More people, more fire, less water: exploring wildfire risks to water security in a changing world

by

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ABSTRACT

Water security is one of the main paradigms presently shaping global water governance. At its very core, water security aims at preserving freshwater resources from any form of risk, natural or human-caused, that could imperil or further delay the stability and the sustainability of societies and ecosystems. The adoption of the Sustainable Development Goals has acknowledged this paradigm as a key avenue towards the alleviation of poverty, the promotion of gender equality, the universal access to sanitation, and the protection of natural environments. However, reaching a universal water secure state, a harsh endeavor in itself, has been further complicated by the pervasive effects of global environmental change, among which widespread climate anomalies and population growth represent the main hurdles. The instability caused by environmental change has led to the emergence of risk situations that have never been encountered before and for which global analysis tools and governance strategies have yet to be designed.

Wildland fires are one of the most important natural drivers of vegetation dynamics at the surface of the globe. In many parts of the world, global environmental change has led to more conducive fire weather patterns combined with a higher ignition frequency due to landscape anthropization. This situation has increased the occurrence, extent, and severity of catastrophic wildfires in many areas critical for the provision of surface freshwater supplies to downstream human and natural communities. Although fire-caused alterations of the hydrological cycle have been recognized for a long time, the upsurge of extensive and severe blazes in many basins' headwaters has shed light on the exposure of downstream populations and aquatic ecosystems to post-fire hydrogeomorphic hazards, such as floods or nutrient pollution. Exposed assets are therefore made vulnerable to harmful consequences such as the degradation of environmental flows, the disruption of the drinking-water supply, or the destruction of infrastructures. The emergence of wildfire-related risks to freshwater

resources is thus becoming a new challenge to add to the long list of threats to water security, and solving this issue will not be done without innovative research efforts.

The research presented hereafter offers the first global exploration of wildfire risks to water security. The resulting work offers three main outcomes. First, it widens the water security paradigm by demonstrating the growing danger that wildfires represent to the freshwater supply. Secondly, it provides an efficient and highly flexible risk analysis framework to researchers, managers, and policy-makers involved in the resolution of water security matters and the design of disaster risk reduction strategies. Finally, it proposes a reflection on the deleterious, though often overlooked, emerging effects of global environmental change affecting the interactions between fire activity and the hydrologic cycle. This work will hopefully help to better guide global water governance by acknowledging the extensive and potentially dangerous effects of wildfires on socio-hydrological systems.

PREFACE

This doctoral research is presented as a paper-based thesis where chapters two, three, four, and five have been or will be published in peer-reviewed scientific journals. The following researchers have been involved in the present work as co-authors:

- Dr Mike D. Flannigan, research scientist, director of the Western Partnership for Wildland Fire Science at University of Alberta;
- Dr Marc-André Parisien, research scientist at the Canadian Forest Service;
- Dr Carol Miller, research ecologist at the Aldo Leopold Wilderness Research Institute;
- Dr Kevin D. Bladon, research hydrologist at Oregon State University;
- Dr Monica B. Emelko, research engineer at University of Waterloo;
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The second chapter has been published as Robinne, F.-N.; Miller, C.; Parisien, M.-A.; Emelko, M.B.; Bladon, K.D.; Silins, U.; Flannigan, M. A Global Index for Mapping the Exposure of Water Resources to Wildfire. *Forests* 2016, 7, 22. I designed the study and ran the analysis. All the co-authors assisted with the interpretation of results and the redaction of the manuscript.

The third chapter is currently under review for *The Science of The Total Environment* as Robinne, F.-N.; Parisien, M.-A.; Flannigan, M. D.; Miller, C.; Bladon, K.D. A global assessment of wildfire risks to human and environmental water security. I designed the study and ran the analysis. All the co-authors assisted with the interpretation of results and the redaction of the manuscript.

The fourth chapter has been published as Robinne, F.; Parisien, M.-A.; & Flannigan, M. D. (2016). Anthropogenic influence on wildfire activity in Alberta, Canada. *International Journal of Wildland Fire,* 25. Marc-André Parisien and I designed the study and ran the analysis. The co-authors were involved in the interpretation of results and the redaction of the manuscript.

The fifth chapter will be submitted to *Natural Hazards* as Robinne, F.-N.; Parisien, M.-A.; Flannigan, M. D.; Miller, C.; Bladon, K.D. Updating a global water risk index using current and future fire activity data. The co-authors are involved in the interpretation of results and the redaction of the manuscript.

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CHAPTER 1 - INTRODUCTION

1.1 OF WATER SECURITY

1.1.1 GLOBAL WATER FACTS

Life on Earth is intimately bonded to the presence of water, and water itself has often been compared to the blood of the planet (Falkenmark and Lundqvist 1998). Fueled by solar energy, the blue molecule follows a complex though well-understood journey known as the hydrological cycle: precipitation fall on lands, then run off to surface waterways or percolate to aquifers, which eventually reach the ocean where massive evaporation takes place, and finally, clouds form and the cycle restarts. The hydrological cycle allows for the movement of approximately 1,386 million km^3 of water on a planetary scale (Korzoun and Sokolov 1978), on which depend many other cycles balancing the global distribution of natural compounds, such as phosphorus or nitrogen, essential to the functioning of natural communities and human societies.

Out of this gigantic amount of water, only ~0.3% is readily available as surface freshwaters extractable from open water bodies (e.g. lakes, rivers) after running off land. It equates to an estimated volume of 45,500 km^3 of renewable freshwater (Oki and Kanae 2006). Renewable freshwater resources (RFWR) are defined as the amount of water recycled on a yearly basis over lands thanks to rain, evapotranspiration, and runoff processes (Oki and Kanae 2006). Terrestrial waters are the very part of the hydrosphere that is essential to the persistence of humankind and ecosystems. At a global scale, terrestrial ecosystems use nearly 30% of RFWR, whereas humankind uses less than 10% of the available resource (Shiklomanov 2000; Rockström *et al.* 2009), out of which 92% is used for agriculture and irrigation, 4.4% for industrial production, and 3.6% for domestic needs (Hoekstra and Mekonnen 2012). However, RFWR are unevenly distributed at the surface of the planet (Fig.1-1), and unequally used as well (Konar *et al.* 2016). The

Equatorial and sub-tropical belts are by far the main surface freshwater reservoirs of the planet, followed by temperate and northern latitudes. On the other end, many tropical and continental areas experience low to almost no surface water availability, making water resources particularly rare and thus precious.

The human-population is also unevenly distributed on the planet, unfortunately not

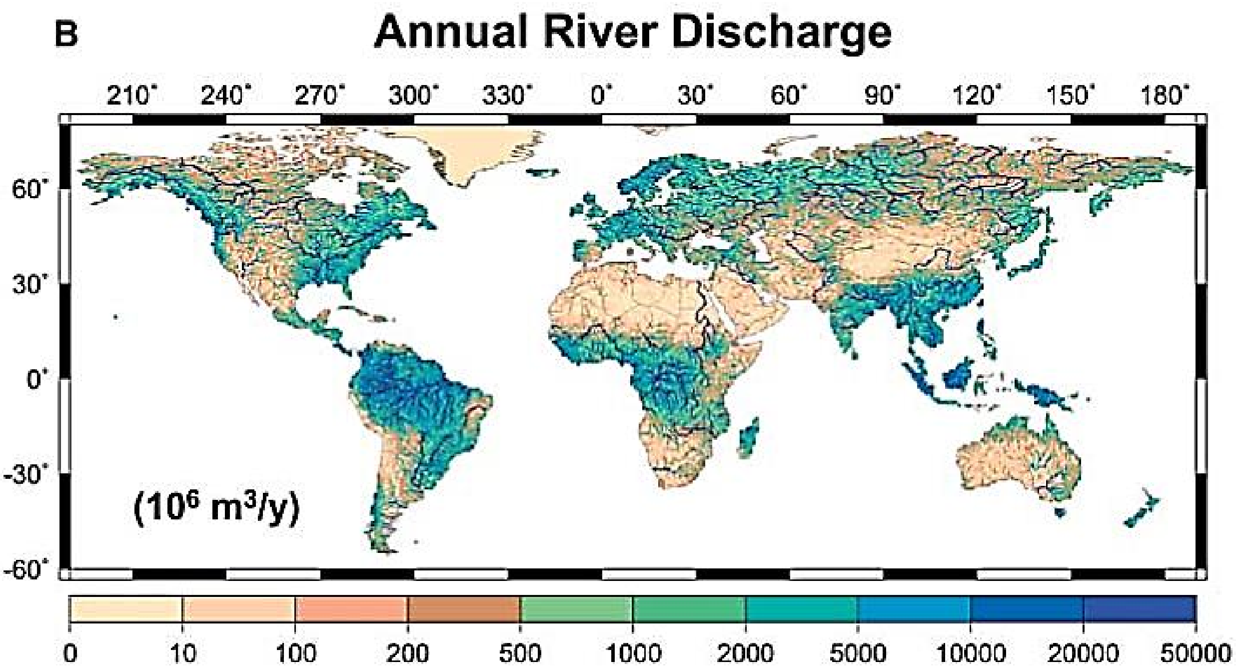

Figure 1-1: *Global river discharge. Published with permission from Oki et al. (2006)*

in a way comparable to RFWR. It is a geographic reality that often leads to water scarcity issues, aggravated by detrimental social, political, and economic conditions in many parts of the world. The absolute minimum drinking water requirement for an adult goes from 2.5 to 5 litres/day depending on climatic constraints, whereas the minimum amount of water to ensure a proper hygiene is estimated around 45 liters/day/person (Gleick 1996). However, according to the United Nations, approximately 2 billion people around the world live in water stress conditions, 2.4 billion people don’t have access to decent sanitation, and almost 700 million people lack an access to an improved water source (United Nations 2016), meaning that those basic water needs are far from being met globally. The level of socio-economic development of a country, as well as its role in the global economy, also drives the pressure put onto RFWR. According to Hoekstra and Mekonnen (2012), the water footprint of nations (i.e. the appropriated volume of RFWR) ranges from 1,250 to 2,850m3/yr/cap in developed countries to 550 to 3,200m3/yr/person in developing countries. Those figures imply a considerable variability between countries related to consumption patterns, the type and amount of crop production, technological and industrial capacities, and total population (Alcamo *et al.* 2008).

Tools to understands interactions between RFWR availability, use, and fate according to the social, economic, and political factors conditioning water-supply accessibility have their roots in the seminal work by Malin Falkenmark (Falkenmark 1977, 1979). These publications treated of the tight relationship bonding natural water resources and the safe development of mankind. She exposed in these studies what would become the very core of the water security paradigm.

1.1.2 WATER SECURITY AND RELATED NOTIONS

Water security concerns emerged in the early 1990's as a broad concept within a post-Cold-War context mixing issues of global stability, sustainability, and new interrogations on the future of a world entering climate change era. The concept of water security was at first either considered as a sub-topic of food security (Falkenmark and Lundqvist 1998) or as a vector of armed conflict (Starr 1991). In 2000, the Global Water Forum in The Hague, Netherlands, gave water security an institutional frame, defining its aim as "ensuring that freshwater, coastal and related ecosystems are protected and improved; that sustainable development and political stability are promoted, that every person has access to enough safe water at an affordable cost to lead a healthy and productive life and that the vulnerable are protected from the risks of water-related hazards" (World Water Council 2002).

This global endorsement gave water security legitimacy, transforming a broad concept into a paradigm that nowadays occupies the forefront of global water governance discussions (Cook and Bakker 2012). The water security paradigm also enforced the development of a large body of related nested concepts and paradigms. The diversity of those connected themes is illustrated with terms such as the human right to water, integrated water resource management, water scarcity, water crisis, virtual water, environmental flows, socio-hydrology, water footprint, etc. This incomplete list underlines the multifaceted nature of water challenges and the many possible avenues toward a "One Water" vision (Lall 2014) organized around the acknowledgement and the efforts to understand the 'global water system' (Alcamo *et al.* 2008). Many of those concepts emerged in the 1990s (Falkenmark 1997; Zalewski *et al.* 1997; Postel *et al.* 1998), but they now all gather under an overarching idea geared toward sustainable development and human security. Water security is now recognized as one of the main motors of the development of nations, without which economic growth is stalled and social stability is unachievable or compromised (Grey and Sadoff 2007; Sadoff *et al.* 2015).

Water security is at the core of the Sustainable Development Goals of the United Nations (United Nations 2016). Indeed, the achievement of each goal is related, in a way or another, to the guaranty of appropriate access to RFWR, thereby illustrating how pervasive the global water problem is (Srinivasan *et al.* 2012a). Although only two goals are explicitly dedicated to water (i.e., Clean water and sanitation, Life below water), goals of gender equality, hunger, or cities sustainability, for instance, will not be met without a reflection on the future relationship between humanity and water has not been undertaken (Bogardi *et al.* 2012; Grey and Garrick 2013). Such a necessity recently led to the emergence of socio-hydrological science, simply presented as the 'science of people and water' whose central goal is to address the coupling of anthropogenic needs for water with the hydrological cycle (Sivapalan *et al.* 2012), referred hereafter as socio-hydrosystems. Socio-hydrosystems are therefore viewed as complex dynamic systems where water security is dependent on past and future anthropogenic pressures on RFWR, themselves dependent on water resource availability. The intrinsic holistic nature of socio-hydrology offers an avenue to connecting researchers and managers, private and public stakeholders, and upstream and downstream users into inter-disciplinary science (Blair and Buytaert 2016). Such holistic approach has often been presented as the only way to address current and future water challenges (Falkenmark and Folke 2002; Pahl-Wostl *et al.* 2013; Vogel *et al.* 2015).

1.1.3 WATER SECURITY AND RISK

Historically, water has been a major control of the emergence and fate of human civilizations (Vogel *et al.* 2015). The location of many cities across the world, such as Mexico City, is historically linked to the existence of surface freshwater nearby to ensure drinking-water supplies and agricultural needs. Maintaining access to substantial amounts of surface freshwater supplies has often been a source of tension inside and between countries around the world. The Water Conflict Chronology List (Gleick 1993) provides a detailed review of 399 historical and modern events for which violence arose, directly or indirectly, from tensions linked freshwater accessibility. With 263 transnational lake and river basins across the world, tensions are common place among countries showing obvious discrepancies in the way they rely on available water for their development. The history of the management of the Nile River, shared by 10 countries but dominated by Egypt, or the Lesotho 'water coup' assisted by South-Africa in 1986 to ease water access, are two illustrations of water-related diplomatic and military combats.

Beside possible or existing diplomatic tensions, socio-hydrosystems also are vulnerable to extreme natural hazards and consequently to disasters. The excess or, on the contrary, the rarity of water is the source of risks taking a significant toll on world populations, environments, and economies. Over the period 2005-2014, floods have affected almost 900 million people across the world, killing nearly 59,000 and costing impacted societies nearly 343 billion dollars (IFRC 2015). A recent study estimates the global exposure of hydro-systems to fluvial and coastal floods around 46,000km², equivalent to $30US trillion (Jongman *et al.* 2012), a figure that will grow with sea level rising due to global change. Over the same period, droughts affected 535 million people, accounting for 20,000 casualties and costing around 65 billion dollars at a global scale (IFRC 2015). Global discrepancies in water security levels are underlined in many floods or drought risk assessments, pointing at the extreme vulnerability of populations in developing countries (95% of fatalities related to floods) and the economic vulnerability in developed countries (Kundzewicz and Matczak 2015; Carrao *et al.* 2016). Literature provides a plethora of examples showing the wide array of adverse effects triggered by extreme hydro-climatic hazards, from the destruction of agricultural yield to the displacement of the impacted population (Dilley *et al.* 2005; Yonetani 2014; Shi *et al.* 2015).

