environmental conditions (south facing slopes), is drought accelerating the loss of the vegetation cover and, consequently, land degradation.

Desertification is commonly limited to semiarid and arid regions in which dominant vegetation are grasses and shrubs (LeHouerou, 1996). Thus, semiarid grasslands are sensitive to drought since soil moisture is necessary to sprout and maintain the photosynthetic activity of grass plants in opposition to the shrubs, which are more drought resistant (Reynolds et al., 1999). Under severe drought conditions, the areas between shrubs may lose herbaceous coverage and under some circumstances this can trigger further degradation due to loss of soil organic matter and due to soil erosion (Allen et al., 2011; Mangan et al., 2004). Under these conditions, the herbaceous cover may have difficulty recovering (Dardel et al., 2014), even

after moisture conditions return to normal, since the soil will have lost organic matter and will have a reduced water holding capacity. In severely degraded locations, the entire soil column can be eroded, leaving only the rock substrate.

In degraded landscapes, shrubs are often located in “islands” of fertility between barren areas where soil fertility and soil water content is low. These shrub communities can be strongly resistant to drought. Reynolds et al. (1999) documented the replacement of large areas of semiarid grasslands by shrubs in the southwestern United States. Shrubs show a high degree of resistance to drought events because they have adapted to water stress through various physiological mechanisms (Wonkka et al., 2016). Whitford et al. (1995) used drought experiments to illustrate how after five consecutive years of summer drought, the growth of the shrubs was not significantly affected and even though perennial grasses and forbs disappeared on the experimental plots, the density and biomass of shrubs had increased.

Some authors have suggested that shrub-dominated semiarid ecosystems have a climatic origin (Brown et al., 1997), however, other studies have demonstrated semiarid grasslands are resilient to drought (Nicholson et al., 1998). Semi-arid vegetation has adapted to drought periods using different physiological, phenological and morphological strategies (Chaves et al., 2003), and they recover quite well after drought events even if the droughts are severe and persistent (Herrmann et al., 2005; Heumann et al., 2007). Therefore, degradation is usually driven by other factors and droughts only exacerbate them. The general recovery of the semiarid grasslands in the Sahel after three very dry decades is an excellent example of how grasslands can recover after precipitation increases. The recovery of the Sahel has shown that, contrary to the hypothesis of irreversible desertification in this area, there is strong grassland recovery once precipitation returns to normal (Dardel et al., 2014; Herrmann et al., 2005; Olsson et al., 2005).

Prince et al. (1998) showed little interannual variability in the rainfall use efficiency and rapid recovery of net primary production following drought in the Sahel. Nicholson et al.

(1998) showed no changes in the Saharan boundary or in vegetation density and coverage in the Sahel, nor a systematic reduction of the potential vegetation productivity and the water-use efficiency. This suggests that although drought effects can be evident in arid ecosystems, long-term trends associated with drought events are difficult to identify. Remote sensing products and field experiments seem to support this hypothesis. Diouf and Lambin (2001) analyzed long-term field vegetation variability in the west Sahel and observed less biomass production as response to drought, but no general trends in vegetation activity and abundance of grass species. An illustrative example of the recovery capacity of these semiarid ecosystems was provided by Nyssen et al. (2009), who developed a long-term study (>140 years) in the Sahel based on photographs. They showed that, independent of the strong drought episodes that affected the region during the twentieth century, the dominant pattern illustrated by the photographs was an increase in vegetation cover during the past 150 years.

Mainguet and Da Silva (1998) note that the consensus has now shifted so that desertification is primarily viewed as human-induced, although separating natural short-term and climate-induced changes from land degradation determined by human activities is quite difficult. This does not mean that drought events are irrelevant in these semi-arid regions, since the reduction of vegetation coverage and biomass reduces the available food for wildlife and livestock, and causes migration and mortality of wildlife (Sinclair et al., 2007) and ultimately can cause human migration, famine and mortality (Haile, 2005).

Recent quantitative studies relating climate drought indices and long time series of vegetation activity from satellites do not provide clear answers about the long-term trends of vegetation change related to drought (Zhang and Wang, 2015). In a global study covering the world semiarid regions, Vicente-Serrano et al. (2015a) showed that drought conditions could not explain most of the observed changes in vegetation activity in these areas during the last three decades. This illustrates the complexity of vegetation processes in semiarid regions, and the challenges of identifying a global response to drought severity. There are many

confounding environmental and anthropogenic factors that influence how vegetation responds to drought, including the effect of higher CO₂ concentrations, which may increase the resistance of semi-arid vegetation to water deficits (Donohue et al., 2010). This stresses the need for more local to regional studies in vulnerable regions to unravel the complex effects of drought.

In summary, there has been substantial debate in the literature about the importance of drought as a driver of land degradation in semi-arid regions. While some studies argue that drought is a major cause of land degradation, the balance of evidence supports the hypothesis that drought is one of the factors causes of land degradation, but it cannot be considered as the sole explanatory or triggering factor. Drought must coexist with other factors, such as erosion, poor land management, or overgrazing, to cause desertification to occur. Similarly, desertification is primarily viewed as human-induced, although separating natural climate-induced changes from land degradation due to human activities is difficult. Due to the the complexity of vegetation processes in semiarid regions, it is difficult to identify long-term trends in vegetation change related to drought.

2.5. Habitat degradation and wildlife mortality and migration

Drought is an important factor that contributes naturally to population regulation and other ecological processes. There is paleoclimatic evidence that population declines and extinctions are related to climatic change, and among these changes, shifts to a drier climate are the most significant (Haynes, 1991). There are relatively few studies that examine the effect of drought severity on wildlife as compared to the impact of environmental droughts on forests and aquatic systems, which are well studied. This is due to the difficulty of undertaking long-term experimental studies to follow animal populations as climate conditions vary and the large diversity and complexity of drought effects on wildlife communities. The effects of drought on animal communities can be complex, and depend on interactions with habitat,

other species, human activities, etc. Nevertheless, existing studies suggest that drought has an important influence on biodiversity and wildlife (Desbiez et al., 2010; Ford, 2011; Grisham and Boal, 2015; Lusk et al., 2007; Ottichilo et al., 2000). The drought effects on animal communities are varied and may include reduction of food and water, increases in disease and mortality, migration or even extinction.

Sinclair et al. (2007) published the results of a long-term experiment in the wildlife ecosystems of the Serengeti showing that droughts created disturbances that affected survivorship of ungulates and birds. They showed the importance of food limitations in the regulation of mammal populations, particularly in wildebeest (*Connochaetes taurinus*) and buffalo (*Syncerus caffer*). In the southern Kalahari, Knight (1995) showed that during the dry season of 1985 a high percentage (10%-35%) of ostrich (*Struthio camelus*) and four ungulate species died because of the drought. Hillman and Hillman (1977) published one of the first monitoring experiments in the Nairobi National Park during the drought period of 1973/74. Through monthly ground counts, they showed mortality rates between 7% and 27% for the species living in the park due to poor physical conditions and reduced disease resistance caused by malnutrition after sixteen consecutive months of below-average rainfall. Waite et al. (2007) also documented a large mammal mortality in the Kumbhalgarh Wildlife Sanctuary in Rajasthan, India related to a drought that occurred in 2000. They showed that because of the drought there was a 50% decrease in *Hanuman langurs* (an Indian monkey) from 1999 to 2001. Seabrook et al. (2011) showed that koala (*Phascolarctos cinereus*) populations of south-west Queensland (Australia) were strongly affected by droughts that affected the region from 2001 to 2009. They identified an 80% decline in koala numbers during their study period, from a mean population of 59,000 in 1995 to 11,600 in 2009. Recher et al. (2009) suggested that populations of different small mammals in a coastal eucalypt forest of south-eastern Australia are also mostly driven by rainfall variability and drought.

Not all species respond to drought in the same way, and the influence of drought can vary depending on life histories (Anderson et al., 2015). Drought may also affect the individuals within a given species in a different way. Drought acts as a means of natural selection since only individuals with the appropriate behavior survive. Foley et al. (2008) explored the survival in an elephant (*Loxodonta Africana*) population in a national park of Tanzania during a severe drought that affected the region in 1993 and showed that young males were particularly sensitive to the drought. Moreover, mortality was also higher among young mothers than among more experienced mothers. They reported high variability in mortality between family groups. The family groups that remained in the national park suffered much higher mortality than the ones that left the park.

Therefore, although droughts can have a noticeable influence on wildlife populations, the drought effects are usually temporary in non-degraded habitats since well-developed and undisturbed animal communities show a high rate of replacement and have a greater capacity for recovery after a drought. For example, Gasaway et al. (1996) analyzed the density of four species of plains ungulates between 1960 and 1994 in Etosha National Park (Namibia). They found that the pregnancy rate of some species was high during drought and that the recruitment of all four species was not related to precipitation. Moreover, a high mortality rate was not recorded during droughts. Gould et al. (2003) collected life-history data on a population of ring-tailed lemurs (*Lemur catta*) at the Beza Mahafaly Reserve, Madagascar and although they identified declines in the population as a consequence of a 2-year drought, the population recovered rapidly after precipitation return to normal. Also, Knight (1995) in the southern Kalahari showed a reduction of ostrich and ungulate species as response to drought events, but that mortality was lower in adult females. This fact favored a rapid recovery of the populations after drought. Bradley et al. (2006) illustrated the high resilience to drought of five species of rodents at Chaparral Wildlife Management Area, near Catarina, Texas. In this area, extreme drought conditions were recorded in 2001, with 13 months of below average

precipitation. They compared the number of unique individuals captured during the drought and the post-drought periods and found a substantial increase in the population (approximately 500%), stressing the strong capacity and rapidity of the recovery of animal populations recorded in undisturbed habitats.

There is also evidence of animal movements and migration in response to droughts. Pople et al. (2007) analyzed the density of red kangaroos (*Macropus rufus*) in pasture lands of South Australia and found strong interannual variability in the spatial patterns that were driven by the precipitation received in each area. In general, they showed that during dry times, red kangaroos were more spatially aggregated. Duda et al. (1999) analyzed the effects of drought on desert tortoise movement in the Mojave desert. They found that tortoises tended to travel a shorter distances during drought years.

In general, droughts tend to have the greatest influence on wildlife communities when they occur in combination with other stressors; among them human activity and competition for wildland habitats are the most important. Experimental studies have stressed this issue. de Leeuw et al. (2001) analyzed the distance between wildlife and livestock in northern Kenya and water sources. They showed that in dry years livestock were concentrated closer to these water sources, while wildlife were farther away. This suggests that livestock and human activities may disturb in the capacity of wildlife to survive under drought conditions. Ali et al. (2017) analyzed the dynamic of an endangered antelope (*Beatragus hunter*) in Kenya between 1985 and 2012 and showed that tree encroachment in the region, indicative of land degradation, was associated with a 98% decline in antelope populations. In this case, drought, overgrazing and habitat were closely related and can explain the near extinction of this species. Thus, the capacity of an animal population to cope with drought events in highly degraded habitats is greatly reduced. Lunney (2001) analyzed the drivers of extinction for native mammals in New South Wales (Australia) during the nineteenth century and concluded that sheep were the main driver of extinction, mostly given livestock colonization of wildlife

refuges during drought events. This assessment is supported by other studies that show stronger mortality of native herbivorous species during late nineteenth century megadrought in Australia in areas infested by rabbits (Godfree et al., 2019).

There are few studies that have analyzed differential drought impacts on animal communities in both humid and arid environments. Nevertheless, in general, drought effects on wildlife mortality are stronger in arid and semiarid regions and in the areas located near the distribution limit of the animal populations. In a recent study of the megadrought period that affected Australia between 1891 and 1903 and using ecohistorical techniques, Godfree et al. (2019) showed that different animal communities located in arid and semiarid regions had mass mortalities mostly concentrated in primary producers and herbivores. Experiences of reintroduction of species in these habitats show strong effects of droughts on semi-arid animal populations. Davidson et al. (2014) reintroduced over 1,000 Gunnison's prairie dogs (*Cynomys gunnisoni*) in the southern portion of their range of distribution in central New Mexico and studied their population dynamics for 8 years. Four severe drought events were recorded during spring, which is when mating, pregnancy, and lactation occur. They found that the introduced community was vulnerable to drought since the estimated survival rate was below 12%. A similar experiment was performed by Facka et al. (2010) to restore populations of black-tailed prairie dog (*Cynomys ludovicianus*) in Chihuahuan Desert grasslands. They found that a 35% reduction in precipitation during 2003 resulted in a juvenile survival near zero. In addition, juvenile growth rates were only about half of the normal rates. Avila-Flores et al. (2012) also showed that natural prairie dog communities in northwestern Mexico were dramatically reduced between 1988 and 2005, a period characterized by severe and persistent droughts. The associated reduction in the availability of food production was the main factor behind the observed population decline. Therefore, populations at the margins of their geographic ranges and in regions affected by habitat degradation can be especially vulnerable to drought. The effects on different species can be complex and recent studies have suggested

that even in arid and semi-arid regions extreme droughts can produce differential results among species, with both winners and losers. Prugh et al. (2018) found that during the extreme 2012-2015 drought in California the abundance of some rare species of birds and reptiles was greater than normal, but the abundance of generalist species and many carnivores decreased sharply. These authors argued that species near in the top of the food chain would only be affected by exceptionally long droughts.

In summary, existing studies indicate drought has a significant influence on biodiversity and wildlife. Animal communities are effected by drought in many ways. These include reductions in food and water, increases in disease and mortality, migration or even extinction. Not all species respond to drought in the same way, and the influence of drought can vary depending on life histories. Although droughts can have a noticeable influence on wildlife populations, the influence of drought is usually temporary in non-degraded habitats since undisturbed animal communities tend to have a high replacement rate and a significant capacity for recovery from stressors. Droughts tend to have the greatest influence on wildlife communities when they occur in combination with other stressors; among them human activity and competition for wildland habitats are the most important. The effects on different species can be very complex and recent studies have suggested that even in arid and semi-arid regions extreme droughts can produce differential results among species, with both winners and losers.

2.6. Quality and health of freshwater biomes

The influence of drought on the quantity of surface and subsurface water is widely known and there are existing reviews that summarize this issue (Tallaksen and van Lanen, 2004; Van Loon, 2015). Here we focus on the environmental consequences of the water reduction in streams, lakes and ponds as a consequence to drought in relation to the freshwater ecosystems and water quality. The reduction of water flows, as a consequence of

drought, causes different environmental effects, including water temperature increases, changes in the chemistry and biochemical conditions of the water and, ultimately, changes at the ecosystem level, including effects on plants and animals. The relevance of drought effects on the health and quality of freshwaters explains the existence of a vast scientific literature about this issue, including recent reviews on the varied and complex effects of droughts on these systems. For example, Lake (2011) published a book that provides a detailed review of the many different ways that drought effects aquatic ecosystems, including the effects on temporary waters, floodplain rivers and wetland complexes, the effects on plants and invertebrates, fishes and finally the exacerbation of drought effects on aquatic ecosystems as a consequence of human activities. Aldous et al. (2011) evaluated drought effects and adaptation strategies in freshwater ecosystems under climate change. There are also other reviews that focus on specific effects of drought on aquatic systems. Mosley (2015) published a review of how drought effects water quality in freshwater systems which stressed that drought effects may be variable and complex and that they depend on many different factors. These include the location and type of reservoir, the local climate conditions and the antecedent weather conditions before the drought event. Finally, Bond et al. (2008) also reviewed drought effects on freshwater ecosystems from the Australian perspective. Here we only provide a brief synthesis of this issue to maintain coherence with the other sections of this article. A full discussion of all of the ways that drought effects aquatic systems is beyond the scope of this paper and the interested reader is referred to these other reviews.

Reductions in water quantity due to drought are usually associated with an increase in water temperature (Hrdinka et al., 2012; van Vliet and Zwolsman, 2008; Ahmadi et al., 2019). Van Vliet et al. (2011) analyzed 157 stations worldwide with data of river temperature to determine the effect of both air temperature and river discharge changes, and showed that discharge decreases of 20% and 40% caused water temperatures to increase by +0.3°C and +0.8°C, respectively. It is necessary to stress that most of the analyses of water temperature

changes were during summer. The possible effects during the cold season are less known, although they are expected to be weaker given the influence of chemical reaction kinetics (Whitehead et al., 2009).

Effects of droughts on water quality can be diverse (Mosley, 2015). Drought usually increases salinity concentrations (Burt et al., 2015; Mosley et al., 2012), but reduces nutrient loads (Cozzi and Giani, 2011; Mosley et al., 2012; Oelsner et al., 2007; Zielinski et al., 2009). Lower flows usually reduce water velocity, which increases residence time of the water in rivers, increasing the risk for algal blooms (Bowling et al., 2013; Garcia-Prieto et al., 2012). Van Vliet and Zwolsman (2008) analyzed the influence of droughts on water quality in the Meuse River between 1976 and 2003, using 24 water quality parameters. They showed a general deterioration in water quality during droughts events based on variables such as water temperature, eutrophication, the presence of major elements, as well as heavy-metal concentrations. The decline in water quality is mostly caused by the development of algae blooms which result from higher water temperatures, longer residence times, and higher nutrient concentrations (Hellwig et al., 2017). Drought tends to have a greater influence on water quality in catchments dominated by agricultural/pastoral activities because they have greater concentrations of bacterial contamination and chemical fertilizers (Caruso, 2002). Water quality was negatively influenced during dry years in the Dutch-German border of the Rhine River with respect to eutrophication and the presence of heavy metals (Zwolsman and van Bokhoven, 2007). Mosley et al. (2012) also showed that low flows in the Murray River in South Australia resulted in significant increases in salinity, total phosphorus, total nitrogen, chlorophyll and turbidity because of the lack of flushing.

Oxygen concentrations and dissolved organic matter can be affected by drought (Worrall and Burt, 2004; Worrall et al., 2003; Worrall and Burt, 2008). Dai et al. (2006) observed that oxygen was depleted in the Pearl River estuary during the drought of 2000 as a consequence of biochemical processes such as nitrification that substantially increased the

consumption of oxygen. Worrall et al. (2003) published a long-term (> 30 years) monitoring experiment of riverine dissolved organic matter in three catchments in Northern England and showed that drought had a strong inter-annual control such that carbon releases decreased with drought intensity. They also showed that recovery to pre-drought conditions required 3-4 years after the end of the drought. Worrall & Burt (2008) extended the analysis to 97 sites of England to illustrate the complex nature of how drought influences organic matter concentrations. The fragmentation of streams generates strong heterogeneity of organic matter throughout the fluvial system (Vazquez et al., 2011).