Drastic adverse changes in the quality of RFWR are a corollary of hydro-climatic extremes. Droughts or water scarcity issues force people to rely on unsafe sources of water to ensure their daily drinking and hygienic needs, leading to an increased exposure to pathogens or harmful pollutants present in higher concentration in low and stagnant flows (i.e. evapoconcentration) (Mosley 2015). Subsequent floods triggered by heavy rainfalls can then be a vector of pollution propagation downstream, although such process is dependent on a large set of environmental variables and could therefore show an important spatial variability (Cann *et al.* 2013; Mosley 2015). Those issues are usually amplified by human-made changes in land use – land cover that deteriorate the health of ecosystems, creating positive feedback loops further enhancing water quality issues (Foley *et al.* 2005). Shwarzenbach et al. (2010) point at the numerous anthropogenic sources of water pollution, arguing they have been impacting RFWR for the past five decades around the world, including pollution from mining activities, uncontrolled disposal of toxic waste, including radioactive compounds, and pharmaceuticals. Several studies point to the impact of anthropogenic development and regulation issues as a major factor of pollution threats to RFWR. The excessive use of chemicals in agriculture and in the industrial sector, deficient sewer systems and the lack of proper treatment facilities to secure sanitation, are issues that could be particularly acute in developing countries (Vörösmarty *et al.* 2010b; Hoekstra and Mekonnen 2012). The global burden of waterborne disease is estimated around 1.6 billion death every year, 99% of which occurring in developing countries, essentially because of poor sanitation conditions (Prüss-Üstün *et al.* 2008; Schwarzenbach *et al.* 2010). The impact of poor quality water also endangers freshwater ecosystems (Vörösmarty *et al.* 2010b; Collen *et al.* 2014), thereby threatening their capacity to provide services to societies (Green *et al.* 2015; McIntyre *et al.* 2016).

The concept of risk is thus an inherent part of the water security paradigm, as any natural, technological, or socio-economic hazard represents a threat to the progress towards a water-secure state (Norman *et al.* 2012; Hope and Rouse 2013). The Carbon Disclosure Project Global water report mentions a total of US$14 billion in the water-related loss for 600 major international companies, most of them (~50 to ~75%) declaring exposure to water-related hazards at some level of their business process (CDP 2016). Even in allegedly water secure countries, cases of water insecurity remain. The problem of unsafe water supply in northern Native communities in Canada has been an ongoing issue for more than a decade (Bakker and Cook 2011). At a global scale, water insecurity maintains or increase the poverty of nations (Brown and Meeks 2013). These situations underline the importance of risk assessment and management to guarantee long-term water security. Socio-hydrology will have a major role to play and will certainly become over the years the strong arm of water security initiatives (Srinivasan *et al.* 2017). As humanity maintains an ever-growing pressure on world's ecosystems, the list of potential threats to socio-hydrosystems keeps growing, leading to the emergence of new risks for which assessment methods remain to be developed (Kumar 2015).

1.2 WILDFIRE THREATS TO WATER SECURITY

1.2.1 GLOBAL FIRE FACTS

Wildfires are a major disturbance affecting ecosystems at a global scale, and could be considered by extension an important disturbing agent in socio-hydrosystems (Chuvieco *et al.* 2014). Wildfires have played a critical role over for the past millennia in shaping the pattern of global ecosystems, therefore acting as an 'evolutionary agent' (Bond and Keeley 2005; Pausas and Keeley 2009) of plant communities and natural landscapes. The world without fire would be significantly covered by more forested areas, with an estimated difference of more than 50% compared to present (Bond *et al.* 2005). Giglio et al. (2013) estimated the yearly average area burned at 348Mha at a global scale (Fig.1-2). Savannas are the type of natural ecosystem the most frequently burned, especially in Africa, whereas forest fires mostly occur in the Boreal part of the Northern hemisphere, and shrublands fire mostly happen in Australia. Although fires occur almost everywhere, global patterns of

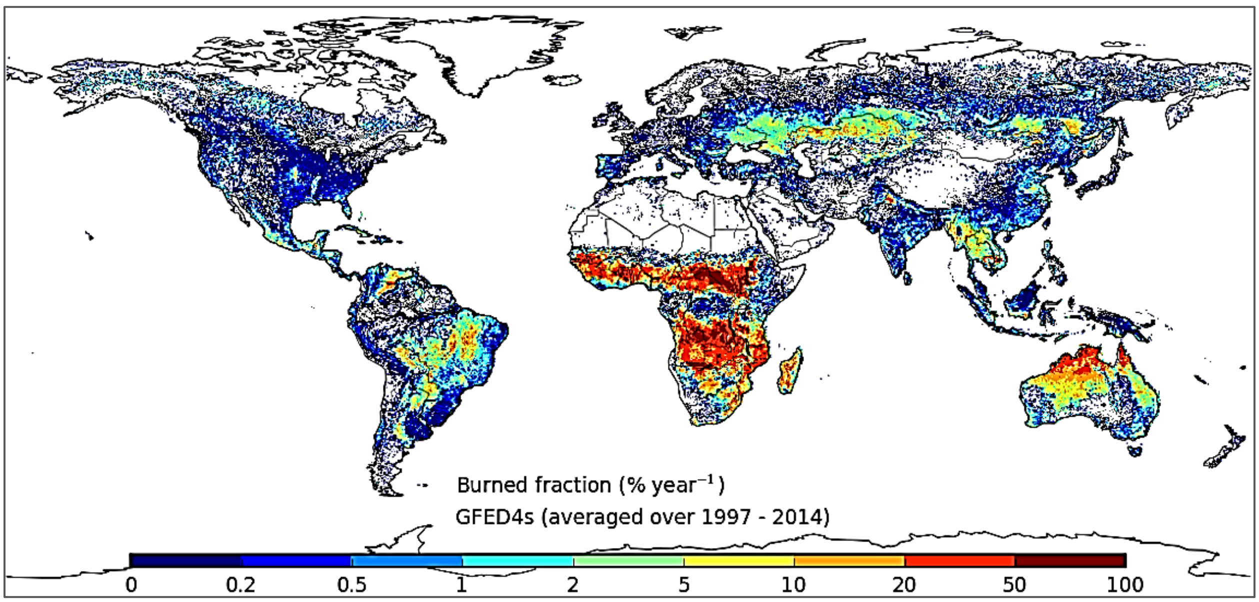

Figure 1-2: *Global mean annual burned fraction. Published with permission from Giglio et al. (2013)*

pyrogeography (i.e. the study of biotic and abiotic factors controlling fire activity) depends upon a fire-productivity relationship (Krawchuk and Moritz 2011; Pausas and Ribeiro 2013) controlled by prevailing climatic conditions (Whitman *et al.* 2015). In low-productivity ecosystems, the lack of biomass limits fires initiation despite fire-conducive weather conditions, as it is the case in the desert. Conversely, high-productivity ecosystems can be limited by the absence of fire-conducive weather, despite plentiful biomass, such as in Amazonia.

Fire activity is an essential motor of element cycling at a global scale. Carbon emissions from aboveground biomass combustion are estimated around 2Pg C yr^{-1}, roughly divided in half between natural and anthropogenic fires. Most of natural emissions come from savanna fires in Africa and anthropogenic emissions linked to agricultural and deforestation burning (Van Der Werf *et al.* 2010; Li *et al.* 2013; van Marle *et al.* 2017). Global emissions of natural and anthropogenic mercury from wildfires are estimated around 675Mg, or 8% of current global emissions, mostly due to boreal and tropical woodland fires (Friedli *et al.* 2009). Nitrogen emissions, including nitrogen-derived species such as ammonia or nitrous oxide, are estimated to be superior to 40Tg yr^{-1}, with ammonia representing 12% of this amount (Lobert *et al.* 1990; Vitousek *et al.* 1997). Phosphorus emissions from biomass burning are estimated around 5% of an annual 1.39Pg yr^{-1} (Mahowald *et al.* 2008). This latter reference also acknowledges the role of terrestrial sources, usually associated with sediment redistribution following soil and erosion, although the role of fire in global sediment dynamics have so far not received much attention (Filippelli 2008). Pyrogenic carbon, a source of organic carbon coming from the partial combustion of vegetation, has an annual production ranging from 116 to 385Tg yr^{-1} and may represent an important terrestrial source of pollutants (Myers-Pigg *et al.* 2015; Santín *et al.* 2015). It is also worth noting that recent efforts to include fire in Earth system modeling have shown that global fire activity has a noticeable role in the functioning of the hydrosphere, accounting for $0.6x10^3$ km^3 yr^{-1} or ~1.3% of the total global runoff during the 20th century (Li and Lawrence 2016).

Systemic interactions between fire, human, and ecosystems have been explained (Lavorel *et al.* 2007) and demonstrated in many parts of the world (Di Bella *et al.* 2006; Archibald *et al.* 2012; Ganteaume *et al.* 2013; Parisien *et al.* 2016). The pyrophilic primate hypothesis advanced by Parker et al. (2016) even suggests that the co-evolution of human and fire started 2 to 3 million years ago, a theory that underlines the importance of understanding coupled human-natural systems for the proper management of wildfire activity (Moritz *et al.* 2014; Miller and Aplet 2016). The Anthropocene has seen a shift in global fire activity driven by anthropogenic drivers (Pechony and Shindell 2010), to a point where Stephen Pyne, historian, qualifies natural fires of 'outliers' at a global scale (Stephen Pyne 2006). Anthropogenic development has become a tremendous force shaping fire regimes (i.e. the natural settings governing fire occurrence and effects in a given environment), extending the natural pattern of fire (Balch *et al.* 2017), even in locations showing no fire proneness (Cochrane 2003; Perry *et al.* 2012). The attempts to control fire, such as fire exclusion efforts during the 20th century, led to deep ecosystem transformation either due to a lack or an excess amount of fire (O'Connor *et al.* 2011; Parks *et al.* 2015). Over the period 2005-2014, wildfires have impacted almost 2 million people across the globe, with 732 casualties reported and $24 billion damages. Americas and Europe were by far the most affected, but casualties were almost equally spread through continents (IFRC 2015). Risks linked to wildfire activity are well recognized (Bowman *et al.* 2011; Gill *et al.* 2013), from direct burning of assets during extreme events (Bowman *et al.* 2017), to emergency evacuation (Beverly and Bothwell 2011), to health issues linked to smoke emission and toxic particle transport (Reisen *et al.* 2014; Johnston *et al.* 2015). High and very high vulnerability of human and ecosystems to fire has been estimated around 55% of the terrestrial landmass (Chuvieco *et al.* 2014).

1.2.2 POST-FIRE HYDRO-GEOMORPHOLOGY

Wildfires, particularly large and severe events, can strongly affect the hydrology and geomorphology of water basins, with consequences on water quantity and quality levels due to the alteration of the soil-vegetation complex (USDA 2005; Shakesby and Doerr 2006; Neary *et al.* 2009). In many cases, fires act as a control of low and high hydrologic flows, as well as of the total water yield of a watershed. They can also favor the occurrence of erosion episodes that redistribute sediments, debris, and chemical compounds along the hydrological network. It is, however, important to underline that despite documented trends, a significant geographic and temporal variability exists depending on the scale of work and the environmental conditions controlling the hydrology of a given hydrosystem, such as, but not only, pre-fire vegetation and soil conditions, as well as terrain characteristics and post-fire precipitation patterns (Moody *et al.* 2013; Murphy *et al.* 2015).

Reported changes in water quantity after a fire are linked to the reduction of the vegetation cover, possibly combined with the alteration of soil hydraulic properties. The disappearance of the vegetal cover generally leads to a decline in precipitation interception, as well as a reduction in evapotranspiration, leading to potential changes in the water balance that can favor the initiation of surface and sub-surface runoff, which eventually increases the annual water yield (Beeson *et al.* 2001; Nolan *et al.* 2014). Water repellency or hydrophobicity is a property that decreases the affinity of soil for water leading to low or absence of infiltration thus resulting in higher runoff. Water repellency can be a critical driver of increased post-fire water flows, as it prevents water from entering and percolating into the soil (Doerr *et al.* 2000). The loss of soil wettability, pore clogging by ash and soil crusting can also cause water repellency (Mills and Fey 2004; Stoof *et al.* 2016). This potentially significant increase in water running off into the stream network may cause greater and faster peak flows, whose erosive power can affect river banks and bed morphology, as well as facilitate debris-laden and woody debris flows (Cannon *et al.* 2001, 2008; Legleiter *et al.* 2003). Following a fire, seasonal flows also tend to be higher than average, as well as annual water yield, although regional and seasonal differences based on geographic specificities, such as climate or geologic substrate, have been reported (Hallema *et al.* 2017). In snow-dominated ecosystems, wildfires can influence the timing of high and low flows (Seibert *et al.* 2010). Indeed, several studies have shown that burned areas can experience larger snow accumulation combined with earlier spring snowmelt due to the reduced forest cover, therefore leading to higher solar influxes and heat absorption by charred material. When associated with lower water interception and lower water consumption from vegetation, the watershed can experience higher-than-normal flows during the spring freshet (Ebel *et al.* 2012; Gleason and Nolin 2016).

After a fire, changes in water quality are often associated with changes in water quantity, although studies reporting those effects have been so far geographically limited and may not represent the potential diversity of response, or the absence of response, of different hydrosystems. However, the often-documented increase in runoff velocity can provoke higher erosion events, sometimes drastic (e.g., rills, gullies, landslides), and can therefore facilitate the occurrence of debris flows (Arseneault *et al.* 2007; Neary *et al.* 2012; Jordan 2015). Higher loads of solids in the waterways can have important effects on fluvial hydrogeomorphology, with greater bank and bed erosion accompanied by higher sediment deposition and the appearance of log dams (Meyer and Wells 1997; Legleiter *et al.* 2003; Short *et al.* 2015). Fire also acts as a rapid decomposer and mineralizer of soil organic matter leading to a surge in the availability of labile chemical compounds, such as organic carbon and nitrogen, available for runoff and leaching (González-Pérez *et al.* 2004; Certini 2005; Wang *et al.* 2012). The alteration of the soil structure due to the heat release can lead to a diminution, or at least a reorganization of soil microdiversity (e.g., fungus, bacteria, insects) that usually ensures a steady recycling of nutrients and other chemical compounds, such as carbon, iron, or mercury (Neary *et al.* 1999; Mataix-Solera *et al.* 2009;

Giesler *et al.* 2017). In cases, high fire frequency may affect vegetation regrowth capacities and may lead to potential soil impoverishment due to repeated erosion and leaching (Mataix-Solera *et al.* 2011). The entrainment of debris and sediments can boost water turbidity, possibly resulting in a concentration of suspended solids that can favor the concentration of sediment-associated pollutants, such as nutrients, polycyclic aromatic hydrocarbons (PAH), and other trace elements such as iron or zinc, and finally dissolved organic carbon, all with a potential to alter water quality (USDA 2005; Bladon *et al.* 2008; Sherson *et al.* 2015). Noteworthy is the likely increase in water temperature due to the absence of shade provided by vegetation that therefore favor incoming light flux and thus temperature rise, which eventually can lead to lower oxygen levels (Tobergte and Curtis 2016). The deposition of smoke has also been reported as a source of higher chemical loads in surface waters (Spencer *et al.* 2003), however this aspect has so far received very little attention and may not represent a significant issue for surface waters (Spinks *et al.* 2006). Layers of ash appearing after severe fires also represent important sources of nutrients, particularly phosphorus, and toxic elements such as arsenic or mercury. In the case of an important runoff-erosion event, they can concentrate into surface water bodies and have a high potential to increase water turbidity (Bodí *et al.* 2014; Santín *et al.* 2015).

The bulk of post-fire hydrogeomorphic effects happen in the short term, although the temporal scale of post-fire effects is highly dependent on precipitation (Murphy *et al.* 2015). A majority of studies have reported effects affecting water basins after up to a decade, but it is not rare that post-fire impacts are not noticeable anymore after less than 5 years (Pierson *et al.* 2001; USDA 2005; Silins *et al.* 2009; Robichaud *et al.* 2016). The response of a watershed to post-fire effects through time seems highly dependent on regional landscape characteristics and its climatic regime, with rugged terrain combined with an alternation of dry spells and heavy rainstorms are more heavily impacted (Hallema *et al.* 2017). For example, Mediterranean areas are characterized by thin and stony soils, which thus make them less prone to post-fire erosion (Shakesby 2011). After the 1988 fire in Yellowstone National Park, incision effects in river channels were still occurring after more than a decade (Legleiter *et al.* 2003). In favorable sites, post-fire hydrogeomorphic effects may be noticeable for an even longer period. In Australia, the regeneration of Eucalypt species, whose young individuals show a high water demand, can diminish hydrologic flows for ~150 years after the fire (Kuczera 1987). Those cases exemplify the potential diversity of post-fire hydrogeomorphic responses and thus the need for further studies, particularly in hydrosystems that have not received much attention on that matter (e.g., in South America of Africa). Although those examples underline the difficulty to generalize these responses to any hydrosystem, some general trends, such as higher peak-flows or higher erosion susceptibility, are nonetheless deemed acceptable after several decades of dedicated field studies (Shakesby and Doerr 2006).