Nutrient concentrations, water turbidity and algal concentrations may also increase during drought events as a consequence of reduced flushing and increased biological productivity (Heisler et al., 2008; O'Farrell et al., 2011). This is caused by external influences [e.g., water inflow, geology and sewage treatment plants (Hellwig et al., 2017)], but also by internal influences (e.g., biological uptake of nutrients and denitrification). Moreover, with human pollution of the streams, water quality is affected by deterioration due to less dilution for nutrients (Eimers et al., 2008), nitrates, and pharmaceuticals compounds (Benotti et al., 2010), which may affect drinking water treatment processes (Wright et al., 2014).

In summary, the majority of existing experimental studies suggest that drought events have a negative effect on water quality in streams and lakes by increasing water temperature, salinity, the frequency of toxic algae blooms and contaminant concentrations, and reducing the concentration of nutrients and dissolved oxygen (Mosley, 2015).

Ecological effects of hydrological drought are complex and depend on streamflow characteristics, human uses and aquatic communities (Lake, 2011). However, in general, the effects are negative and they are driven by two factors: (i) the reduction in water availability, and/or (ii) decreases in water quality. The reduction of water flow, the fragmentation of streams and the drying of ponds and streams make it difficult for some plant and animal communities to survive. There are several sampling studies that illustrate how this has a

negative effect on an aquatic population's density and diversity (Boix et al., 2010; Bond and Lake, 2005; Walls et al., 2013). As a representative example, Walls et al. (2013) analyzed the influence of drought on salamander (*Ambystoma talpoideum*) populations in isolated wetlands in the southeastern U.S. from 2009 to 2012. They found that the number of ponds in which the salamander was present declined from 22.3% to 9.9% as a response to drought conditions.

Monk et al. (2008) analyzed the response of macroinvertebrates to inter-annual variations in streamflow regimes at 83 sites across England and Wales and showed that two major supra-seasonal droughts (1990-1992 and 1996-1997) had noticeable effects on the diversity and density of these populations. Chase (2007) investigated changes in community composition in response to water deficits among ponds with similar environmental conditions. He showed strong spatial differences in the composition of pond communities during humid periods. However, in the ponds affected by drought, Chase (2007) found higher similarity among communities. The species that could not tolerate this stress were removed from the ponds through an adaptive selection process. There are other examples of the drought effects on aquatic biodiversity. Jeffries (1994) analyzed macroinvertebrate community fauna in temporary and permanent ponds of Scotland and found that permanent ponds that dried up in 1992 lost the majority of species characteristic of permanent water conditions, but acquired other new species characteristics of temporary ponds. Jeffries (1994) also showed that colonization rates were low for the species characteristic of permanent water, but high for the species characteristic of temporary ponds. Moreover, the resistance and resilience of invertebrate communities is quite different between seasonal dryness episodes and supra-seasonal droughts. Extreme droughts may cause drastic changes in the streams, with a transformation from perennial to intermittent flows and although the species richness can recover quickly after river flows return to normal conditions, studies suggest that changes in the community structure may delay the ecological recovery to pre-drought conditions (Bogan et al., 2015).

In general, the recovery of the aquatic species after a drought episode shows that species are strongly resilient (Morrongiello et al., 2011) and recovery times are usually less than three years after a drought (Niemi et al., 1990). Nevertheless, there are exceptions when the disturbance results in physical alteration of the existing habitat, or the system becomes isolated and recolonization is suppressed if refuge is not available, or the connectivity between the different parts of the aquatic system is lost (Bond et al., 2008; Lake, 2011). Although drought events can be a severe disturbance to aquatic populations, it is often human influence that exacerbates drought severity and constrains the capacity of recovery, rather than drought per se (Crook et al., 2010).

The decreased water quality that results from a drought event may affect aquatic wildlife (Pillay and Perissinotto, 2008). Sparks et al. (1990) showed in the Mississippi river that the absence of a flood during the 1976-1977 midwestern drought caused contaminants in key species to increase temporarily because of the reduced dilution. Parr and Mason (2003) analyzed the influence of water quality changes on macroinvertebrate populations in eastern England. They showed that low flows associated with droughts are the main factor of population declines in the rivers since they cause a reduction in dissolved oxygen and a significant increase in chloride concentrations, oxidized nitrogen and reactive phosphorous, which ultimately affect the macroinvertebrate populations. The effect of droughts on aquatic communities can be also indirect via flood occurrence after drought events. Whitworth et al. (2012) found increased fish and crustacean mortality as a response to the hypoxic blackwater events in the Murray-Darling Basin, Australia. These events were characterized by increases in dissolved organic carbon, which depletes dissolved oxygen and causes mortality. They identified these events after ten years of severe drought affected the region. This prolonged drought meant that when water returned to some floodplains that had remained dry for a long time, large stores of organic matter were mobilized, increasing hypoxic processes in the river. Aherne et al. (2006) and Eimers et al. (2007) showed an increase of sulphate efflux in central

Ontario wetlands (Canada) following drought events. This occurs because the water table drops during drought periods and this promotes oxidation of previously stored sulphur compounds, with subsequent efflux of sulphates upon re-wetting. This stresses the complex influence of droughts on water quality and aquatic communities, since the effects of the drought may not occur until after wetter conditions return due to the complex geochemical processes.

In summary, the majority of studies demonstrate that drought events have negative effects on water quality in streams and lakes by increasing water temperature, salinity, the frequency of toxic algae blooms and contaminant concentrations, and reducing the concentration of nutrients and dissolved oxygen. Although the ecological outcomes of drought are generally negative, including, for example, significant declines in the macroinvertebrate population, the responses are complex and depend on the streamflow characteristics, human uses, and aquatic communities. The literature indicates that aquatic species can recover relatively quickly after a drought ends. There are exceptions when the disturbance results in physical alteration of the existing habitat, or the system becomes isolated. Recolonization is suppressed if refuge is not available, or the connectivity between the different parts of the aquatic system is lost.

2.7. Long-term/permanent environmental drought effects

As illustrated in the previous sections, the majority of environmental drought effects are temporary. Thus, drought is a normal part of the climate system, and not an anomaly (Glantz, 2003), so it is expected that non-perturbed environmental systems have a strong capacity of resilience and recovery after drought perturbations. Even the most vulnerable systems affected by drought (e.g., semiarid regions, aquatic systems and perturbed areas) typically recover once moisture conditions return to normal. This occurs in many different systems, ranging from soil microbial fauna and biochemical environments (Rousk et al., 2013)

to forests (Gazol et al., 2018; Pasho et al., 2011) and to freshwater ecosystem populations (Lake, 2011; Caruso, 2002; Ahmadi et al., 2019). The resilience of these different systems to drought tends to be high, even in regions affected by decadal droughts. A complete recovery of the system to the pre-drought conditions is the normal state of affairs, if given enough time (Anyamba et al., 2014). However, there are important spatial and environmental differences in drought recovery that depend on the climate and carbon cycle dynamics (Schwalm et al., 2017; Yu et al., 2017) as well as the resilience of different ecosystems to drought (Gazol et al., 2017 and 2018). Drought resilience is also a function of drought time-scales (Vicente-Serrano et al., 2013) and the duration and severity of the drought (Ahmadi et al., 2019b). Thus, recovery times have been found to vary among systems; they are usually short for soil and freshwater systems (Borken and Matzner, 2009; Morrongiello et al., 2011), whereas they can be longer (years) for the recovery of the vegetation in areas affected by drought and forest fires of extraordinary virulence that may have affected soil properties and the seed bank (Diaz-Delgado et al., 2002). However, even in the case of forest fires, recovery will occur in the years following the fire (Vicente-Serrano et al., 2011a). Recent studies have suggested that drought recovery time, as measured by an environmental metric like gross primary production, is increasing (Schalm et al., 2017). This may result in more permanently damaged environmental systems.

There are some circumstances where recovery does not occur. In these cases, the effects of drought can persist over longer periods or even to be structural. Reynolds et al. (1999) illustrated the ecological mechanisms that cause the replacement of dominated semiarid grasslands by shrub landscapes, a process favored by droughts, are persistent and usually irreversible. Also, drought may favor changes in the vegetation community structure and composition. In the Sahel, several studies have stressed the recovery capacity of the ecosystems after severe and long-lasting droughts. Prolonged drought conditions, like those that affected the Sahel between 1970s and 1990s, have triggered changes in the composition

of the dominant species once vegetation activity and coverage have recovered. Hänke *et al.* (2016) showed in northern Burkina Faso that after the strong decrease in tree cover in the 1970s and 1980s, there has been an increase since the 1990s. However, they also showed that the species composition had changed towards a higher proportion of drought-resistant species. This was also found by Maranz (2009) in the Sudano-Sahel zone, where there was a loss of mesic trees in favor of arid vegetation species. These strong ecological and landscape changes are persistent and may need decades, characterized by more humid conditions, for the replacement by more mesic species. These examples stress the complexity of current landscape and vegetation changes after a drought event and suggest that the recovery of vegetation coverage/activity is not a sufficient indication of post-drought recovery at the ecosystem level; rather species diversity and richness must also be considered.

Although semiarid ecosystems are highly resilient to water deficits, drought effects on soil quality can also be structural if biogeochemical processes are affected by strong drought events. Evans and Burke (2013) examined carbon and nitrogen cycling in the shortgrass steppe of the U.S. Great Plains during an experimental eleven-year drought. They showed that soil inorganic nitrogen was up to five times higher during drought conditions, but plants and microbial communities could not use this nitrogen due to diffusion limitations. Thus, the drought plots had strong N₂O fluxes once they received rain, suggesting that accumulated N may be highly vulnerable to loss. This suggests that droughts can degrade soil quality, and during particularly extreme events, the lack of vegetation coverage favors erosion processes and structural drought effects (Dardel *et al.*, 2014; Reynolds *et al.*, 1999). This process has been observed at the local scale in xeric soils affected by droughts like in the Sahel (Dardel *et al.*, 2014) and semiarid areas of Europe (Vicente-Serrano *et al.*, 2012b).

Permanent effects of droughts have also been identified in animal communities. The near-extinction of the introduced populations of prairie dogs (Avila-Flores *et al.*, 2012; Facka *et al.*, 2010) is an excellent example of irreversible drought effects in populations located near

their limits of distribution. Endangered populations can be permanently modified by drought events, and freshwater ecosystems are particularly vulnerable (Hammer et al., 2013; Perkin et al., 2013). Trape (2009) illustrated how droughts may cause the extinction of populations by means of the study of the relict tropical fish in the Adrar mountains (Mauritania). Trape (2009) found that the extreme drought that affected the Sahel between 1970s and 1990s caused the only known stream with populations of *Barbus mirei* to dry up in 1984, and the fish became extinct. Trape (2009) also showed that of the thirteen previously recorded populations in regional streams, four became extinct during the drought. Hammer et al. (2013) found similar results in the southern Murray-Darling Basin, Australia as a response to the drought that affected the region between 2000 and 2010. Low stream flows combined with habitat degradation and alien species caused five threatened fish species to reach a critical point (imminent species extinction) during 2007-2010. Also, Willson et al. (2006) analyzed the responses of semi-aquatic snakes inhabiting isolated wetlands of the southeastern United States to a strong drought that affected the region from 2000 to 2003, and showed that banded watersnakes (*Nerodia fasciata*) suffered a dramatic population decline as a response to the drought event that also affected their capacity for reproduction. Insect populations can also be dramatically affected by droughts. Among the few existing studies, Ehrlich et al. (1980) showed that several populations of checkerspot butterfly (*Euphydryas*) became extinct and others were strongly reduced as a response of the California drought of 1975-77.

In some severe and persistent drought events, structural drought effects may also affect the equilibrium among species in an ecosystem. Harrison (2000) analyzed the effect of the strong drought that affected Indonesia in 1998 on the Figs (*Ficus spp.*) and their species-specific pollinators, the fig wasps (*Agaonidae*). The survival of these two species is tightly coupled because the wasps pollinate the figs and the inflorescences on the figs serve as a place for the wasps to raise their offspring. The severe drought that affected northern Borneo in 1998 due to the strong El Niño, reduced the production of inflorescences on figs and caused

the extinction of the wasps. This also could cause cascading effects on the animals which feed on the figs and act as seed dispersers.

We have stressed in this paper that most environmental drought effects are temporary since drought is a natural part of climate variability, and environmental systems are usually well adapted to the occurrence of water deficits. Nevertheless, in some circumstances, permanent environmental drought effects are observed. These permanent effects usually occur in regions/systems strongly modified by human activities or in populations that live near of their limits of distribution, or under very limiting conditions that may constrain the capacity of recovery of the system/population when severe drought conditions occur.

3. Observed changes in drought severity and related environmental effects

If drought frequency and severity increase as a consequence of climate change, this may exacerbate the effects on environmental systems or push them past their resilience limits. This section analyses the recent changes in drought severity at the global scale.

Determining possible changes in drought severity is difficult since drought is an effect-related phenomenon. Therefore, the best way to identify changes in drought severity is to monitoring effects in the affected systems. This type of comprehensive monitoring and quantification is challenging. Moreover, drought effects not only depend on climate, but also on the vulnerability of the different sectors and systems affected, an aspect related to the resistance and resilience to drought. Environmental vulnerability may change as a consequence of land cover changes, ecological and hydrological processes, human interventions (e.g., water regulation and damming), and other factors. All these factors make it extremely difficult to objectively assess possible changes in drought severity over the time. For this reason, changes in drought severity are usually assessed using time series of climate variables measured at meteorological stations (Heim, 2002). Although the climatic component is not sufficient for characterizing drought, the majority of drought events have a climatic

origin, so using climate information is an objective and spatially and temporally comparable way to quantify drought severity and to determine possible changes in drought frequency and severity.

Even from a meteorological point of view, drought is extremely difficult to measure since there are several technical problems with drought quantification. For example, drought cannot be measured by a single instrument located in the field and there is not a unique variable that we can use to have a complete assessment of the meteorological drought severity (Vicente-Serrano, 2016), given the varying influence of precipitation and atmospheric evaporative demand on drought severity in different climate regions (Vicente-Serrano et al., 2015) and environmental systems (Vicente-Serrano et al., 2019). Determining the effects of climate change on drought frequency and severity is an issue that it is similarly not easy to address, even from a climatic perspective (Vicente-Serrano et al., 2017). There is no consensus on how drought severity is changing in the current climate, given the data limitations and atmospheric circulation processes (Trenberth et al., 2014). There are also uncertainties related to the role played by the precipitation and/or the atmospheric evaporative demand to explain drought severity (Dai, 2011; Sheffield et al., 2012). These facts explain why the most recent Intergovernmental Panel on Climate Change (IPCC) report that focused on the evolution of drought severity (Seneviratne, 2012) suggested only low-to-medium confidence in recent drought trends worldwide. Available precipitation data are not sufficient for a global assessment, and other climate variables that affect drought severity (e.g. the atmospheric evaporative demand, AED) are even less available (Trenberth et al., 2014). Regional studies show a variety of results, since drought severity has increased in recent decades in some regions as a consequence of precipitation decreases (Hoerling et al., 2012) or increases in the AED (Vicente-Serrano et al., 2014b), but other regions show stationary conditions or decreases in drought severity (Sheffield et al., 2012).

Independent of these results, there is other evidence that drought severity has increased as a consequence of increased AED associated to warming processes such as observed in the Mediterranean region (Vicente-Serrano et al., 2014b). Although there are uncertainties in the AED trends given the influence of all the variables (McVicar et al., 2012a; McVicar et al., 2012b), studies suggest increased vapor pressure deficit (VPD) in large regions worldwide (Vicente-Serrano et al., 2017), accompanied by strong warming trends (Arndt et al., 2017; Jones et al., 1999). All these trends will cause a positive trend in AED and likely increase drought severity.

Here we include an assessment of trends in drought indices using data from the Climate Research Unit Timeseries (CRU TS v. 3.25; <https://crudata.uea.ac.uk/cru/data/hrg/>) for precipitation and temperature (Harris et al., 2014) and from the European Centre for Medium-Range Weather Forecasts (ECMWF Re-Analysis, ERA)-Interim and ERA-20C (20th Century) datasets (<https://www.ecmwf.int/en/forecasts/datasets>) for dew-point temperature, wind speed, surface pressure and downward solar radiation, variables used to calculate AED by means of the Food and Agricultural Organization, FAO-56 Penman-Monteith equation (Allen et al., 1998). Data cover the period 1971-2017 at a monthly frequency and a spatial resolution of 0.5° degrees. Using these data, we have calculated two widely used drought indices at the global scale: (i) the Standardized Precipitation Index (SPI) (McKee et al., 1993), which is based on precipitation data, and (ii) the Standardized Precipitation Evapotranspiration Index (SPEI) (Vicente-Serrano et al., 2010), which is based on the climatic water balance, the difference between the precipitation and AED. Boreal areas and deserts were masked given the technical limitations to calculate and to define drought indices in these areas. These two indices can be calculated on different time-scales to adapt the different response times of environmental systems to water deficits (McKee et al., 1993; Vicente-Serrano et al., 2011b). Here we have chosen the 12-month time-scale, which is representative of the annual moisture conditions. To identify drought periods in a given grid cell, we used a threshold equal to -1.28, which

represents, according to the standard normal distribution that characterizes SPI and SPEI, the 1 in 10-year drought event (see details in Vicente-Serrano and Beguería, 2016).

Figure 2 shows the percentage of global surface affected by droughts according to the SPEI, SPI and the difference between them. SPI does not show noticeable changes in the global surface area affected by drought between 1970 and 2016, but the SPEI shows a clear increase during the same period. Thus, between 2000 and 2016, the SPEI shows between 2% and 10% more of the world surface area affected by drought than the SPI. Spatial patterns of drought trends recorded at the global scale suggest a decrease in the SPEI over many regions. A large decrease in the SPEI for the period 1970-2016 is recorded in large areas of South America, the Mediterranean, west of North America, South Africa and the Middle East (Figure 3a). This pattern contrasts with the observed trends in the SPI, which are not as pronounced in some of these regions (Figure 3b). Thus, in large areas of the world the difference between the magnitude of change between the SPEI and the SPI is statistically significant (Figures 3c, d). This suggests that changes in the AED may have increased the severity of drought events. Some uncertainties remain in these results given data constraints, but agreement with previous studies in some regions, as already observed in the Mediterranean region (Stagge et al., 2017; Vicente-Serrano et al., 2014b), Australia (Murphy and Timbal, 2008) or in South Africa (Edossa et al., 2016), gives support to this assessment. Moreover, observed trends in drought-related effects in economic, hydrologic and agricultural systems worldwide suggest increased drought severity associated with the AED (Vicente-Serrano et al., 2017).