Documented post-fire hydrogeomorphic changes have mostly focused on surface and sub-surface processes, and their extended consequences on open water bodies. Studies looking at the potential impacts of fire on aquifer hydrology remain rare. Therefore, my research only focuses on surface water resources, referred to as RFWR. However, a few other studies have pointed at the potential hydrogeomorphic impacts of wildfires on aquifers –considered non-renewable or fossil water– and on coastal waters. In Australia and Portugal, severe wildfire activity has been linked to changes in the chemistry of percolated water (Mansilha *et al.* 2014; Nagra *et al.* 2016), with noticeable increases in sulphate and potassium concentrations, as well as PAHs, all being potentially harmful to human and ecosystems in high concentration. In California, fire occurrence in coastal watersheds have led to a significant increase of total suspended solids (TSS) and PAH (Stein *et al.* 2012), with measurable impacts on the nearshore environment (Bowen *et al.* 2015; Morrison and Kolden 2015). However, existing knowledge of post-fire hydrology is, to date, spatially limited to a few countries in which wildfire science is well developed, making post-fire effects in many ecosystems of the world mostly speculative.

1.2.3 IMPACTS OF WILDFIRES ON SOCIO-HYDROSYSTEMS

The role of naturally vegetated areas in controlling the water cycle, and thus the water supply, has been known for a long time (Ellison *et al.* 2012). Their importance in the achievement of water security as providers of water-related ecosystem services has gained in interest (Creed *et al.* 2016). A growing number of publications particularly points at the importance of ecosystem services provided by forests and thus advocate for a greater

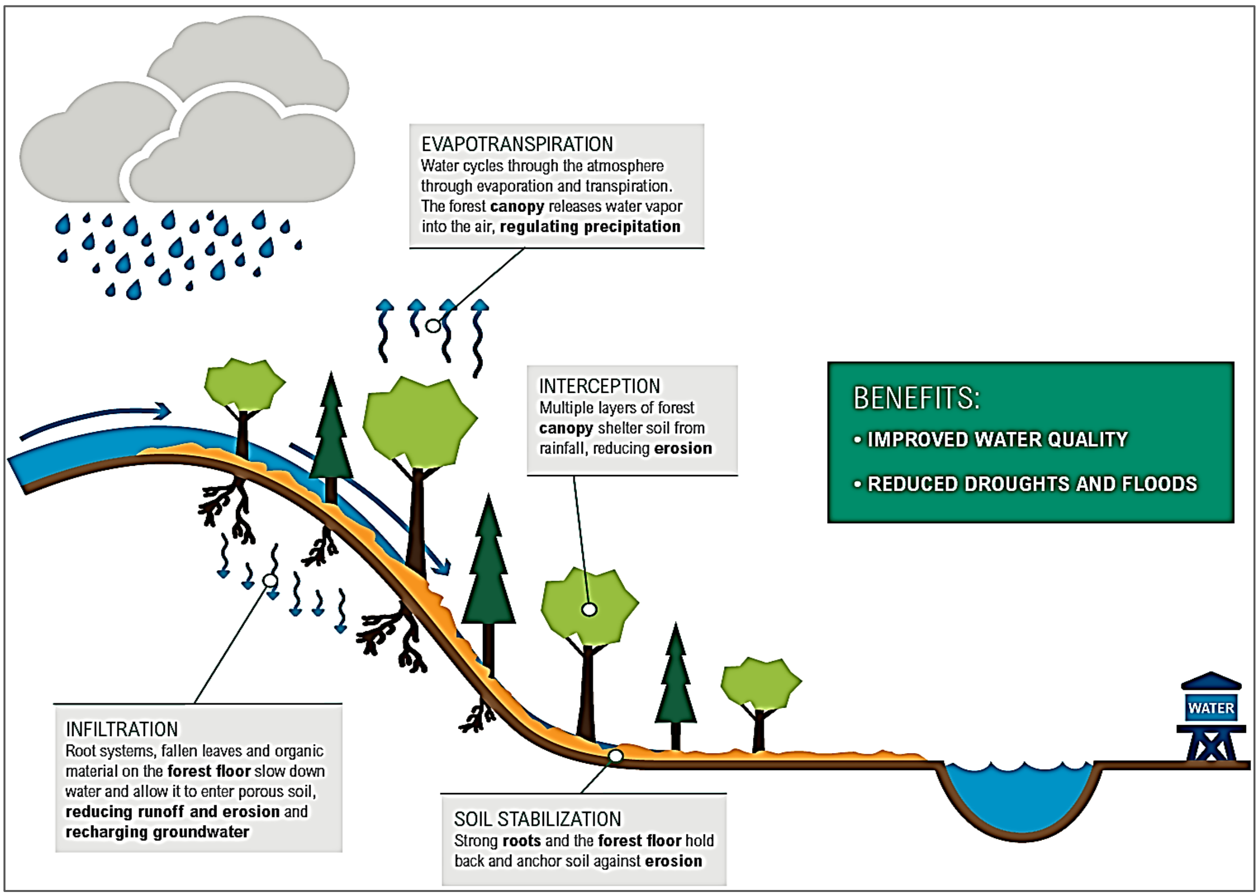

Figure 1-3: *The importance of forests for water security. World Resource Institute (2016)*

respect of natural areas regarding the provision of safe water and the protection against hydro-meteorological events (e.g. rainwater storage capacity preventing floods). Wildfires can impact socio-hydrosystems through the degradation of ecosystem services provided by forests, which must be seen as an extensive network of water supply infrastructures, also called natural capital, providing large storage capacities, filtration and purification functions, and flow regulation facilities, on top of regulating water vapor transfers (Fig.1-3) (Brauman *et al.* 2007; Neary *et al.* 2009; Ellison *et al.* 2012). In the case of a massive fire, the removal of vegetation and organic matter by the flames, along with the often-observed degradation of soil properties through heat transfers may severely impair the provision of those water services (Fig.1-4) and reduce RFWR treatability, thereby creating potential

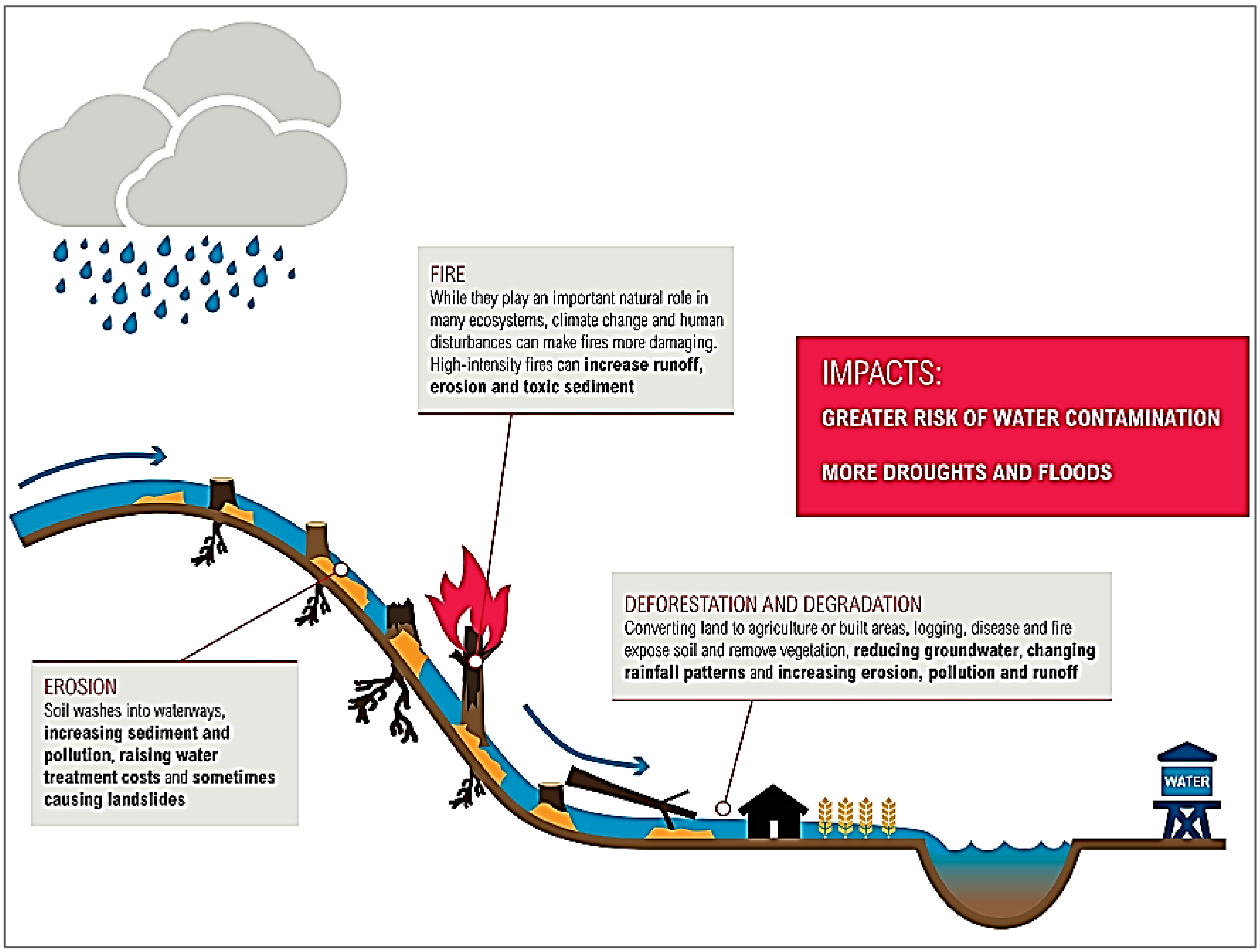

Figure 1-4: *Fire effects on forest water services. World Resource Institute (2016)*

threats to the provision of safe water to socio-hydrosystems (Certini 2005; Smith *et al.* 2011; Dahm *et al.* 2015).

One of the first often overlooked effects of massive fires on socio-hydrosystems is the potential inability of water infrastructures to operate properly when the flames threaten urban environments, either through to direct damages to water-distribution infrastructures or indirect impairments such as power failure or the impossibility for operators to access the plant. Ho Sham et al. (2013) have listed the different factors that, if impaired, can lead to

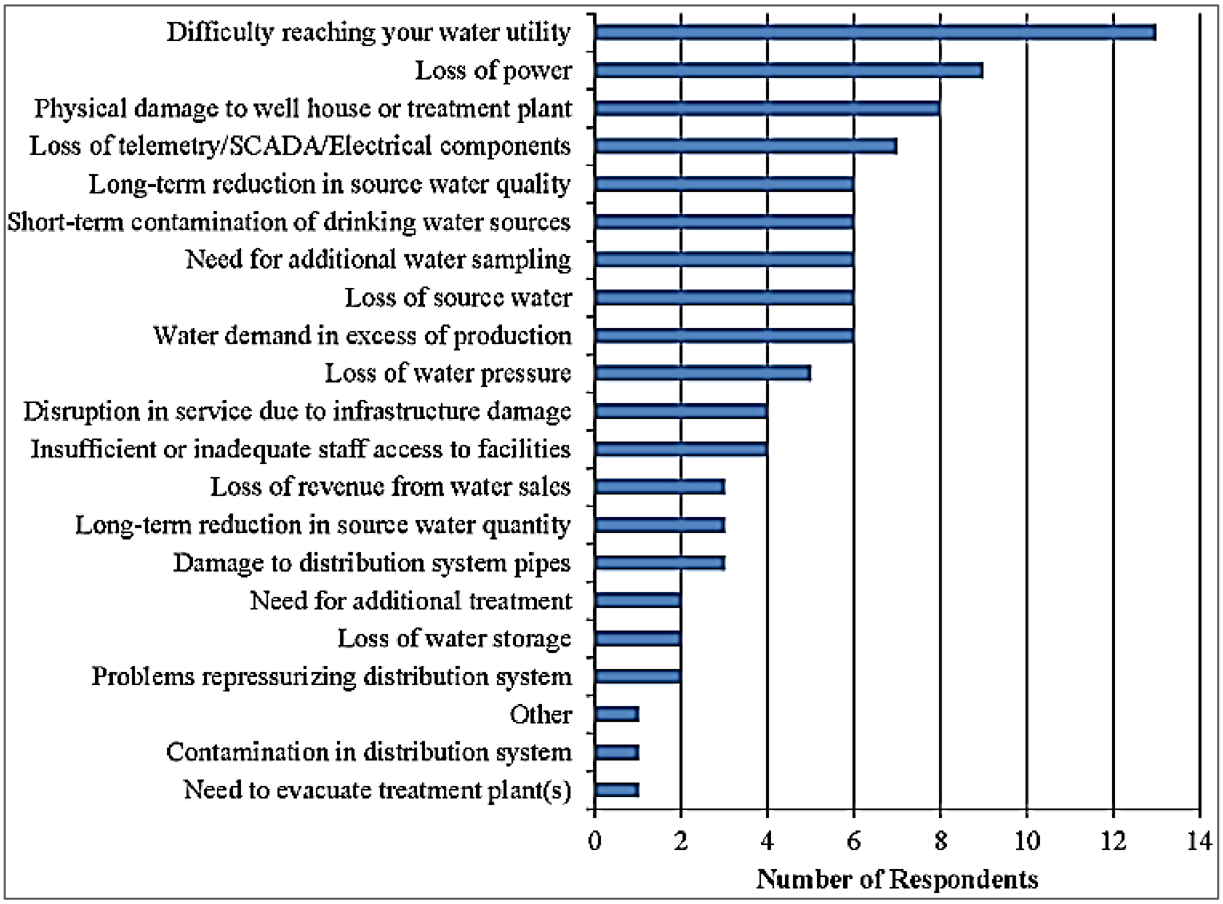

Figure 1-5: *Damages sustained by drinking water utilities during a wildfire. Ho Sham et al. (2013)*

the collapse of the water distribution system. Such concerns have been well illustrated during the 2017 fires in Chile, where many water distribution infrastructures were damaged by the flames or shut down due to the collapse of the power grid (International Federation of Red Cross and Red Crescent Societies 2017). Moreover, wildfires happening in wildland-human interfaces may even have a greater impact on RFWR as post-fire runoff can mobilize anthropogenic chemicals resulting from the combustion of human infrastructures or from burnt soils polluted by industrial activities (Steiniger and Hay 2009; Burke *et al.* 2013).

The possible amount of post-fire hydrogeomorphic hazards upstream of socio-hydrosystems can provoke a broad range of consequences endangering the water supply of downstream ecosystems and human communities. In this respect, wildfire must be seen as a source of cumulative effects (Stonesifer 2007). A wildfire occurring in a healthy watershed might be buffered by inherent resilience capacities, whereas a fire in an already disturbed watershed may aggravate water availability and quality on top of existing issues (Zhang and Wei 2012). Riverine and lacustrine ecosystems are particularly exposed to post-fire changes in hydrogeomorphology, with associated possible food chain shifts in aquatic environments, as presented in Section 1.2.2 (Beakes *et al.* 2014; Emelko *et al.* 2015). Those effects are far from benign when one takes into account the importance of healthy environmental flows for the preservation of aquatic ecosystems, considering that 90% of the world relies to some extent on freshwater fisheries for their livelihood or recreation (Venn and Calkin 2011; McIntyre *et al.* 2016). Although this research focuses on the adverse effects of wildfires on RFWR, it is important to underline the necessary ecosystem functions fulfilled by fires, including benefits for riverine ecosystems. Wildland fires help to maintain major nutrient cycling (e.g., carbon, nitrogen, phosphorus) and large-scale hydrological fluxes, to rejuvenate landscapes, and to promote habitat diversity (Lane *et al.* 2006; Arseneault *et al.* 2007; Klose *et al.* 2015; Tobergte and Curtis 2016).

When it comes to fire-affected water resources supplying human communities, several studies in North-America and Australia point at the potential threats linked to an increased probability of upstream mass movements, a surge in the amount of coarse woody debris, and higher and faster peak flows (Cannon *et al.* 2008; Seibert *et al.* 2010; Jordan 2015). The combination of those factors can generate landslides and flash floods with critical damaging power to anthropogenic infrastructures (Jordan *et al.* 2006; Cannon *et al.*

2008). Moreover, waters heavily loaded in suspended solids can run into man-made reservoirs, which can represent an important source of drinking-water and a source of renewable energy for power generation (Reneau *et al.* 2007; Sankey and McVay 2015). The degradation of reservoir's water supply can impair water treatability and can possibly force hydrosystems to restrict water use, as shown by the necessity to build an additional drinking-water treatment plant in the Australian Capital Territory to restore full water-provision capacity (AFAC 2017). The surge of sediments may also compromise the lifespan of the reservoir, and the cost of sediment removal is prohibitive, as demonstrated by the $30-million dredging operation conducted by Denver Water after the Hayman Fire (Moody and Martin 2004; Denver Water 2010).

The second main potential post-fire issue is related to the quality of RFWR, which influences its treatability and therefore the time and costs associated with producing drinking-water. The probability of damages to water intakes or filter clogging due to the amount of debris is of critical concern (Ho Sham *et al.* 2013). An eventual and sudden increase in turbidity, debris, sediment, and chemical concentration (e.g., nutrients, trace elements) after a fire may pose serious challenges to water providers. First of all, in the case of higher-than-normal phosphorus and nitrogen inputs may lead to the eutrophication of reservoirs (Smith *et al.* 2011). Moreover, difficulties to properly treat water resources contaminated by high loads of nutrients may happen, with potential consequences on water odor, color, and taste (Emelko and Sham 2014). The potential for a higher concentration of PAH and DOC post-fire may strain conventional treatment capacities and force operators to use greater amounts of chemicals used to treat water. Such scenario can theoretically lead to the appearance of carcinogenic disinfection-by-products affecting drinking-water reliability, although no case has ever been reported (Emelko *et al.* 2011). RFWR contamination by chemical compounds or suspended solids can therefore threaten the efficiency of common water treatment operations, to a point where the local water distribution system may not perform at its full potential and where authorities must rely on external water sources (Santín *et al.* 2015). However, in spite of the numerous threats to public health exposed above, associated to a growing number of documented challenges to water-treatment capacities, it is paramount to stress that there is no evidence that "fire-polluted" water has ever been distributed to communities. Moreover, to my knowledge, there is no study on the impact of wildfires on industrial and agricultural water uses, although a brewery reported the impossibility to run its facility due to the ash contamination of the River La Poudre (The Associated Press 2016), and the US government advices farmers to test surface waters coming from burned watersheds (McDonald and Doan-Crider 2014).

Wildfire impacts to RFWR have been known for several decades, although their acknowledgment as a source of risk to water security is rather new and has never been formally defined. Therefore, the research presented hereafter relies on the concept of the wildfire-water risk, which has been defined as the potential harmful effects of wildfire activity on water quality, quantity and seasonality; in proportions that can impair the freshwater supply to downstream human and natural communities (Robinne *et al.* 2016). There are many recent examples illustrating the WWR: the 2002 Hayman Fire in Colorado, USA; the 2009 Black Saturday bushfires in Victoria, Australia; the 2013 Rim Fire in California, USA; and the 2016 Horse Creek Fire in Alberta, Canada. All these fires triggered significant real-time and post-fire concerns as to the consequences on RFWR, many of them still measurable (Stevens 2012; Feikema *et al.* 2013; Wang *et al.* 2015). As fire is a quasi-ubiquitous phenomena as the global scale and that water-treatment capacities are far from being even across the world, the number of cases of water resources polluted by wildfires that are left unreported is likely many times more common that what we know. This underlines existing knowledge limitations to local or regional socio-hydrosystems in "fire-wise" countries, such as the USA or Australia where an important amount of research on this topic has been done over the past years. In this respect, the wildfire-water risk proposes a generalized vision of interactions between fire activity and water resources in socio-hydrosystems that could be applied in any geographical context. It places water services from forested areas at the center of the rationale, which will help to understand the effect of management policies on current WWR levels and to envision the future of the risk to our socio-hydrosystems.

1.3 GLOBAL CHANGES AND ENVISIONED IMPACTS

As the overarching human imprint on the biosphere, referred to as the Anthropocene (Crutzen and Stoermer 2000), is now supported by univocal scientific evidence (Waters *et al.* 2016) and that the rate of change in the global climate has been directly linked to anthropogenic disturbances (Rosenzweig *et al.* 2008), it is impossible to conduct a research on natural hazards and risks that would ignore global environmental change. The world is now fighting a decisive battle to understand and respond to the consequences of anthropogenic global shifts in the Anthropocene, with the hope that the Earth system remains a safe place for human development and the biosphere in general (Rockström *et al.* 2009). Although climate change has received most of the attention, many other sources of alteration have been identified (Fig.1-6), with particular concerns related to population growth and environmentally destructive consumption patterns (DeFries *et al.* 2004; Lambin and Meyfroidt 2011; World Economic Forum 2017). Besides the disruption of our climate, an ever-growing population combined with environmental degradation, wealth inequalities, and peace instability will certainly have effects on global fire activity (e.g., increasing ignition density) and socio-hydrosystems (e.g., increasing water consumption). The interconnected character of the modern world implies that the likely changes in the patterns and the levels of the wildfire-water risk have to be envisioned at a global scale (Helbing 2013).

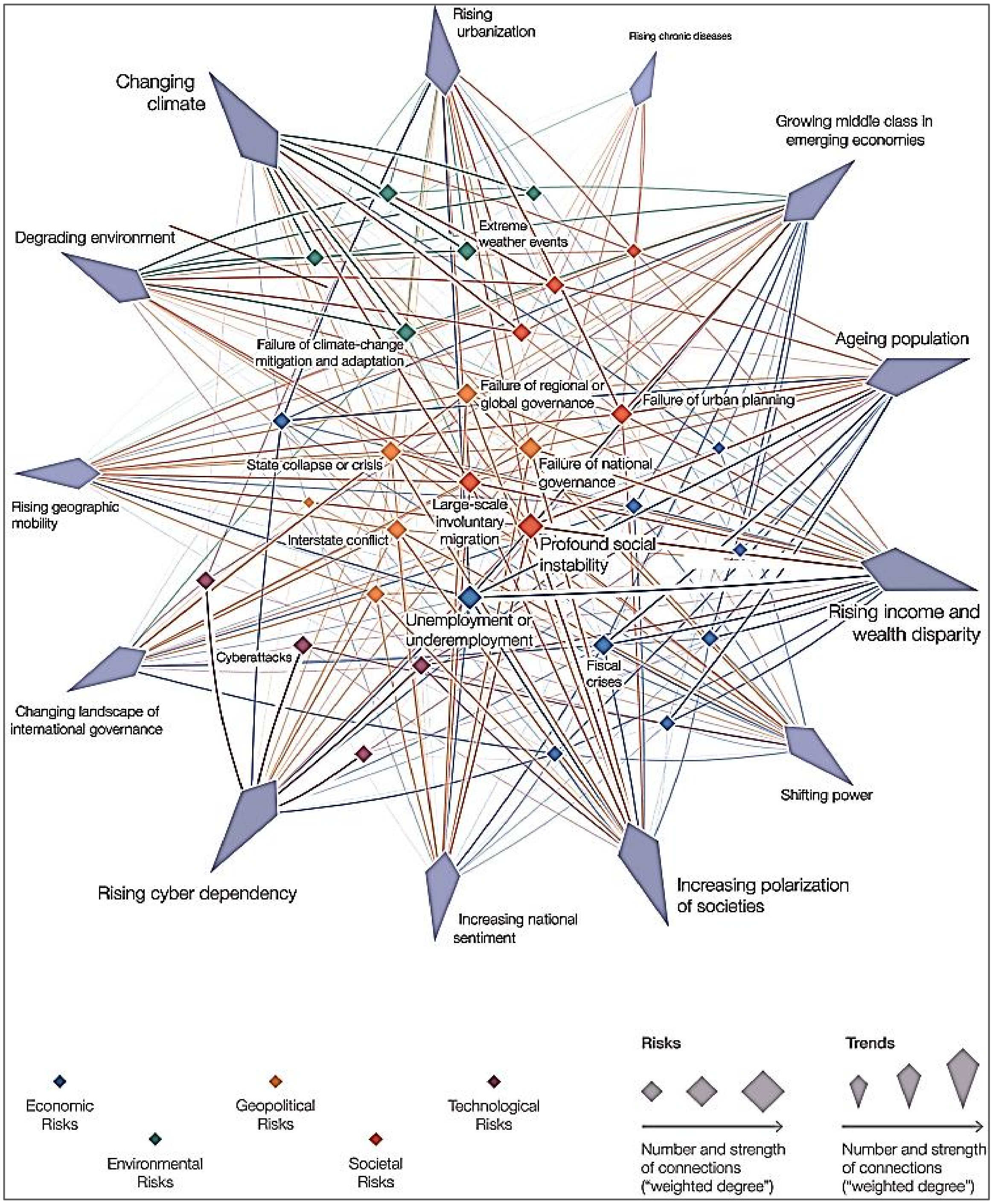

Figure 1-6: *The multiple facets of global change. The World Economic Forum (2017)*

1.3.1 THE WATER CYCLE AND WATER SECURITY

Global environmental change is likely to have a pervasive effect on the future of water resources, and the worldwide water crisis the world is currently experiencing will certainly be exacerbated (Rockström *et al.* 2014). From a climatic standpoint, the intensification of the water cycle (Huntington 2006) will modify the natural alternation of dry and wet seasons, causing more droughts and extreme precipitation (Mann *et al.* 2017), thereby heavily disrupting the reliability of the water supply. The availability of freshwater

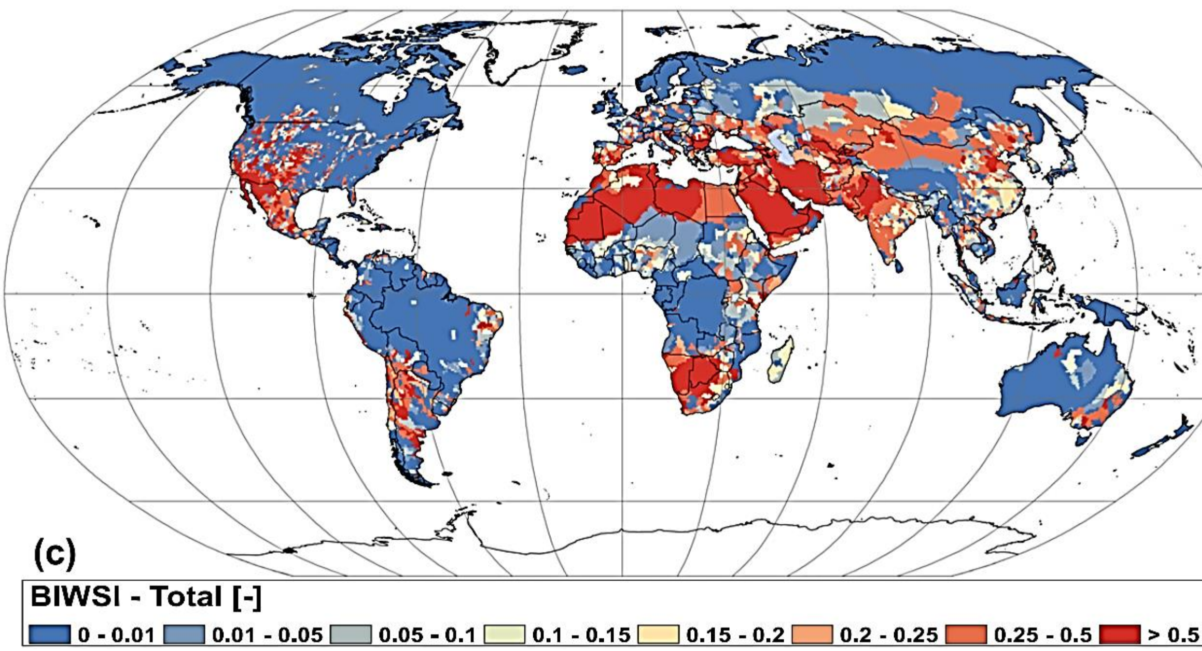

Figure 1-7: *Blue water sustainability indicator for 2069-2099. Higher values represent lower sustainability capabilities. Published with permission from Wada & Bierkens (2014)*

resources highly depends on the health of headwaters, a role often fulfilled by mountainous areas (Viviroli *et al.* 2007). The expected rise in temperature, the decrease in precipitation, and the shifts in plant functional types may lead to changes in snow accumulation, thaw timing, and eventually water yield. The impact on the total water discharge of world river basins will make it more difficult to populations to rely on a stable source of water (Viviroli *et al.* 2011; Caldwell *et al.* 2016). The predicted rise in the occurrence of droughts might diminish basin runoff, which would lead to lower-than-usual flows (Döll and Schmied 2012).

The following increase in water scarcity, with more than 50% expected increase in world population exposed, represents critical challenges for food and water security (Mancosu *et al.* 2015; Veldkamp *et al.* 2016). Changes in temperature will impact the thermal regime of water, and a coupling with lower water flows will threaten water quality and impact the aquatic trophic chain with higher concentration of potentially infectious organisms (Whitehead *et al.* 2009; Van Vliet *et al.* 2013). Overall, current projections of ongoing climate change consequences on water resources point at serious concerns for the future of the water supply (Delpla *et al.* 2009).

From an anthropogenic standpoint, the greatest ongoing challenge lies in the water consumption patterns of humanity. Several studies suggest that the increase of consumptive water use is at least equivalent, if not more prominent in some parts of the world than the disturbance of annual flows linked to climate change (Vörösmarty *et al.* 2007; Sterling *et al.* 2012; Murray *et al.* 2012), and that those effects tend to interact and worsen water supply issues (Veldkamp *et al.* 2015). As the world population is predicted to reach 10 billion individuals by 2050, subsequent changes in land use and land cover necessary to sustain and feed humanity will intensify water-related issues (Foley *et al.* 2005). Moreover, global urbanization dynamics will add to this existing pressure. Already 80% of the largest world cities rely on RFWR, and uncontrolled urbanization in developing countries represents significant water management challenges (Cohen 2006; McDonald *et al.* 2014; Wada and Bierkens 2014; Ceola *et al.* 2015).

The way forward to achieve water security lays in the widespread adoption of a socio-hydrologic approach in order to better understand the water problems at stake in different parts of the world (Sivapalan *et al.* 2014), problems that mainly concentrate in developing countries (Hope and Rouse 2013; Vogel *et al.* 2015). However, the chronic lack of hydrological data (Vörösmarty *et al.* 2001) remains a major limitation to drawing global water policies and to initiating targeted management actions (Vörösmarty *et al.* 2001;

Global Water System Partnership 2013; Garrick and Hall 2014). The impact of those changes are already apparent in many parts of the world (Foti *et al.* 2014) and innovative solutions for global water security are more than ever needed.

1.3.2 PYROGEOGRAPHY AND WILDFIRE HAZARDS

If water occupies the forefront of the concerns related to global environmental change, many recent events and other environmental issues have underlined the relationship between fire and humanity, as well as the vulnerability of human societies to severe blazes. From a climate change standpoint, modifications of global climate patterns are expected to affect the future distribution of fire activity, with a trend towards more fire-

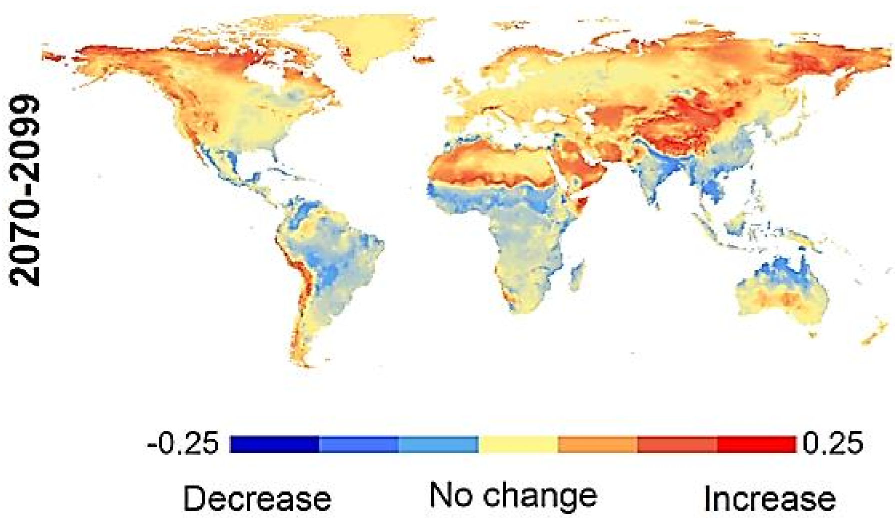

Figure 1-8: *Mean change in global fire activity compared to present. From Moritz et al. (2012)*

conducive weather conditions (Flannigan *et al.* 2009; Flannigan *et al.* 2013). Future climate will likely be the source of more frequent dry spells, along with more lightning activity, impacting fire activity (Littell *et al.* 2016). Despite the remaining uncertainty in the future directions of wildfire activity (Bowman *et al.* 2014), large-scale pyrogeographic studies agree on an increase of fire occurrence in mountain ranges and forested areas of the northern hemisphere (Krawchuk and Moritz 2011; Moritz *et al.* 2012). However, several global studies have predicted a decrease in area burned (Krause *et al.* 2014; Lasslop *et al.*

2014) due in part to changes in the availability of fuel with more constraining climatic conditions for plant growth (Pausas and Ribeiro 2013; Batllori *et al.* 2013; Martinez-Vilalta and Lloret 2016). On the other end, many studies report future increase in area burned (Bradstock *et al.* 2009), along with an increase in fire severity (Flannigan *et al.* 2009). Many ecosystems, especially in the circumboreal forest, are expected to be more impacted by wildfires, with an increase in carbon emissions due to the high carbon loading of peatlands (Turetsky *et al.* 2014).