Identification of recent trends in environmental drought effects is a complex task, mostly as a consequence of the paucity of long-term monitoring studies of drought effects. The majority of published literature about this topic is related to vegetation, forestry, and wildland fires (Hartmann et al., 2015; Ma et al., 2012) with no long-term studies on other environmental aspects like soil and air quality, animal communities and water quality. Case studies suggest an increase in the environmental influences of droughts related to more

frequent forest mortality episodes and declines in vegetation growth. Ma et al. (2012) analyzed biomass changes in the Boreal forest of Canada. They found a decrease in the rate of biomass change over the last decades in western Canada. Van Mantgem et al. (2009) used data from unmanaged mature forests in the western U.S. to show an increase in the mortality rates in the last few decades. Peng et al. (2011) also showed an increase of tree mortality rates by $4.7\% \text{ yr}^{-1}$ from 1963 to 2008 across Canada's boreal forests. These results suggest that water stress could be the main driver of the general increases in mortality rates recorded across tree species and biomes.

Nevertheless, other studies have suggested only weak drying and drought-induced mortality trends in dry climates (Steinkamp and Hickler, 2015). In any case, although severe tree-mortality episodes have been recorded in recent decades, it does not seem that drought has induced vegetation changes in many ecosystems. Martínez-Villalta and Lloret (2016) reviewed the literature and found evidence that drought-induced vegetation changes cannot be generalized. There are very few observations that support a replacement of the vegetation type following a drought, since even after substantial episodes of tree mortality due to drought, the dominant pattern is a self-replacement by the same dominant species.

Adams et al. (2017) suggested that natural vegetation is more vulnerable to drought under global warming since it produces hotter droughts and more rapid mortality. This is consistent with fundamental physiology. Moreover, the AED increases nonlinearly with temperature during drought given the Classius-Clapeyron relationship. A growing number of studies have been devoted to determining the effect of increased temperatures and atmospheric evaporative demand in drought severity on forest ecosystems. Bershears et al. (2005) proposed the term "global change type drought" to refer to drought conditions that are enhanced by a higher AED caused by increased temperatures. Adams et al. (2017) have shown experimentally that warmer temperatures increase the risk of mortality associated with drought events. This helps to explain the widespread occurrence of tree mortality episodes

due to the recent global warming. Nevertheless, other studies have stressed how additional variables that play a key role in AED may be more important for explaining forest stress and mortality. Eamus et al. (2013) used a physically-based model to determine drought influences on forest respiration and productivity. They showed that an increased VPD has a larger detrimental role on net primary production (NPP) than temperatures alone. In the western USA, using tree-ring chronologies, Truetnner et al. (2018) demonstrated that the drying effects of warm-season VPD negatively affected radial growth of different *Pinus* species. Will et al. (2013) used seedlings of 10 tree species in the Southern Great Plains to demonstrate that increased VPD associated with a temperature increase of 3°C increased mortality by 13% in relation to control conditions.

Wildfires also seem to have increased in areas in which drought severity has increased in recent decades. There are several studies that connect changes in drought severity with forest fires and show that global wildfire danger has increased from 1979 to 2013 (Jolly et al., 2015). Moreover, in relatively unmanaged areas in which the drought effects on forest fires can be isolated, it appears that drought effects may be increasing. Xiao and Zhuang (2007) indicated that fire activity has increased in Canadian and Alaskan forests in the last few decades. Abatzoglou and Williams (2016) used climate models to estimate the influence of anthropogenic climate change on observed increases in forest fire activity in the western United States. They found that increases in temperature and VPD significantly decreased fuel moisture content. Fuel load is function of both biomass and moisture content, but they suggested that climate change accounted for 55% of the observed increase in fuel loads. They estimated these anthropogenic climate changes increased the wildfire potential in recent decades and contributed to nearly doubling the burned area that occurred. These trends are not unique to the United States (Pausas and Fernandez-Munoz, 2012). Therefore, it is clear that the recent increases in wildfire activity are partly due increased drought severity. Also in agreement with this, Dennison et al. (2014) examined large wildfires in the western U.S.

between 1984–2011 and found a significant increasing trend in the number of large fires and the burned area at a rate of 355 km² per year. These trends were greatest in areas where drought severity had increased. Changes in wildfire regimes due to drought may cause long-term or permanent effects in some regions. In the Amazonian region, Brando et al. (2014) found an abrupt increase in forest fires (between 226%-462%) during the severe drought events that have affected the region during last decade. This caused a sharp decline in canopy cover (between 23% and 31%) and aboveground live biomass (between 12% and 30%) and the expansion of flammable grasses (63%-80%).

Further studies are necessary from a multi-system perspective to determine whether changes in environmental droughts in recent decades can be associated with increased drought stress or drought vulnerability in different systems. Long-term monitoring of desertification, animal communities and air and water pollution are also necessary to confidently assess the influence of droughts on these systems.

4. Future drought severity scenarios and projected environmental effects

Previous research has suggested that drought severity will increase given future climate change scenarios due to changes in precipitation and an increase in AED associated with warming (Dai, 2011). Nevertheless, it is quite difficult to assess possible future changes in drought given substantial uncertainty in precipitation projections (Maraun et al., 2010) and the uncertainties in other key variables that determine AED like relative humidity, solar radiation, and wind speed (Kingston et al., 2009). In addition, the varying ability of different drought indices to identify wet/dry processes associated with climate change projections is an additional cause of uncertainty (Mukherjee et al., 2018).

Here we have calculated future SPEI projections based on different scenarios for greenhouse gas concentrations (representative concentration pathways [RCPs] 4.5, 6.0 and 8.5 (Meinshausen et al., 2011)) which were obtained from the coupled model intercomparison

project 5 (CMIP5) climate models (Taylor et al., 2012). We have chosen to focus on SPEI since it has shown better performance than SPI for identifying drought effects in different systems (Vicente-Serrano et al., 2012a). There are a number of studies that have compared the usefulness of different drought indices for identifying the agricultural, hydrological, socioeconomic and ecological effects of drought. The SPEI has been included in several of these analyses, demonstrating that it provides better results than the SPI for explaining temporal variability and anomalies of different variables which are affected by drought such as soil moisture (Scaini et al., 2015), streamflow (McEvoy et al., 2012; Vicente-Serrano et al., 2014b), and crops (Potop et al., 2012). In addition, studies on multi-sectorial effects also suggest that the SPEI performs better than the SPI in relation to precipitation droughts (Bachmair et al., 2015; Blauhut et al., 2016).

Multi-model averages for precipitation, maximum and minimum temperature, dew point temperature, downward solar radiation and wind speed at the spatial resolution of 2.5° were used to calculate the SPEI for the period 1860-2100 (Figure 4). The period 1860-2010 is based on using observed greenhouse gases concentrations to force the models, and from 2010 to 2100 the three different RCPs were used. The SPEI was calculated based on the 1901-2014 reference period and the threshold used to select drought periods was, as explained above, an SPEI equal to -1.28.

Figure 4 shows that between 1860 and 1980 the models suggest a percentage of world regions affected by drought that oscillates around 10%, with some interannual variations, but clearly a stationary behavior. Since the 1980s there has been an increase in the area affected by drought, which coincides with the results shown in Figure 2. After 2010, the models show a large increase (much higher than in the observations) in the area affected by droughts. The affected area continues to increase until 2050, and then around 50% of the world regions would be affected by drought considering RCPs 4.5 and 6.0, but a continuous increase with RCP 8.5. Trends toward higher drought severity affect much of the land surface, including a

clear trend toward drier conditions in the Amazonia, South Africa, the Mediterranean, Australia and the United States.

As noted above, environmental systems respond to more than just precipitation and therefore it is also relevant to compare future trends in soil moisture conditions under different scenarios for greenhouse gas concentrations. Figure 5 shows the trends (2000 to 2100) in the volumetric water content (VWC) of the soil (% decade⁻¹) using the three most widely used emission scenarios by the IPCC (RCP 2.6, 4.5 and 8.5). The results are based on the average soil moisture trends across three CMIP5 models (CanESM2, MIRCO5, NorESM1-M). The patterns shown in Figure 5 are qualitatively similar to the patterns shown in Figure 4. This makes sense because the SPEI accounts for moisture supply (precipitation) and atmospheric demand (evapotranspiration). The majority of North and South America, Africa and Europe and western Asia is projected to have drier soil in 2100. The projected soil moisture trends are strongly driven by projected changes in precipitation. The surface soil moisture trends are more pronounced for the higher greenhouse gas emission scenarios (RCP8.5). As compared with most drought indices, the future trends in modeled soil moisture are not as pronounced.

We must take into account that the drought projections have uncertainties related to the models selected, the large variety of variables involved, and the emission scenarios. Nevertheless, the drought projections that we have provided here agree with those provided by other studies showing that drought severity may increase as a consequence of strong positive trends in AED in the 21st century (Jeong et al., 2014). This has been observed in regions such as Argentina (Sordo-Ward et al., 2017), Korea (Nam et al., 2015), the Mediterranean (Dubrovsky et al., 2014), the central U.S. (Feng et al., 2017), as well as at the global scale (Cook et al., 2014b).