From an anthropogenic standpoint, global wildfire activity will suffer an amplification of fire-conducive conditions comparable to future human-caused water issues. As shown in many studies, the influence of humans on natural fire activity is a strong environmental control that makes it difficult to decipher anthropogenic from other natural effects (Parks *et al.* 2015; Parisien *et al.* 2016). Landscape anthropization in many parts of the world due to population growth altered the environmental niche of fire (Balch *et al.* 2017), and anthropogenic controls on fire activity, such as the length of wildland-urban interface or the density of roads, are expected to be more influential (Knorr *et al.* 2014), continuing a trend already observed (Archibald *et al.* 2012; Bowman *et al.* 2017). As humans keep spreading in the landscape, a transition towards an anthropogenic fire regime (i.e., a fire regime where human use of the landscape is the main control of fire occurrence and spread) (Guyette *et al.* 2002) may initially lead to an increase in fire frequency and area burned (Cochrane and Laurance 2002; Robinne *et al.* 2016), until the level of landscape fragmentation no longer produce extensive fire events despite high ignition frequency (Syphard *et al.* 2007). Even if large fires decrease at a global scale (Andela *et al.* 2017), the multiplication of interface fires represents a substantial threat (Radeloff *et al.* 2005; Ganteaume *et al.* 2013). There is also a growing interest in the effects of small and low-to-medium-severity fires, which represent 35% of current burned area (Randerson *et al.* 2012), given that they are not harmless for the stability of socio-hydrosystems.

1.3.3 ENVISIONING THE FUTURE OF WILDFIRE RISKS TO WATER SECURITY

Bladon et al. (2014) propose a glimpse of what global changes could mean for the future of wildfire risks to water security. At a global scale, the concomitant expansion of humankind in natural ecosystems, the increasing number of disturbances in water basins, and the lower stability of the global water cycle may cause significant changes to the reliability of RFWR (Zhang *et al.* 2017). Extreme natural hazards will occur more often with higher levels of damages (Huppert and Sparks 2006; Myers and Patz 2009). Therefore, the role of fire in the hydrosphere is likely to increase, but in proportions that have not been studied yet. Potential future alternation of droughts and extreme rainfalls may increase the occurrence of post-fire floods (Moody and Ebel 2012).

The introduction of the wildfire-water risk concept, as a generalized framework to identify and study post-fire hydrogeomorphic threats to RFWR and socio-hydrosystems, enforces this new idea of the wildfire-society-water nexus (Martin 2016) and other efforts to depict wildfires as an emerging, and potentially a future primary source of risk to water security in the Anthropocene (Kinoshita *et al.* 2016). The growing number of studies involving fire as a threat to RFWR is also encouraging and lays the foundation for an increasing integration of wildfire activity as an emerging risk to water security. The field of socio-hydrology is up-and-coming and will probably play a major role in our further understanding of coupled human-water systems, acting as a catalyst for the exploration of new sources of water-related hazards, for the integration of new risk management approaches, and for the development of resilient socio-hydrosystems.

The loss of ecosystem services due to global environmental is a significant source of concern, and current capacity of our global ecosystems to provide water-related ecosystem services has been severely impaired (Dodds *et al.* 2013; Costanza *et al.* 2014). As a growing number of studies point at the degradation of water-related services after wildfires (Hurteau *et al.* 2014), protecting headwaters and river basins remains more than ever the base of the management of the future of wildfire risks to water security (Palmer *et al.* 2009; Capon and Bunn 2015). It is, therefore, necessary to capitalize on existing protected areas for the protection of freshwater provisioning (Green *et al.* 2015; Harrison *et al.* 2016), which also implies the reintroduction or the maintenance of a natural fire regime as a necessary element of ecosystem stability. The reflection on the preservation of water services provided by healthy water basins is already well started, which also include their protection from unnatural fire activity (Emelko *et al.* 2011; Workman and Poulos 2013; Abell *et al.* 2017). While waiting for the completion of comprehensive datasets on the diversity of socio-hydrosystems of the world, the protection of the global natural capital, with fire if necessary, remains an efficient way to maintain the flow and the quality of RFWR (McDonald and Shemie 2014).

1.4 RELEVANCE OF THIS RESEARCH

1.4.1 RESEARCH QUESTION

Water security is one of the main paradigms driving current global water governance, but it has been so far a challenge to achieve, and future conditions will likely not make it easier. In a changing world where system stationarity is a now a dead concept (Milly *et al.* 2008) and unknown environmental tipping points expose coupled human and natural systems to brutal changes (Lenton 2013), scientists and managers need to re-evaluate the current realm of global threats to water security. Phenomena that were not seen as harmful in the past now raise concerns over extensive areas, triggering social and physical disruptive dynamics for which existing responses may not be adapted. Exploring new avenues for the identification of emerging risks and their potentially disastrous consequences will offer new insights to better design global change adaptation strategies. Also, the limited focus that the WWR has received so far prevent researchers and managers from giving guidelines that widely applicable across the world, and make the reach of our knowledge very short-sighted. Therefore, my research question is the following:

"Where does fire activity stand in the global quest for water security in the context of global environmental change?"

I believe the present work is in line with the 2015-2030 Sendai Framework for Disaster Risk Reduction, implemented by the United Nations Office for Disaster Risk Reduction. The first goal of this framework is to identify emerging risks caused by global changes (UNISDR 2015). The interest for risks to water security from wildfire activity has been growing for less than 15 years, with the bulk of related publications —less than 30— published those past five years. However, the issue has never been exposed using this terminology and has been mostly limited to watershed-scale studies. There is therefore no official framework to handle this issue, despite the almost ubiquitous nature of wildfires. Only the recent concept of the wildfire-water-society nexus exposed by Deborah Martin, attempted to formalize a vision of wildfire hazards as a source of adverse effects on socio-hydrosystems functioning (Martin 2016).

A macro-scale study of this risk is missing as well. Based on the different assemblages of hydrologic resource availability, population pressure, and fire activity across the globe and the inherent uncertainty in their future patterns, it is likely that wildfire risks to water security will show different configurations. However, existing publications are geographically limited to a few countries able to undertake such specific research. I propose hereafter a global vision of this risk and a method to benchmark future evolutions according to expected global changes. This work, heavily conceptual, could also help to raise awareness, leading to more global research efforts and suggesting more attention on this issue from local water managers. This work is also in line with the pledge for multi-scale assessment of water security issues expressed in the Bonn Declaration on Global Water Security (Global Water System Partnership 2013).

Vogel (2015) calls for an update of the analysis of risks to coupled socio-hydrosystems. He argues that research, engineering, and management practices must evolve in the face of global change and take into account what could have been unimportant or unthinkable in the past. In this respect, I contend that the work presented here, supported by recent interest in fire effects on water supply, participate to the enlargement of this vision.

1.4.2 OBJECTIVES

The present work is the first-ever attempt to integrate wildfire activity as a significant disruptor of water security. To do so, it introduces the wildfire-water risk as a concept addressing the pressing needs that have been mounting for the past few years, after large severe wildfire events gave way to major water security concerns. Moreover, the fast-growing interest for water-related ecosystem services reliability and protection in the face of global changes justifies the creation of a new exploratory framework supporting RWFR sustainability. Therefore, this research is based on the three following core objectives:

1) To introduce the WWR concept and propose a robust analysis framework allowing for easy understanding, flexibility, benchmarking, and policy making;

2) To apply the framework to a spatial analysis and provide a global vision of the WWR, from the exposure of water resources to fire activity to the risk induced by upstream wildfires to downstream human and natural communities;

3) To integrate measures of global change to the framework and evaluate the future evolution of the WWR according to them.

1.4.3 METHODOLOGICAL OVERVIEW

To fulfill the objectives cited above, the present work principally relies on indexation modeling, an approach that has been extensively used for global-scale environmental risk analysis (Halpern *et al.* 2009; Vörösmarty *et al.* 2010b; Freudenberger *et al.* 2013; Padowski *et al.* 2015). In indexation modeling, the work consists in the gathering of datasets used as proxy indicators representing different aspects of the problem at stake. In a second phase, those indicators are transformed using different methods in order to make them comparable. Finally, all indicators are merged into a composite quantitative index and are tested for robustness through sensitivity analysis methods. Such approach can either provide spatial or non-spatial results. This is the modeling approach used for the three global-scale chapters.

The Alberta chapter is based on more classical statistical modeling approach, in which 31 years of area burned data are tested for abnormal patterns potentially explained by anthropogenic transformation of the landscape. The originality of this peace resides in the spatial interpretation of the changes that is based on the concept of frontier, a fairly old geographic concept but totally appropriate to describe the evolution of the Canadian landscape.

1.5 THESIS LAYOUT

The present work is articulated in four core chapters. Each one provides advancement in the exploratory concept of current and future wildfire risks to water security. The first chapter presents a global analysis of the exposure of water resources to wildfire activity. The second chapter builds on the previous one and presents a global analysis of the potential downstream effects of wildfires on different elements defining water security. The third chapter, at the scale of Alberta, doesn't directly involve water resource information but has been done as part of a larger project focusing on the potential for the disruption of the water supply due to forest fires. It also helps to introduce the notion of change in this research, specifically human-induced changes on landscape fire activity. The fourth chapter is a comparative study that includes future predictions of freshwater availability and fire activity and evaluates the degree of change in the wildfire-water risk with current conditions. A fifth chapter provides a synthesis of the aforementioned studies and answers the research question defended by this work, thereby making a case for the inclusion of wildfire activity and post-fire water-related hazards in future freshwater security assessments.

CHAPTER 2 - A GLOBAL INDEX FOR MAPPING THE EXPOSURE OF WATER RESOURCES TO WILDFIRE

2.1 ABSTRACT

Wildfires are keystone components of natural disturbance regimes that maintain ecosystem structure and functions, such as the hydrological cycle, in many parts of the world. Consequently, critical surface freshwater resources can be exposed to post-fire effects disrupting their quantity, quality and regularity. Although well studied at the local scale, the potential extent of these effects has not been examined at the global scale. We take the first step toward a global assessment of the wildfire water risk (WWR) by presenting a spatially explicit index of exposure. Several variables related to fire activity and water availability were identified and normalized for use as exposure indicators. Additive aggregation of those indicators was then carried out according to their individual weight. The resulting index shows the greatest exposure risk in the tropical wet and dry forests. Intermediate exposure is indicated in mountain ranges and dry shrublands, whereas the lowest index scores are mostly associated with high latitudes. We believe that such an approach can provide important insights for water security by guiding global freshwater resource preservation.

2.2 INTRODUCTION

Wildfires are essential to ecosystem function across the globe (Bond *et al.* 2005), influencing a wide spectrum of ecosystem components and natural processes (Lavorel *et al.* 2007), among which is the hydrological cycle. Accordingly, an abundant literature has described the effects of vegetation burning and post-fire recovery on local hydrology in different biogeographic areas (Kuczera 1987; DeBano 2000; USDA 2005; Seibert *et al.* 2010). Vegetation cover, litter and soil organic matter can be dramatically reduced by large fires and can lead to higher surface runoff and soil erosion, increasing water quantity, but decreasing water quality, although this general pattern may likely display regional variability. The water requirements of rapidly growing post-fire vegetation can subsequently limit water quantity (Kuczera 1987), even though water quality may improve (Dunnette *et al.* 2014).

Although a significant number of studies have examined such second-order fire effects on surface freshwater resources (Scott *et al.* 2012), most have been conducted at a local or regional scale (Moody and Martin 2004; Weidner and Todd 2011; Boerner *et al.* 2012; Scott *et al.* 2012; Thompson *et al.* 2013), whereas global-scale studies do not exist. Despite the ubiquitous nature of fire and the potential for adverse consequences on ecosystems and populations (Emelko *et al.* 2011), large-scale assessments of the risks that fire can pose to water resources are lacking. Such knowledge may prove to be useful to address current and emerging water security issues, especially in areas where water is already scarce and where fire is heavily used as a landscape transformation tool, as well as in areas where ongoing environmental change may both increase fire occurrence and pressures on water resources. However, several important advancements in natural resources global mapping (Shi *et al.* 2015) and the development of innovative methods and global databases now make it possible to better understand the intersection of wildfire activity (Bowman *et al.* 2009; Knorr *et al.* 2014) and water resource availability (Oki and Kanae 2006; Davies and Simonovic 2011) at the scale of the planet.

The large diversity of data types and derived metrics in these global databases creates a challenge for conducting global assessments, particularly when combining data from two fields, pyrogeography and hydrogeography. Often, resource or risk indices are created by aggregating proxy variables, called indicators, that are known to play a role in the occurrence of the studied phenomenon (Halpern *et al.* 2009; Vörösmarty *et al.* 2010a; Freudenberger *et al.* 2012; Dickson *et al.* 2014). In such an approach, raster datasets representing indicators are selected and normalized to assign each pixel a score. Each indicator is then assigned a weight according to its assumed importance to the phenomenon of interest. Numerous indicators can then be aggregated to create a final raster index, whereby pixel values reflect the degree of risk or resource availability. Finally, the index can be tested for its sensitivity to each indicator and assigned weight. Several key global studies used this approach to underline issues in water security and riverine biodiversity (Vörösmarty *et al.* 2010a), ocean vulnerability to human impact (Halpern *et al.* 2009) and to identify natural areas of great importance for ecosystem functioning (Freudenberger *et al.* 2012).

Inspired by this effective approach, we introduce here the concept of the wildfire water risk (WWR), which we define as the potential for wildfires to adversely affect water resources important for downstream ecosystems and human water needs for adequate water quantity and quality. We present a spatial framework as a foundation for assessing this underappreciated risk and introduce the global wildfire water exposure index (GWWEI) as a first step toward an integrated global assessment of the WWR. We then evaluate the sensitivity of the GWWEI to seven indicators relevant to fire and to water resources. Finally, we discuss how inclusion or variation in individual exposure indicators affects the interpretation of the index.

2.3 MATERIALS AND METHODS

We detail below the procedure of the GWWEI concept, starting with a precise description of the data selected to be used as indicators in our framework (Fig.2-1). We then explain how those data were transformed to obtain normalized indicators, resulting in pixel values ranging from 0–100. We follow with an explanation of the weighted aggregation process of indicators' scores, known as indexation, and finally, we perform a thorough sensitivity analysis of the resulting index to test its stability.

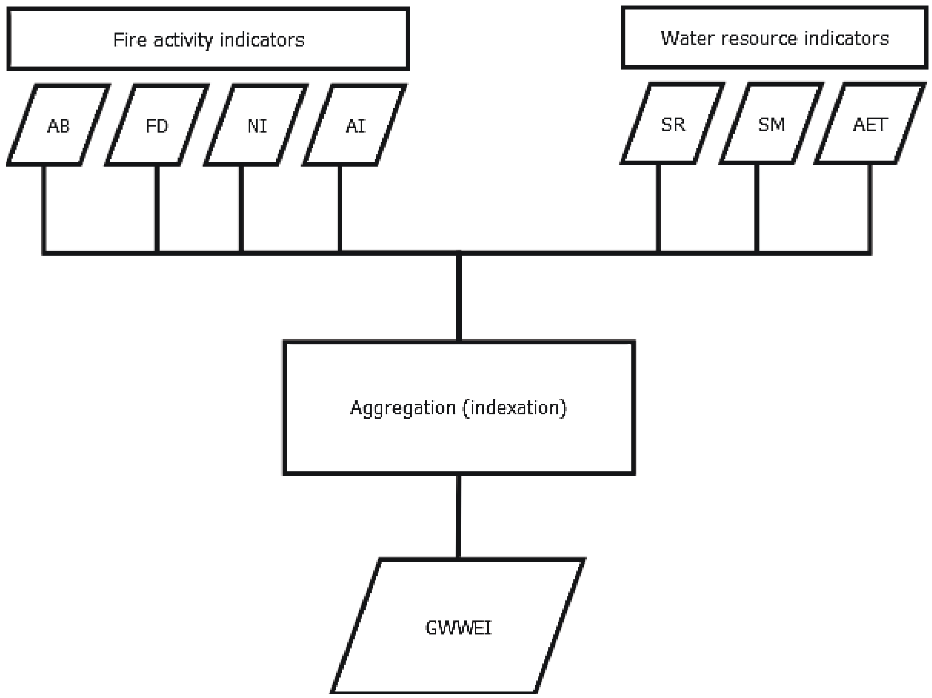

Figure 2-1: *Schematic of the global wildfire water exposure index (GWWEI) framework. AB, area burned; FD, fire danger; NI, natural ignitions; AI, anthropogenic influence; SR, surface runoff; SM, soil moisture; AET, actual evapotranspiration.*

2.3.1 DATA SELECTION AND INDICATORS' DEFINITION

We selected a parsimonious set of global indicators that described the potential for wildfire activity and the availability of surface freshwater resources (Tab.2-1). A total of seven indicators were selected based on their availability at the global scale, their relevance to the GWWEI and the nature of the information (i.e., yearly to multi-decadal averages). All data used in this study are "off-the-shelf" and freely available on the Internet or by request from the authors. Although our data have some discrepancies in their time period, they are the product of large-scale long-term monitoring, which substantially smooths spatial and temporal variability, making them suitable for use in a global model. Although slight temporal mismatches may be responsible for some inaccuracies, there is reason to believe that these would be relatively minor.

***Table 2-1**: Summary of datasets used to develop the GWWEI indicators. NASA: National Aeronautics and Space Administration, SEDAC: Socioeconomic Data and Applications Center, GWSP: Global Water System Project, CGIAR-CSI: Consortium of International Agricultural Research- Consortium for Spatial Information*

Indicator	Data Source	Units	Native Resolution	Coverage Years
Area Burned (AB)	Giglio et al.	Ha/month	0.25°	1997–2013
Fire Danger (FD)	NASA	unitless	0.5° × 2/3°	1980–2014
Natural Ignitions (NI)	NASA	Flashes/km2/year	0.5°	1995–2010
Anthropogenic Influence (AI)	SEDAC	Unitless (0–100)	0.08°	1960–2004
Surface Runoff (SR)	GWSP	mm/year	0.5°	1950–2000
Soil Moisture (SM)	Terrestrial water budget; data archive	mm/m	0.5°	1950–1999
Actual Evapotranspiration (AET)	CGIAR-CSI	mm/year	0.08°	1960–1990

Area burned (AB) has been found to be a good global proxy for fire activity (Krawchuk *et al.* 2009), especially as fire size is an important factor of post-fire impact to water resources (USDA 2005). Mean monthly area burned (hectares) for large fires (>120 ha) was extracted from the Global Fire Emission Database (GFED) V4, a database derived from remote-sensing imagery acquired with several sensors. Data span 1995–2014, and are spatially aggregated at a 0.25° pixel resolution (Giglio *et al.* 2009, 2013). Our AB indicator, as an average of the monthly area burned for the past 20 years, provides a view of areas experiencing most of the fire activity across the planet.

Fire danger (FD) is a measure of the potential for a fire to ignite and spread across the landscape and therefore is critical to assess water resources exposure. The most common fire danger metrics are calculated using the Canadian Fire Weather Index (FWI) System (Van Wagner 1987), which estimates existing fire danger across an area as derived from observations of four fire-weather elements (i.e., temperature, relative humidity, wind speed, and precipitation). An increasing index value means lower fuel moisture, higher wind speed and, consequently, a greater fire danger. Data come from the Global Fire Weather Database (GFWED), a global database of the FWI system and its components. Data are derived from the Modern Era-Retrospective Analysis for Research and Applications (MERRA) climate product provided by NASA and ground weather stations, compiled for 1980–2012 at a resolution of 0.5° latitude × 2/3° longitude (Chen *et al.* 2008; Field *et al.* 2015). Our FD indicator, based on the final FWI, provides information about the potential for fire activity, but does not account for actual area burned, vegetation composition or human influence on fire activity.

In many places of the world, lightning activity is an important factor of fire ignition (Krawchuk, Moritz, *et al.* 2009) that can lead to a large area burned when it occurs in remote areas (Ramos-Neto and Pivello 2000; Stocks *et al.* 2002; Bond and Keeley 2005). We used the mean annual lightning flash rate as an indicator of natural ignitions (NI), expressed as the number of flashes per km2 and per year. Data come from the High Resolution Flash Climatology, a sub-product of the Gridded Lightning Climatology dataset produced by the Lightning and Atmospheric Electricity Research Team at NASA using LIS/OTD remote-sensing observations. It is the result of flash counts per area scaled by the detection efficiency of sensors and gridded at a resolution of 0.5° for 1995–2010 (Cecil *et al.* 2014). We build our NI indicator considering that a higher lightning flash rate is associated with a higher chance for lightning to reach the ground, potentially starting a fire when the strike occurs in a vegetated area. As it does not account for individual strikes, lightning activity should not be considered as an actual fire ignition product.

The anthropogenic influence (AI) on fire activity is well known, but is still a matter of debate, as the nature of this influence is complicated (Aldersley *et al.* 2011; Archibald *et al.* 2013; Bistinas *et al.* 2013). Nonetheless, a recent study argues that human influence tends to decrease fire activity at the global scale (Knorr *et al.* 2014) and, consequently, the area burned. We thus consider higher levels of AI as an indicator of lower fire activity. As a proxy for AI, we used the Human Footprint Index (HFP) V2 data from the Socioeconomic Data and Applications Center from NASA, computed from 1995 to 2004 at a one-kilometer pixel resolution. This data depicts the extent and density of human features, both commonly associated with increased levels of anthropogenic disturbances in natural areas, with lower values showing a lower footprint, on a 0-100 score scale (Sanderson *et al.* 2002). However, scores are scaled per biome and, thus, encompass different socio-environmental configurations, which, in turn, have different effects on fire activity across the globe (Archibald *et al.* 2013).

Surface runoff (SR) is excess precipitation contributing to surface river-stream networks after evaporative and drainage losses. It can be greatly increased due to changes in water interception by vegetation and alteration of soil properties caused by wildfires. SR data is available as long-term average runoff, derived from a global water-balance model and river gauging stations, computed in mm/year at a 0.5° pixel resolution over the 1950–2000 period (Fekete *et al.* 2002; GWSP Digital Water Atlas 2008). For this study, our indicator assumes that areas showing higher levels of SR play a prominent role in the amount of available water resources and are thus more vulnerable to disturbances. We thus considered them as preferential areas of post-fire runoff increases; although one may argue that lower levels of SR would put limited water resources at a higher danger from fires and that regional variability exists and should be acknowledge. That said, if natural SR increases when vegetation cover is reduced, it becomes more difficult to predict and can lead to greater erosion levels and greater variations in base- and peak-flows, however in proportions that might be difficult to represent at a global scale.

Soil moisture (SM) reserves are critical to sustain surface runoff and dry season river-stream base-flows. Although high levels of SM favor runoff and water availability, it is also sensitive to post-fire changes in vegetation cover (Kasischke *et al.* 2007). SM data were compiled from the Atlas of the Biosphere (Willmott and Matsuura 2001) and based on the Terrestrial Water Budget Data Archive produced by the Center for Climatic Research at the University of Delaware (Legates and Willmott 1990a; b; Willmott and Matsuura 2001). Data were derived from several thousands of weather stations records from 1950 to 1999 and interpolated at a 0.5° pixel resolution using the Thornthwaite climatic water budget algorithm. Our indicator assumes that a drop in soil moisture content after a fire is caused by greater inputs of radiative energy (Moody *et al.* 2015), which, in turn, could negatively impact low flows during the dry season, although here again this proxy may vary substantially depending on the scale of analysis, underlying environmental conditions, and post-fire vegetation dynamics (Andréassian 2004).

The reduction of the vegetation cover after a fire might impact actual evapotranspiration (AET) levels (Nolan *et al.* 2014), which is the effective quantity of water released by vegetation transpiration and water evaporation from the soil. AET data come from the Global High-Resolution Soil-Water Balance dataset produced by the Consortium of International Agricultural Research Centers-Consortium for Spatial Information (CGIAR-CSI) (Zomer *et al.* 2006). It is based on a water budget modeling approach essentially based on the combination of potential evapotranspiration (Hargreaves method), maximum soil water content, and reference vegetation water demand. It shows the average of AET in mm/year at a 0.08° pixel resolution, from 1950–2000, based on WorldClim inputs. Our indicator is used as a proxy for post-fire water-balance change, based on the reasonable assumption that without vegetation interception and respiration, AET will mainly be converted to runoff. This process would be limited, however, by expected increases in post-fire soil-water evaporation.

2.3.2 DATA PROCESSING AND AGGREGATION

All data were rasterized, reprojected to the WGS84 geographic coordinate system and resampled to a 0.5° pixel resolution. We used the FWI layer, which does not account for desert areas, as an extraction mask for other layers. Therefore, we avoided result biases by including arid areas where climatic conditions restrain water availability, as well as vegetation growth and, consequently, wildfire activity. We also processed the grids in order to match the spatial coverage of FD. Finally, small islands without consistent coverage through the different layers were removed, as well as Greenland and Antarctica (28% of global land surface). Data were processed with ArcGIS 10.1 (Environmental Systems Research Institute 2012) and exported as GeoTIFF images for post-processing.

Prior to the indexation process, data were normalized between 0 and 100 scores and then considered as actual indicators of the GWWEI (Fig.2-2). Normalization, in this context, makes indicators comparable to each other by replacing initial values (e.g., mm or ha)

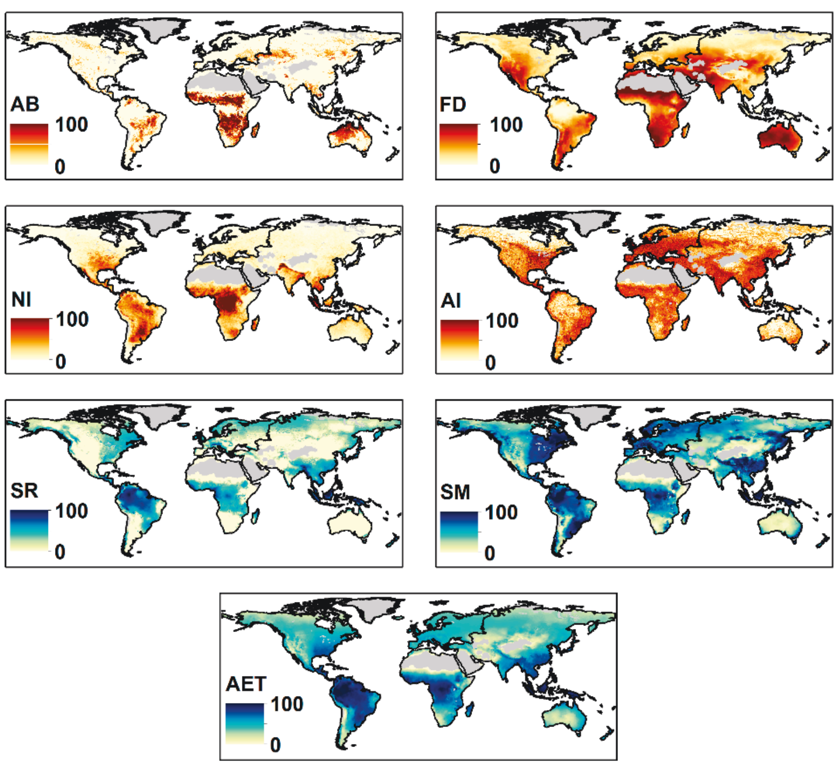

Figure 2-2: *Map series of selected spatial indicators. Wildfire indicators are shown in orange tones; water indicators are shown in blue tones.*

according to a common and standard scale, here 0–100. Our raw exposure index is then a simple pixel-wise additive aggregation process of the selected indicators, based on their respective attributed weight:

$$I = \sum_{i=1}^{n} w_i x_{n,i}$$

where I is our final risk index; n is the number of indicators (i.e., 7); w_i is the relative weight of each indicator; $x_{n,i}$ is the normalized value of each indicator (Biber *et al.* 2011). As this work is a first exploration, we wanted to create a baseline result in which no factor is considered as dominant, so our weighting scheme assigns 50% to fire indicators and 50% to water indicators and equally partitions the weights within each of these groups. Therefore, we assigned a 16.6% weight to each water indicator (3, total 49.8%) and a 12.5% weight to each fire indicator (4, total 50.2%), so the overall weighting scheme equals 100%. As a result, one pixel's final score in the index theoretically ranges from 0 to 100, a higher score meaning a higher concentration of exposure factors. This method is inspired by the work of Freudenberger et al. (Freudenberger *et al.* 2012). Data normalization and index calculation were carried out using Insensa-GIS (0.2.0.1), 64-bit version (Biber *et al.* 2011).

2.4 SENSITIVITY ANALYSIS

It is critical in indexation models to test the robustness of the aggregated index to evaluate the level of confidence in the final score (Oecd 2008). Specifically applied to our index, it will help to better understand the factors controlling the final score and to update the number and weight of indicators included in order to produce a better version of this index so the results are more reliable. We thoroughly evaluated the sensitivity of the raw index to the seven indicators using one non-spatial approach and six spatial approaches (Tab.2-2). The main product of this analysis is a measure of score variability expressed as a coefficient of variation that was computed from the re-weighting of the indicators and by omitting some indicators from the calculation in order to assess their relative weight to the final score.

Table 2-2: *Details pertaining to each sensitivity analysis method.*

Sensitivity Analysis Method	**Procedure Detail**	**Weight Variation Scheme**	**# of Modified Indices**
Spearman/Pearson correlation	Calculus of correlation coefficients between index and indicators	-	-
Stepwise	One-by-one addition of each indicator until final index	-	-
Jackknifing	Iterative exclusion of each indicator in the aggregation process	-	7
Low/high case scenario	Bounded weight variation based on indicator distribution	Within 6.5% and 18.5% for fire indicators; Within 10.6% and 22.6% for fire indicators	2
Random variation	Bounded random weight variation	Within 6.5% and 18.5% for fire indicators; Within 10.6% and 22.6% for fire indicators	14
Systematic variation	Incremental bounded weight variation	Within 6.5% and 18.5% for fire indicators; Within 10.6% and 22.6% for fire indicators	28

The first common technique we applied was to non-spatially analyze index sensitivity by measuring the level of correlation between the GWWEI and each indicator separately, as well as among indicators (Tab.2-3). The Spearman correlation coefficient table was generated as a measure of dependency, such that indicators highly correlated with the final index have an overall higher influence on final index scores.

The simplest spatial approach we used for our sensitivity analysis was the “stepwise” method. We reprocessed GWWEI adding one indicator at a time. We started with the weighted aggregation of only AB and SR, as the former is the recorded fire activity and the latter is the recorded natural water availability; together, these indicators logically provide the simplest possible index. Then, we added the other indicators individually, alternating fire and water indicators until all were included (i.e., the GWWEI itself). This simple stepwise approach to sensitivity analysis allowed us to monitor the spatial changes caused by the addition of each new variable included and to assess variation in the spatial distribution of risk scores.

Insensa-GIS (Biber *et al.* 2011) also implements several modes allowing for a thorough spatial sensitivity analysis. We used jackknifing; low-high case scenario weighting; random weighting; and systematic weighting of indicators. These four methods captured the variability in indicator aggregation, giving information about their intrinsic influence when compared to the original index results (Tab.2-2). For all weight variation modes, we computed a pixel-wise mean and coefficient of variation and averaged them into one final map of the index's overall coefficient of variation.

The jackknifing mode involves the iterative exclusion of each indicator from the aggregation procedure. As this process removes our seven indicators successively to create a new index each time, jackknifing produced eight modified indices; in other words, one for each missing indicator.

Lower and higher case scenarios modify the weight of indicators according to a predefined range of variation, which is based on their influence on the aggregation result. As such, if an indicator favors high index scores, its weight will be depreciated, yet not below the predefined minimum. The opposite is true for a higher case scenario, whereby an indicator lowering the final index scores will be over-weighted, below or equal to the upper bound of the range of variation. We set the lower case weight boundary to 6.5% and the higher case weight boundary to 18.5% for fire indicators and 10.6%–22.6% for water indicators, a range we consider wide enough to capture index variability. This process produced two modified indices, one for each scenario.

Random weight variation involves the randomization of each indicator's weight during the aggregation procedure, according to a predefined variation range. We set the same variation range as for the previous mode, which means that an indicator can randomly be assigned any weight in this range during indexation. We applied this procedure several times to increase the detection of variations in index scores, which resulted in 14 new modified indices.

Finally, we created a rule set to apply the systematic weighting variation mode. We kept the same range of variation that we used for previous modes with a 3% step increment. The process is repeated for each indicator, resulting in 28 new modified indices. In total, the sensitivity analysis created 51 modified versions of the index (not shown), with the coefficient of variation computed for each of the four modes. We averaged those to produce a map of the per-pixel mean variability of the GWWEI scores, where areas showing higher variability are thus more sensitive to changes in the indicators' values.

2.5 RESULTS AND DISCUSSION

2.5.1 GEOGRAPHY OF THE GWWEI

Our GWWEI (Fig.2-3 a) shows the distribution of the exposure of water resources to wildfires across the globe. Highest scores are concentrated in the tropical latitudes, more specifically in the forests of the Amazon basin, the Congo basin and Indonesia. Moderately high scores are mostly located in the subtropical humid forests of southeastern Asia, southeastern North America, Central America and in fire-prone dry forested savannas of Africa, southeastern South America and southeastern Oceania. A large part of northeastern North America, as well as many mountain ranges across the globe also shows moderately high scores. Intermediate scores are shown in dry savannas, dry steppes and dry shrublands on all continents, as well as in the Mediterranean, the northwest of the Eurasian boreal forest and the southern range of the North American boreal forest. The lowest index scores are seen in the temperate prairies of North America, South America and Eurasia, as well as in the northern boreal and the tundra (Fig.2-3 b).

At this stage of our framework development, it is important to recall that the GWWEI does not describe a quantitative likelihood or probability of impacts on water resources. It rather depicts the geographic overlay of important drivers of the WWR and identifies areas where such quantitative assessments must be carried out. Working at the global scale usually smooths regional differences, and in this regard, the scores should be interpreted according to specific environmental, socio-cultural and economic factors. For instance, high scores in African savannas are mostly driven by the AB, as those ecosystems experience most of fire activity on Earth (Giglio *et al.* 2013), whereas high scores in mountain ranges are mostly driven by intermediate to high scores of SM. It is important to note that indicators are global-scale proxies that may not be suitable when estimating fire risk or water discharge across small areas.

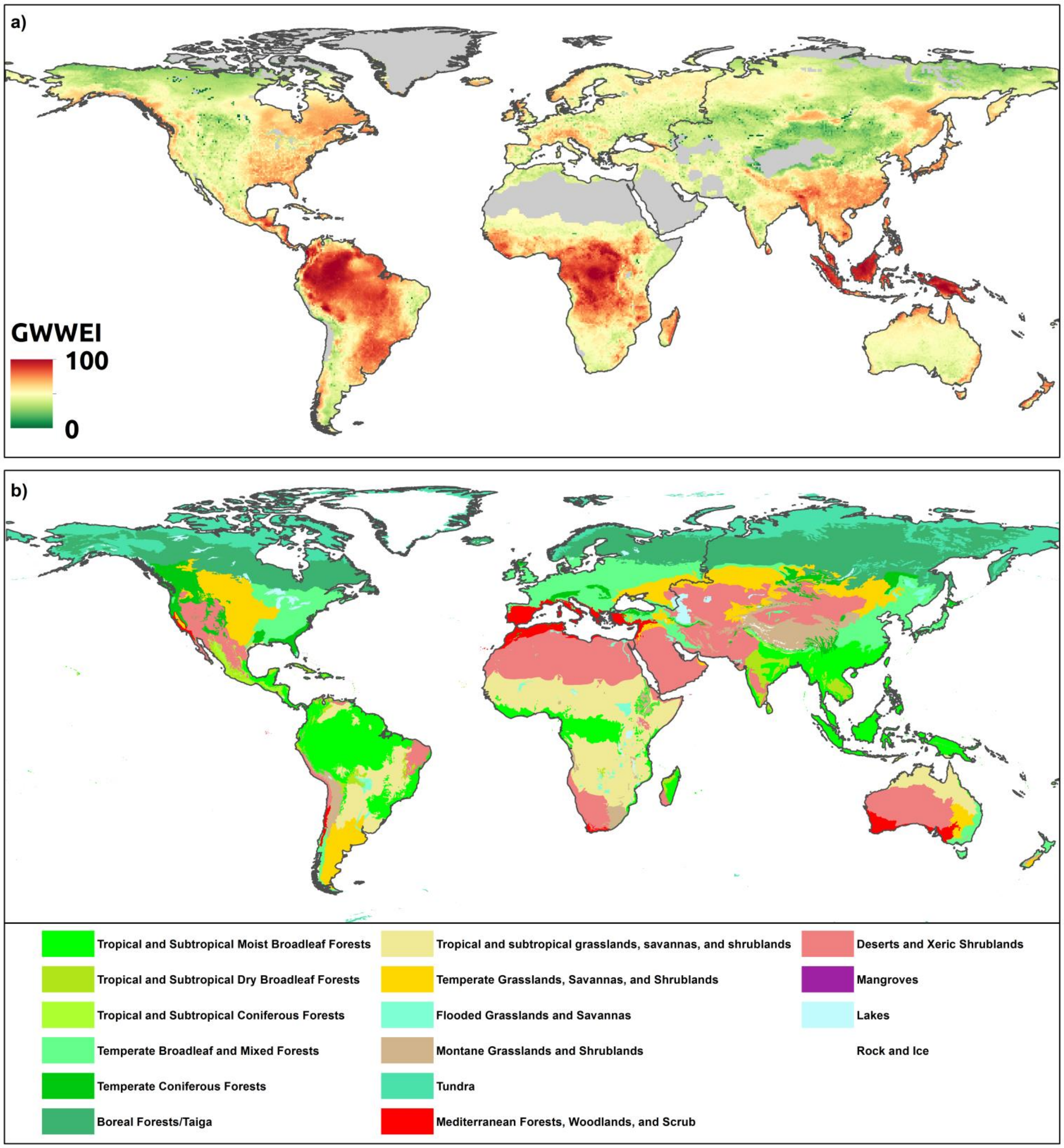

Figure 2-3: *(a) Map of the global GWWEI as provided by additive aggregation. The index is dimensionless; scores stretched to 0–100. Higher values (100, dark red) mean a higher concentration of risk factors. (b) A map of terrestrial biomes (Olson et al. 2001) also provided for comparison purposes (see Section 3.1).*

2.5.2 SENSITIVITY OF THE GWWEI

The Spearman correlation coefficients (Tab.2-3) between the GWWEI and indicators show that the most influential water indicators are AET (0.76), SM (0.74) and SR (0.66), whereas the most influential fire indicators are NI (0.55) and AB (0.21). The correlation between the index and water indicators explains the pattern of high values in tropical areas, which naturally concentrate a very dynamic hydrological cycle. This influence of water resource indicators is confirmed by the stepwise approach, where the inclusion of AET in the simplest version of the index (Fig.2-4 a) sets a pattern that is conserved and enforced through all steps (Fig.2-4 b–f), with SM being critical in setting the pattern for mountain ranges, such

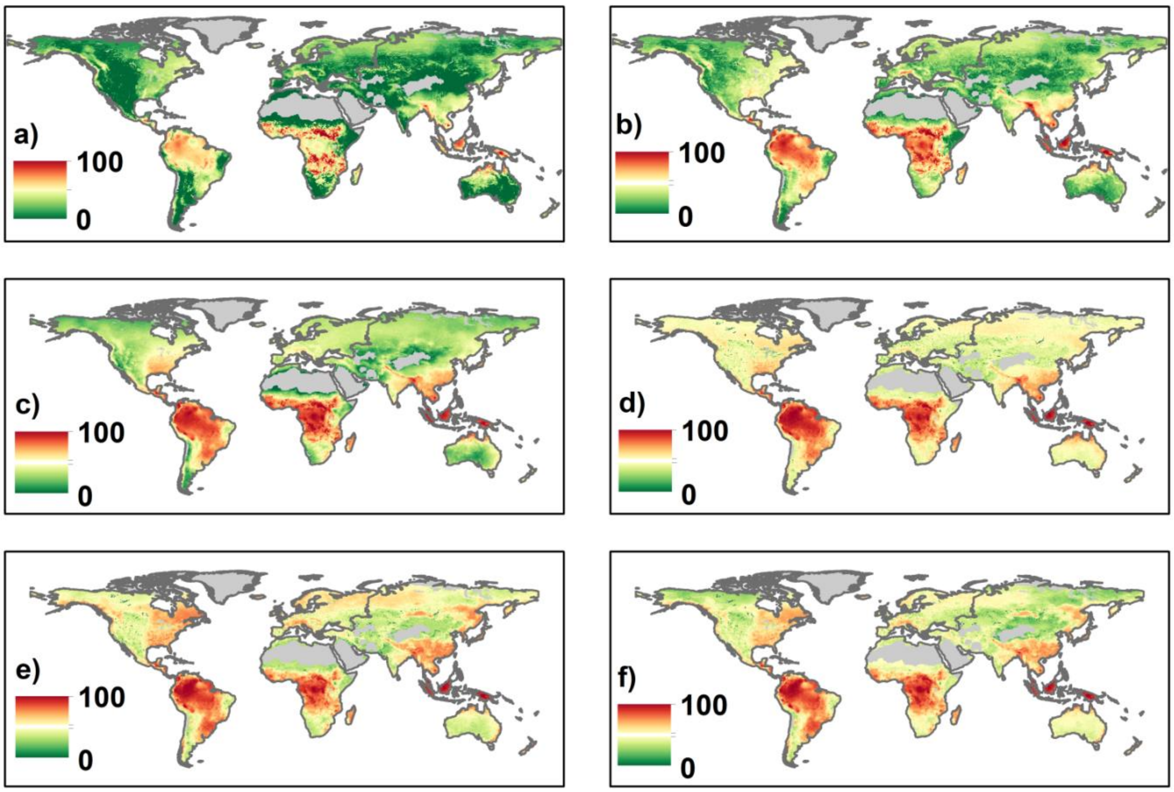

Figure 2-4: *Map of the stepwise sensitivity analysis as provided by additive aggregation. (a) AB + SR; (b) AB + SR + NI; (c) AB + SR + NI + AET; (d) AB + SR + NI + AET + AI; (e) AB + SR + NI + AET + AI + SM; (f) AB + AET + NI + SM + AI + SR + FD, i.e., the GWWEI (see Figure 1). The index is unitless; scores stretched to 0–100. Higher values (100, dark red) mean a higher concentration of risk factors.*

as the Himalayas or Southern Alaska, as well as increasing scores for the southern fringe of the boreal forest (Fig.2-4 c). NI and AB are the fire indicators that add the most to the pattern of the final index, whereas FD and AI show surprisingly low influence. We assume that the strong pattern shown by water indicators may mask information contained in fire indicators, thus showing lower levels of correlation in them.

Table 2-3: *Spearman correlation coefficients between the GWWEI and source indicators, as well as between indicators.*

		GWWEI	Fire				Water		
			AB	**FD**	**NI**	**AI**	**AET**	**SM**	**SR**
GWWEI		**1.00**	0.21	−0.11	0.55	−0.06	0.76	0.74	0.66
Fire	**AB**	0.21	**1.00**	0.36	0.35	−0.06	0.20	−0.13	−0.04
	FD	−0.11	0.36	**1.00**	0.41	−0.31	−0.14	−0.58	−0.57
	NI	0.55	0.35	0.41	**1.00**	−0.41	0.65	0.15	0.15
	AI	−0.06	−0.06	−0.31	−0.41	**1.00**	−0.40	−0.16	−0.02
Water	**AET**	0.76	0.20	−0.14	0.65	−0.40	**1.00**	0.68	0.63
	SM	0.74	−0.13	−0.58	0.15	−0.16	0.68	**1.00**	0.76
	SR	0.66	−0.04	−0.57	0.15	−0.02	0.63	0.76	**1.00**

Although several nonlinear relationships and interactions might exist, they are not explored with these simple correlation coefficients. The highest values of the coefficient of variation (Fig.2-5) (i.e., where the index is less robust) are mostly concentrated at northern latitudes (i.e., the tundra and northern fringe of the circumboreal forest), where water indicators have the most influence on the wildfire water risk exposure pattern (Fig.2-6). Moderately high to high variability in index scores is also shown in areas of dense human

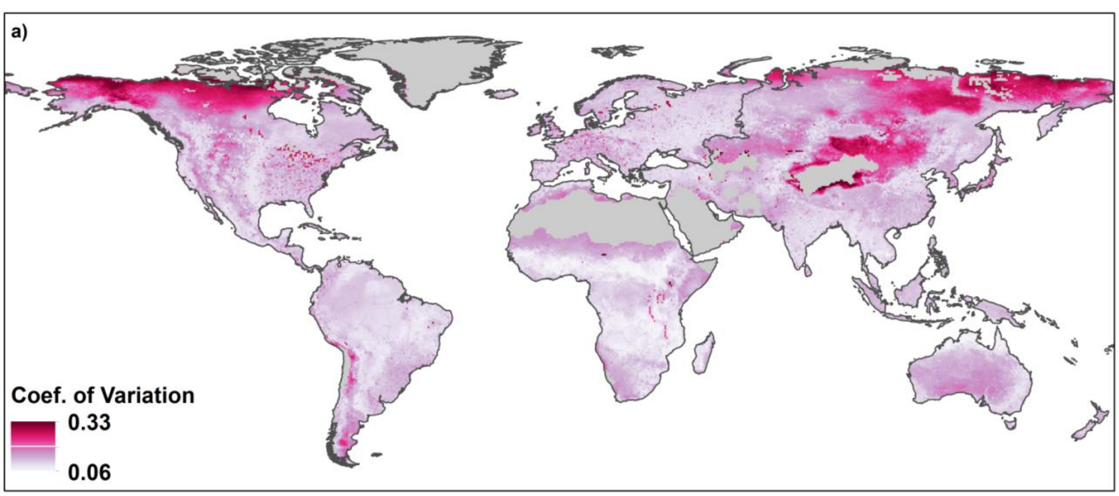

Figure 2-5: *Map of the average coefficient (Coef.) of variation derived from modified indices. Higher values (dark purple) show higher sensitivity to the weighting scheme used.*

pressure, like Japan, Western Europe and eastern North America. That said, this pattern is clearly localized, giving clusters of spotted areas on the map. Several mountain ranges, such the Andes, the Rocky Mountains, the European and New-Zealand Alps, also show this range of moderately high values.