Numerous studies have utilized climate change projections to evaluate potential environmental influences using statistical or process-based models, which, although they may have some uncertainty (Keenan et al., 2011; Lindner et al., 2014), can provide a general picture

on the expected drought effects in the future. It is possible that drought will induce more frequent heat waves in the future (Solberg et al., 2008). Increased wildfire risk is one obvious and direct effect of increased frequency of droughts and heat waves in future scenarios; there has been substantial effort invested in modelling wildfire response to droughts under future scenarios (Dury et al., 2011; Fyllas and Troumbis, 2009; Liu et al., 2013; Malevsky-Malevich et al., 2008; Mitchell et al., 2014a). Liu et al. (2013) developed a global assessment of future fire scenarios for the end of this century (2070-2100) and showed a potential increase in the United States, South America, central Asia, southern Europe, southern Africa, and Australia. They stressed that fire scenarios would be driven by temperature increases in the U.S., South America, and Australia and by the combination of temperature increases and drought in the other regions. Several regional studies seem to agree with these patterns. For example, Mitchell et al. (2014b) analyzed future climate and fire interactions in the Southern U.S. and predicted that an increase in drought severity will result in increased wildfire frequency. In Spain, Sousa et al. (2015) have projected a large increase in mean burned area. Thus, by 2075, mean burned area could be two to three times larger than at present. This pattern has also been observed in boreal forests. For example, Tymstra et al. (2007) used a wildfire model in Alberta boreal forest to illustrate that a 2X and 3X CO₂ scenarios may result in an increase of area affected by wildland fires of 12.9% and 29.4%, respectively. Changes in species composition can also be driven by scenarios of more frequent forest fires due to more severe droughts. This can result in the replacement of less drought-tolerant species with those that are more drought-tolerant and it can result in more simplified landscapes in terms of the dominant vegetation species (Liang et al., 2017).

Future increases in area burned by wildfire has the potential to negatively affect regional air quality. Liu et al. (2010) projected changes in global wildfire potential at the end of this century (2070–2100) using general circulation models (GCMs). They showed that wildfire potential will substantially increase in North America, South America, central Asia, southern

Europe, southern Africa, and Australia. The increase in wildfire risk is greatest in southern Europe and least in Australia. There is also a tendency for the length of the fire season to be a few months longer than it is now. The increased fire risk is due to both warming and drying of the climate. However, the relative importance of each of these factors varies from one region to another. Carvalho et al. (2011) estimated the future forest fire emissions over Portugal under the IPCC Special Report on Emission Scenarios (SRES) A2 scenario. Their results demonstrated that increased future forest fire activity will elevate O₃ concentrations and PM₁₀ by 2100.

Vegetation changes and a general increase in tree mortality are also projected in response to the increased drought severity forecasted by climate models, although uncertainties remain that may propagate from climate to ecosystem models (Girardin et al., 2008; Gustafson et al., 2015). In any case, the model simulations tend to agree with stronger future drought influences in forest ecosystems (Gustafson et al., 2015; Hlásny et al., 2014; McDowell et al., 2016), in which increased AED driven by stronger VPD could play a significant role (Restaino et al., 2016). Williams et al. (2013) suggested that for the southwest U.S. by 2050s the forest drought stress will strongly increase as a consequence of the VPD. They predicted that the water stress in forests by the 2050s could exceed that driven by the most severe droughts recorded in the past 1,000 years. Hanson and Weltzin (2000) modelled the response of U.S. forests to future climate scenarios and suggested a reduction in NPP as a consequence of reductions in leaf respiration and stomatal conductance and increased mortality of small stature plants. Dury et al. (2011) forced an ecosystem model with the A2 IPCC emission scenario to show that across Europe the NPP might increase in high latitudes and altitudes, but it might decrease in the Mediterranean region by up to 80%. The strong decrease in NPP is associated with increased summer dryness. Similar results were provided by Hickler et al. (2012) who projected the future distribution of potential natural vegetation in Europe using a generalized dynamic vegetation model. They showed widespread shifts from

forests to shrublands in Southern Europe as a result of drought. In the Mediterranean area, forest changes are predicted as response to more frequent drought conditions with a 30% growth reduction in some fir species (Sanchez-Salguero et al., 2017).

Increased mortality related to more water stress that drives more frequent vascular damages has been suggested in different studies. For example, Anderegg *et al.* (2015) indicated that climate models project that drought would exceed the current mortality thresholds in arid sites of United States by the 2050s. In Australia, Mitchell et al. (2014) suggested that under a scenario characterized by hot droughts, the frequency of drought events that cause significant tree mortality would increase from 1 in 24 years to 1 in 15 years by 2050. On the contrary, other studies have suggested that atmospheric humidity and increased CO₂ concentrations could counteract the negative effect of drought and warming on forest mortality (Liu et al., 2017), although a constant relative humidity scenario under global warming remains under debate (Vicente-Serrano et al., 2018).

In humid cold environments, the patterns are complex and it is difficult to assess the possible future role of droughts since scenarios tend to predict more precipitation in these areas. Schlyter et al. (2006) used different emission scenarios to model the response of boreal forests to drought in northern Europe. They found an increase in drought-related effects in this humid environment. Nevertheless, Girardin et al. (2008) found in Canada that a 2 x CO₂ scenario obtained from a process-based model resulted in a longer growing season that could counteract the expected negative influence of increased drought frequency and severity.

While all these projections agree that water stress will increase, vegetation decline and forest mortality must be considered with caution since the delineations between temporary and permanent drought effects are very diffuse in these natural systems. Natural communities are generally resilient to extreme events (Gazol et al., 2018). Studies have suggested that extensive dieback could be more of a regional process and that changes in forest composition will be a gradual phenomenon (Hanson and Weltzin, 2000).

Other future drought effects have also been discussed in the scientific literature. As has been shown in this paper, air quality is strongly influenced by drought conditions. Wang et al. (2017) applied a relationship between drought and air pollutants to examine how future projected changes in drought frequency will diminish air quality. They estimated that by 2100 ground-level O₃ will increase by 1.6% and PM_{2.5} will increase by 1.16% in the U.S. solely due to changes in drought frequency.

Drought also influences carbon and nitrogen storage in the soil (Batjes, 2014). Crowther et al. (2016) found future projected warming may cause considerable reductions in carbon storage in the high latitudes. Davidson and Janssens (2006) also determined that drought may accelerate carbon decomposition in the soil and plant-derived carbon at the same time. Matías et al. (2011) experimentally investigated the effect of three climate change scenarios on carbon (C), nitrogen (N), and phosphorus (P) concentrations in soil and microbial communities of three characteristic Mediterranean habitats, including forests, shrublands and grasslands. They found that increased drought scenarios are characterized by the accumulation of nutrients in the soil, which would increase the risk of nutrient loss by erosion processes.

The connections between future changes in drought severity and fauna are more complicated and less direct. Walls et al. (2013) developed a model to predict changes in wetlands and showed that precipitation changes could facilitate extinctions of species adapted to humid conditions and, ultimately, modify the composition of amphibian communities. Duncan et al. (2012) predicted the future effects of drought on ungulate populations in arid and semi-arid environments. They showed that predictive population models for grazing species show a dramatic increase of the probability of extinction under future emissions scenarios. Oliver et al. (2015) analyzed time series data on butterflies in Great Britain for simultaneous modelled responses to different climate conditions and showed that population extinctions could occur in 2050 under RCP8.5. Overall, all these projections must be considered

carefully given the uncertainty of the climate model projections as well as the internal aspects of the ecological and environmental models forced by the climate projections. There is a large and growing literature that indicates drought severity will increase in the future and it is reasonable to think that this will have a substantial influence on different environmental systems. It is particularly important to focus on the most vulnerable systems (regions with high human pressures, semi-arid areas and key regions with high biodiversity) to increase our understanding of the possible future environmental drought effects.

5. General remarks and conclusions

This paper provides a comprehensive review of the different environmental effects associated with droughts (Figure 1). These effects transcend ecological systems to affect the entire environment (e.g., including soil, water and air quality and geochemical processes). Drought is a major source of disturbance in environmental systems all around the world, regardless of whether these systems are located in arid or humid regions. It is common that drought causes a cascade of effects that will affect many different environmental systems in a region: soil processes, vegetation growth, wildlife, water quality, aquatic ecosystems, etc. This reinforces that, in addition to the classical distinction between four drought types (meteorological, agricultural, hydrological and socioeconomic), it is necessary to include a fifth one, which is much more complex and diverse: environmental drought. This provides a much more complete description of the nature of drought effects. The classical drought classification has focused on sectors with clear social and economic implications. New paradigms also consider the environment as a sector with crucial socioeconomic implications, since it provides several services to human societies. It is also quite relevant given the need for better conservation and management of environmental resources.

Here we have stressed that the majority of existing studies that examine environmental drought indicate that the effects are dominantly non-structural. The majority of environmental systems recover once moisture conditions return to normal. This means that

these environmental systems show strong resilience to climate variability. Of course, the velocity of the recovery varies across these systems (usually days or weeks for soil biochemical cycles to years/decades for wildland fire related droughts). Very few examples of permanent environmental drought effects have been found, and they mostly are related to vegetation and animal communities. For example, changes in community structure and diversity or extinctions in communities that are near their environmental limits or in highly disturbed landscapes.

Knowledge of current trends in environmental drought influences is quite limited given the lack of long-term studies. However, both observational studies and climate change projections indicate that drought severity is increasing in many regions of the world and it is projected to continue to increase through the end of the next century. These changes are driven not only by precipitation decreases, but also increased atmospheric evaporative demand. The current state-of-the-art climate models predict increased drought severity in large parts of the world and more regions affected by drought. Under these conditions, the current resistance of environmental systems to drought will be severely challenged. We do not know if the future will follow these predictions, but independent of them, it is necessary that through improved management and use of natural resources we enhance the resilience of environmental systems to drought.

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Figure Captions

Figure 1. Schematic diagram showing the general dimensions and complexity of environmental droughts.

Figure 2. Evolution of the global surface lands affected by drought conditions (12-month SPI/SPEI < -1.28, the expected return period of 1 in 10 years). Data source: <https://digital.csic.es/handle/10261/153475>

Figure 3. Spatial distribution of the magnitude of change in 12-month SPEI (a), SPI (b), and the difference (c). Dark gray areas in the plot of significant differences (d) represent areas in which the rate of change using SPI and SPEI are statistically significant different. Bottom plot (e) is a density plot showing the change in SPI and SPEI in global areas. Data source: <https://digital.csic.es/handle/10261/153475>

Figure 4. a) Spatial patterns of the magnitude of change (SPEI units per decade) in the 12-month SPEI using three emission scenarios by the IPCC. b) evolution of the global surface lands affected by drought conditions in the three models (12-month SPEI < -1.28, the expected return period of 1 in 10 years). Data source: <https://esgf-node.llnl.gov/projects/cmip5/>

Figure 5. Global trends (2000 to 2100) in the volumetric water content (VWC) of the top 1 m of the soil (% units per decade) using three emission scenarios by the IPCC. Areas in blue indicate locations where the soils are projected to get wetter and the areas in red indicate locations where the soils are projected to get drier. The left plot shows the ensemble mean from three CMIP-5 models (CanESM2, MIRCO5, NorESM1-M) run using RCP2.6 emissions scenario, middle plot shows the results from ensemble means from RCP4.5 and the right plot shows the results from ensemble means from RCP8.5 (data source: <https://esgf-node.llnl.gov/projects/cmip5/>).

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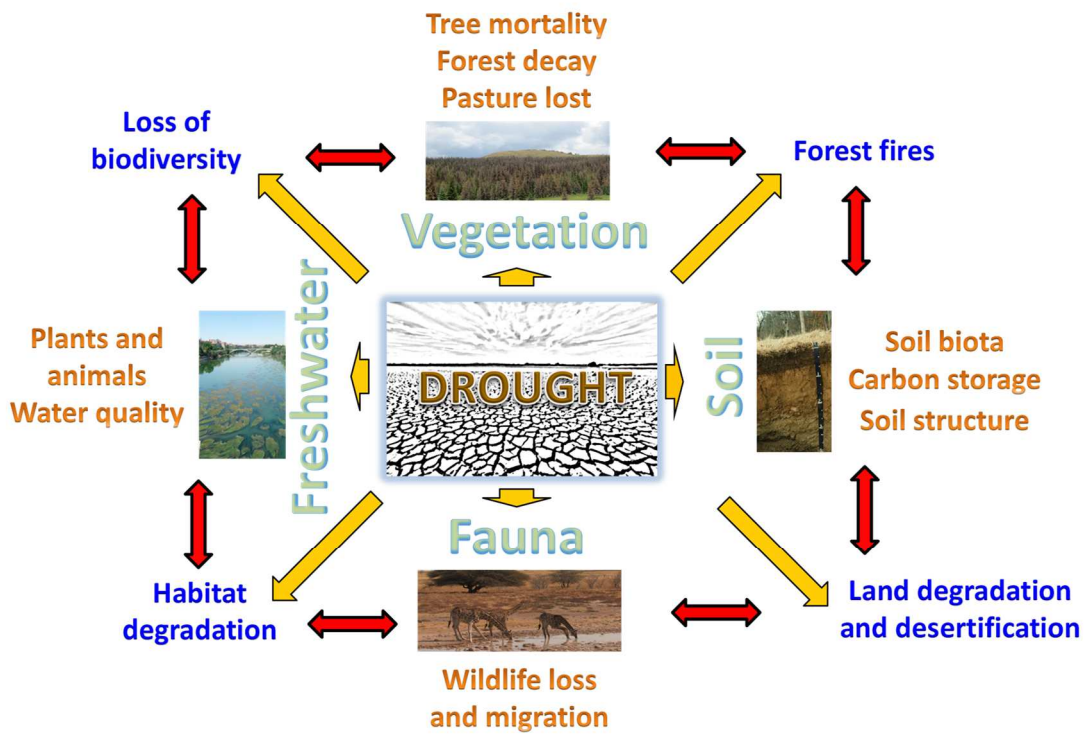
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Environmental effects of drought



TEMPORARY vs. PERMANENT

Figure 1:

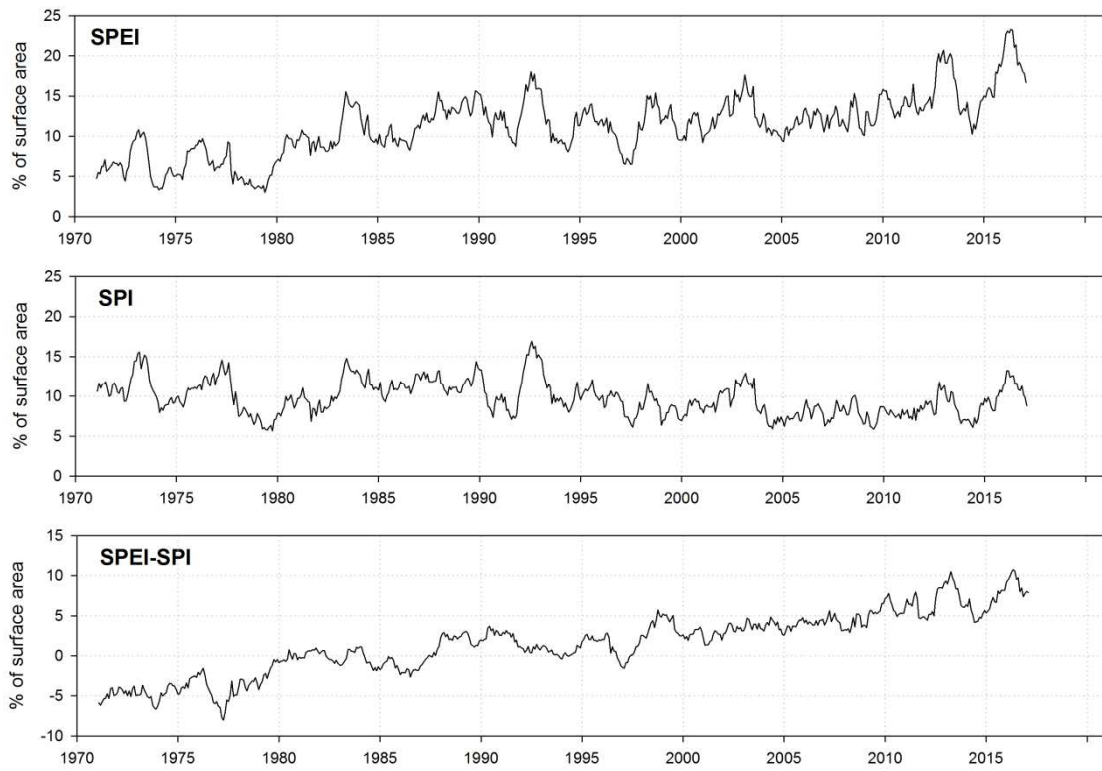


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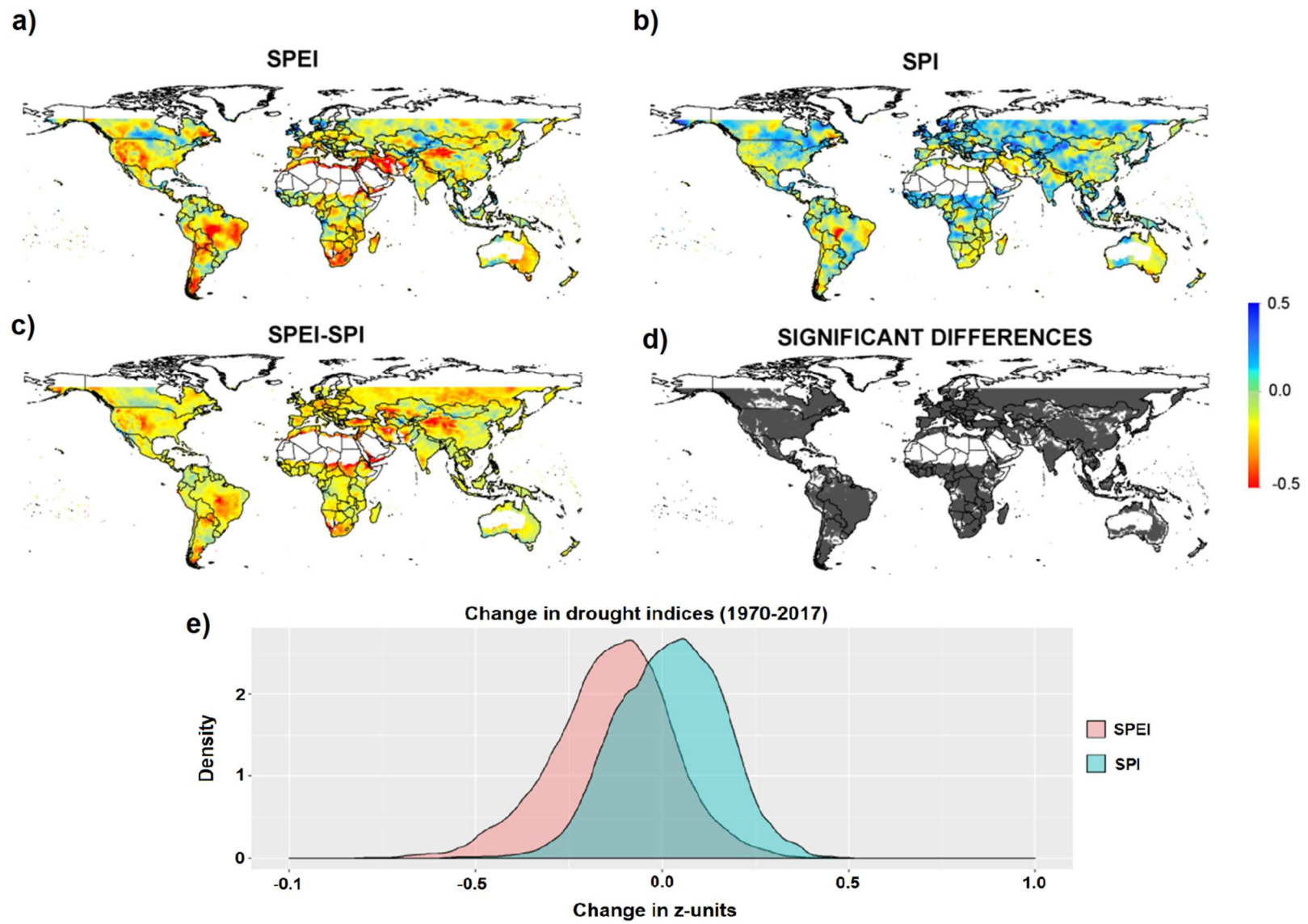