Moderately low levels of variability cover most of circumboreal, temperate, tropical, and sub-tropical forests and dry shrublands in both hemispheres. Robust estimates of the GWWEI, i.e., the lowest coefficient of variation values, are concentrated in the tropical

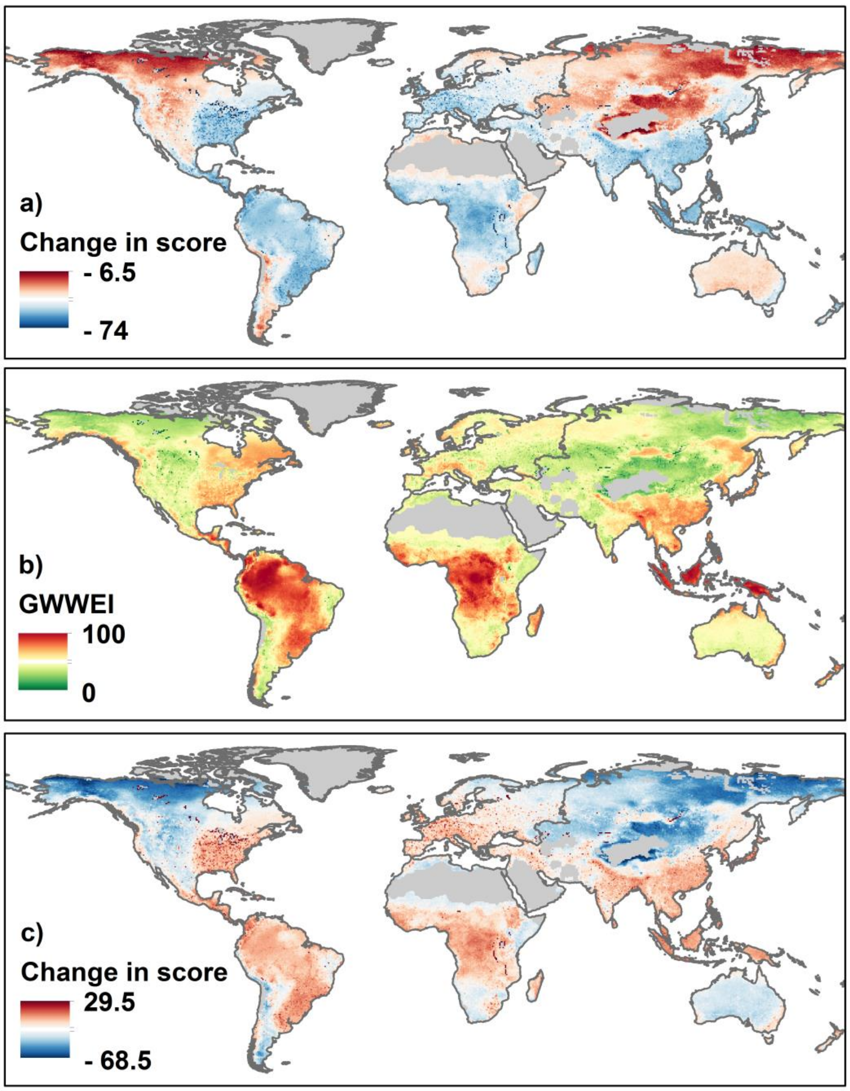

Figure 2-6: *Map set of the relative change (%) in final GWWEI scores using the low case (a) and high case (c) scenario mode in the sensitivity analyses. The role of water resource indicators is made clear by the pattern of change when compared with the GWWEI (b).*

savannas and the dry temperate steppes, except for the North American prairies, which show a wide range of variability, and for northern Australia, which shows a constant high level in all individual indicator scores.

2.5.3 THE GWWEI AND ITS IMPLICATIONS FOR WATER RESOURCE PROTECTION

Water security was originally defined as the guarantee of a safe, affordable and sustainable amount of water to fulfill one's basic daily needs (Falkenmark 2001). This definition has been subsequently extended to include the amount of water resources necessary to secure ecological functions, as well as agricultural or industrial activities (Norman *et al.* 2010). Knowing the potential exposure of water resources to wildfire activity, as provided by this study, as well as the current pressure on the water supply worldwide (Postel *et al.* 1996; Falkenmark *et al.* 2009; Vörösmarty *et al.* 2010a), we argue that a high level of GWWEI can have potential implications for water security. This is especially true in areas that are dependent on surface water coming from a highly fire-prone river basin. Regional studies from Thompson et al. (Thompson *et al.* 2013), Santos et al. (Santos *et al.* 2015) and Moody and Martin (Moody and Martin 2004), as well as reports from the U.S. Forest Service (Weidner and Todd 2011) and the Water Research Foundation (Ho Sham *et al.* 2013) showed that wildfire risk in source watersheds raises concerns for water treatment and supply. The global information provided by our index might be a good way to identify regions across the planet showing higher levels of exposure, potentially requiring more detailed wildfire water risk analysis for regional water planning and management. Besides, it can support further research efforts in supposedly exposed areas where post-fire hydro-geomorphic effects are not well known, if at all. Such finer-scale effort could, in return, provide meaningful information to test and improve the robustness of the index.

Surprisingly, wildfires are rarely considered as a critical threat to water resources by international authorities. This lack of recognition, despite major worldwide concerns about water sustainability and scarcity, is underlined by the absence of dedicated mentions in most of the global reports and mapping initiatives focusing on water security, water management issues or forested water basin monitoring; in contrast, the role of forests for water resource preservation is widely acknowledged. The GWWEI, as a part of a larger WWR framework, can contribute to knowledge improvement, especially in mountainous areas, known as "water towers", across the globe. Viviroli et al. (Viviroli *et al.* 2007) indeed showed that several mountainous regions provide at least a "supportive" amount of water for downstream supply needs, and Nogués-Bravo et al. (Nogués-Bravo *et al.* 2007) pointed out the extreme sensitivity of headwaters to natural hazards in the context of climate change, though wildfire was not considered. Mori and Johnson (Mori and Johnson 2013), for instance, demonstrated that mountains might experience significant changes in their fire regime because of climate change, whereas Moody and Martin (Moody and Martin 2004) showed the critical exposure of mountainous reservoirs to wildfire impacts, although limited to the western U.S. In this respect, our framework can be used to identify and prioritize sensitive areas and initiate the creation or improvement of resource management plans or mitigation actions.

A recent study by Green et al. (Green *et al.* 2015) shows global population dependence on upstream freshwater sources. According to our index, water resources' exposure to fire activity potentially threatens the water supply of a large portion of the human population, as underlined by several localized events. For example, the 2013 Rim Fire raised concerns with California State authorities when it threatened the Hetch Hetchy reservoir, which provides most of the water supplied to the San Francisco Bay Area, i.e., 2.6 million people. This recent event brought to light the threat induced by large and severe wildfires to communities dependent on surface freshwater to ensure daily potable water needs, as is the case with 78% of large cities on the planet (McDonald *et al.* 2014). Other major blazes that occurred in the past decade had significant impacts on several cities' water supply, such as Melbourne in Australia and Denver, Boulder and Santa Fe in the United States (Bladon *et al.* 2014), as well as numerous large cities across the world, such as San Salvador, Caracas and Istanbul, all of which are considered exposed to potential water provision issues in case of a major fire in their watershed (Dudley and Stolton 2003).

In cases where fire activity would increase the net quantity of water downstream, potentially severe impacts on water quality and timing/magnitude of flows (e.g. base flow or peak flow) can impact a wide range of ecological and human water resource uses (i.e., drinking water). This aspect will be further explored in future versions of the index, in which we will also try to integrate other sources of disturbances on water resources triggered by forest degradation and human activities to which watersheds are exposed. It could help us to decipher the role of wildfires from other environmental pressures and thus better advocate for their management as one of the many sources of watershed cumulative effects (Millar and Stephenson 2015).

2.5.4 LIMITATIONS AND IMPROVEMENTS

The sensitivity analysis showed that water resource indicators tend to overwhelm fire-related indicators in the pattern of the GWWEI. This raises a question about the assumption of equal weight used in the aggregation process. While we considered this assumption acceptable to create our framework, the variability in the spatial pattern of the index shows that different weighting schemes could improve its robustness (i.e., lower score variability). The following versions will integrate an intermediate step based on a survey of scientists, in order to obtain a robust rating of the score we should assign to each indicator. This step has been previously used in several studies describing the creation of risk and resource indices (Halpern *et al.* 2009; Vörösmarty *et al.* 2010a; Green *et al.* 2015). Another sensitivity aspect, more conceptual, pertains to the limited inferential conclusions that can be drawn from this work. It is important to underline that such index is based on an association of potentially influencing factors, not on statistical predictions or process-based modeling. Any interpretation should thus be done with high care.

Our initial pool of data is a collection of variables used to approximate a general – though variable in time and space– set of known factors controlling wildfire activity and freshwater availability. Our indicator list is intentionally simple, though we expect to extend it to explore the effect of alternative variables in future versions of the index. For instance, area burned could be replaced by adding different variables that contribute to fire probability, such as ecosystems' net primary productivity or drought proneness (Moritz *et al.* 2012). Similarly, the Build-Up Index of the FWI System may be a better proxy to fire impacts than FWI, because it better reflects burn severity, a critical determinant of post-fire hydrological effects. The correlation in water resource indicators must also be addressed by the inclusion of innovative information on surface freshwater availability, such as lake density or stream network connectivity. Moreover, resulting estimates of GWWEI could be improved if indicator data were averaged for biome-specific fire seasons, rather than annually. It is important to underline that we were dependent on data availability; the improvement of our index will therefore depend on the creation of and enhancements to global datasets, especially regarding water-related indicators, given that several wildfire indicators already exist.

Our current version of the framework only considers overlapping additive effects mostly based on long-term indicator averages. Further versions should also address possible indicators interactions (i.e., multiplicative effect), as well as address downstream cumulative effects in space and time, and explicitly consider existing connectivity in water systems that could potentially lead to adverse effects on the water supply (Alcamo *et al.* 2007). Extending the WWR framework to take into account the induced risk to the downstream water supply implies the integration of a "spatial transmission" process, in other words the capacity of a hazardous process to impact geographically-distant values at risk (Miller and Ager 2013). This process has been translated in the "downstream routing" method recently used in several studies related to the impact of human activities on water security at the global and continental scale (Vörösmarty *et al.* 2005, 2010a; Green *et al.* 2015) and may be considered in future versions of the GWWEI.

2.6 CONCLUSIONS

A unique global view of the potential exposure of water resources to wildfire activity and a valuable approach complementary to recent worldwide assessments of global exposure towards natural hazards was presented herein (Lerner-Lam 2007; Peduzzi *et al.* 2009). The highest exposure scores were mostly clustered in tropical wet forests, whereas intermediate scores tended to be localized in tropical dry forests and shrublands, as well as in several mountain ranges and boreal forests. The lowest levels were found in the tundra, temperate forests and temperate prairies. These results represent an important source of information that can be considered in the international governance of forested areas and freshwater resources.

Notably, the sensitivity analysis showed an overwhelming influence of water resource indicators on the final index scores, which indicates the need for several modifications in the weighting scheme, such as incorporating expert opinion or including a larger set of variables. Future improvements to the WWR framework should also explore restricting indicators' score range to worldwide fire seasons and develop new complementary indices that allow for the assessment of downstream water supply vulnerability and the subsequent risk to dependent populations and ecosystems.

The global index presented in this study can help us pinpoint regions of potential concern that may require a more detailed assessment of wildfire-induced risk to water resources. Indeed, high exposure levels may reveal the potential for deleterious impacts on water quality and downstream cumulative effects that might in turn affect local to regional water security, especially in river basins serving large populations. Although wildfires can impair water provision services from ecosystems, they are a natural and essential ecosystem process. Therefore, a trade-off has to be found between the preservation of natural fire regimes and the need for risk mitigation and source water protection. In this regard, the definition of a WWR opens new perspectives in the understanding of the global water and land systems. For instance, knowledge of this risk may help to enlarge the scope of post-fire hydrology to address potential large-scale post-fire hydrogeomorphic dynamics that may impede the achievement of the Sustainable Development Goals in some parts of the word. This framework adds an important component to the global water security paradigm in the context of climate change that does not presently encompass global wildfire water risk.

2.7 ACKNOWLEDGMENTS

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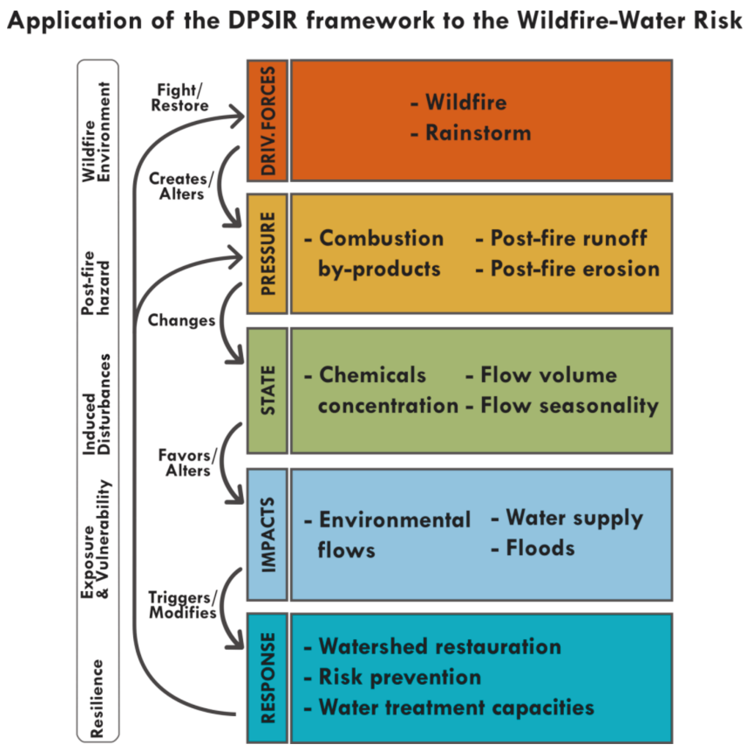
CHAPTER 3 - A GLOBAL ASSESSMENT OF WILDFIRE RISKS TO HUMAN AND ENVIRONMENTAL WATER SECURITY

3.1 ABSTRACT

A global increase of wildland fire activity in recent years has revealed the vulnerability of freshwater resources. The extensive hydrogeomorphic effects from a wildfire can impair the ability of watersheds to provide safe drinking water to downstream communities and high-quality water to maintain riverine ecosystem health. Safeguarding water use for human activities and ecosystems is required for sustainable development; however, no global assessment of wildfire impacts on water supply is currently available. Here, we provide the first global evaluation of wildfire risks to water security, in the form of a spatially explicit index. We adapted the Driving forces-Pressure-State-Impact-Response risk analysis framework to select a comprehensive set of indicators of fire activity and water availability, which we then aggregated to a single index of wildfire-water risk using a simple additive weighted model. Our results show that water security in many regions of the world may be vulnerable, regardless of socio-economic status. However, in developing countries, a critical component of the risk is the lack of socio-economic capacities to respond to disasters. Our work highlights the importance of addressing wildfire-induced risks in the development of water security policies; the geographic differences in the components of the overall risk could help adapting those policies to different regional contexts.

3.2 GRAPHICAL ABSTRACT



3.3 HIGHLIGHTS

- Severe wildfires may endanger the water supply of human and natural communities
- We created a global index to assess wildfire risks to water security
- We used the DPSIR framework to select and aggregate 34 risk indicators into one index
- Beyond post-fire hazards, potential impacts and resilience capacities drive the global risk
- Wildfire risk to water security can occur globally but may be particularly acute in water-insecure countries

3.4 INTRODUCTION

Ensuring water security, which is defined as the assurance of sufficient and safe freshwater resources for human development and ecosystem functioning, has long been a challenge in developing countries (United Nations 2005), and is a growing issue in more developed countries as population pressures increase consumption and pollution (Norman *et al.* 2012). Despite measurable improvements within the past decades, water insecurity still threatens or affects many countries. For instance, more than 2-billion people do not have access to an improved source of water (United Nations, 2016; Gain et al., 2016). Many of the critical issues are due to water pollution, diversion, or depletion (Meybeck 2003; Schwarzenbach *et al.* 2010; Hoekstra and Mekonnen 2012). Complex relationships among social stability, ecosystem health, and freshwater availability have been recognised, all of which condition water security (Rockström 2009; Dodds *et al.* 2013; Padowski *et al.* 2015). These relationships may be modified or enhanced by the occurrence of extreme natural disturbances (Grigg 2003; Huppert and Sparks 2006), thereby increasing the challenge of maintaining or achieving water security (Srinivasan *et al.* 2012b; Hall and Borgomeo 2013).

Recent catastrophic wildfires in the USA (e.g. California and Colorado), Canada, and Chile have drawn attention to the nexus among fire, water, and societies (Martin 2016). These natural disasters have increased interest in the wide range of consequences a severe and large wildfire can have on the reliability of surface freshwater resources (Emelko *et al.* 2011; Kinoshita *et al.* 2016). The hydrogeomorphic effects of wildfires can be numerous, spatially extensive, and long-lasting. These effects can potentially include increased annual water yields and peak flows, shifts in the timing of runoff due to earlier snowmelt, and decreased water quality due to high sediment and nutrient loads (Shakesby and Doerr 2006). Bladon et al. (2008) and Emelko et al. (2015) noted significantly higher concentration of trace elements, phosphorus and organic carbon in the water downstream of severely burned sites, persisting after several years. In the USA, Hallema et al. (2016)

attributed a +219% increase in annual water yield in a watershed of Arizona, while Moody and Martin (2001b) documented a 200-fold increase in erosion rates in two watersheds of Colorado. Conedera et al. (2003) estimated that a post-fire debris flow that happened in a small mountain catchment of Switzerland was the result of a 200-year flood triggered by a 10-year