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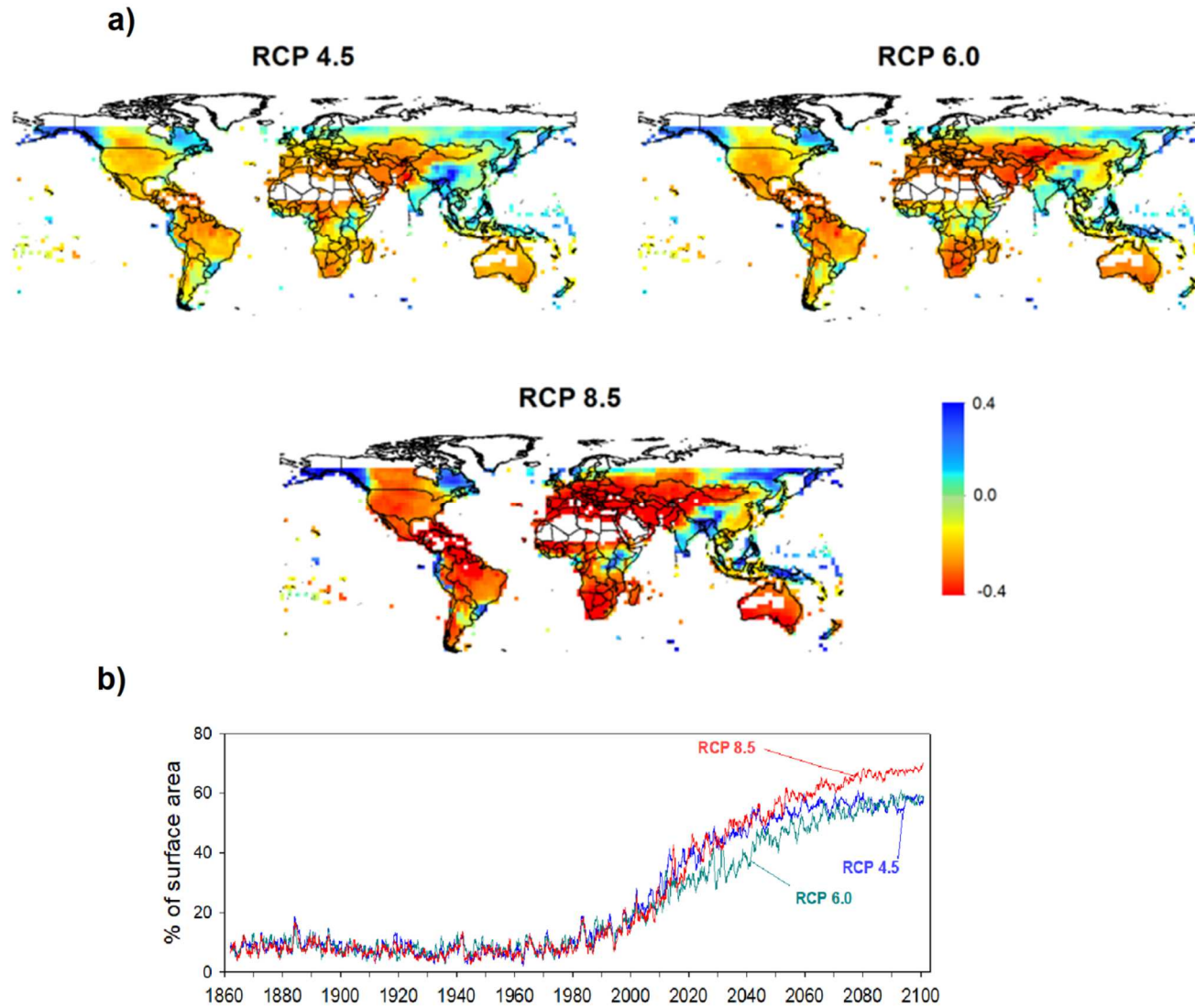


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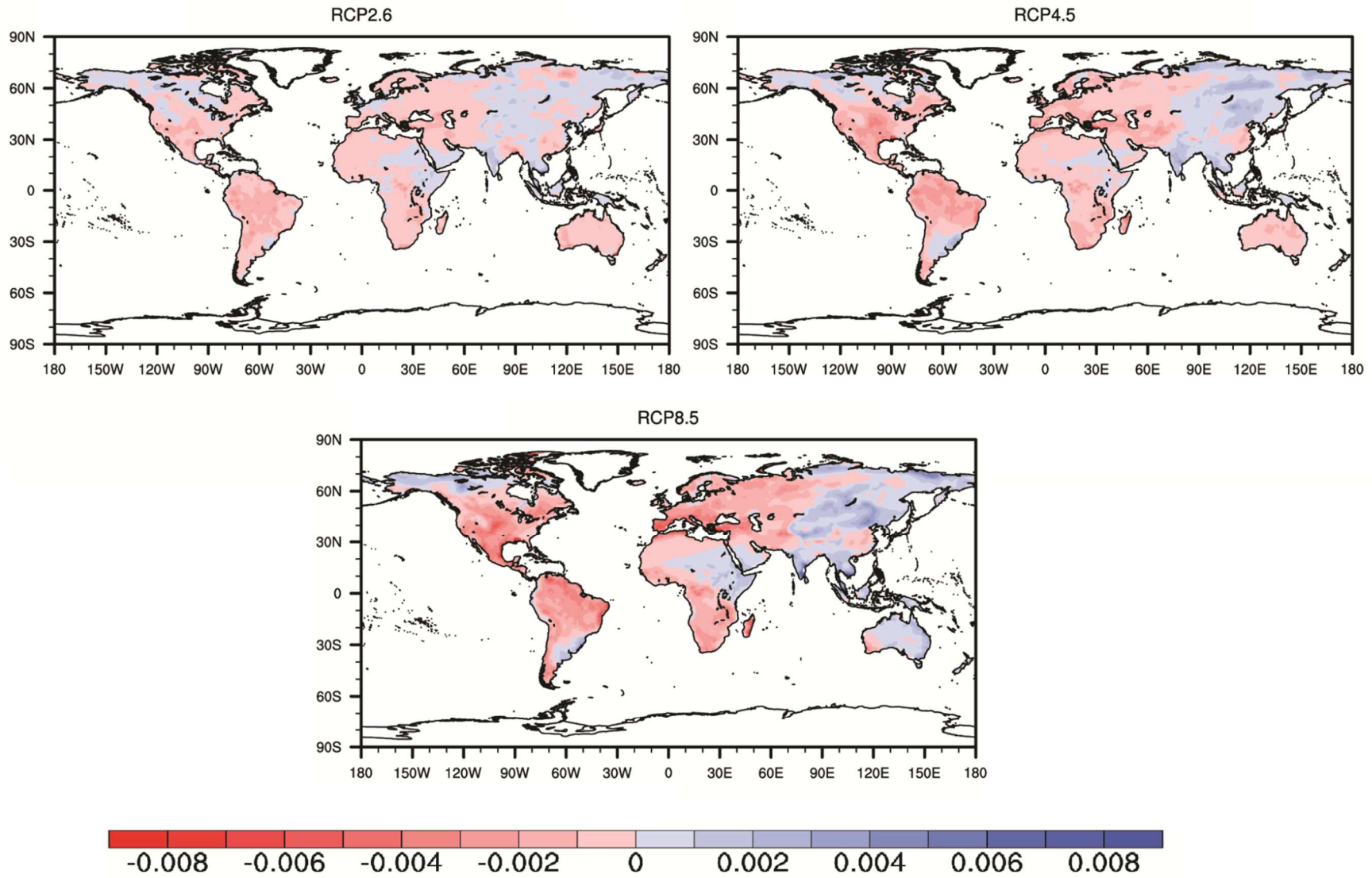


Figure 5: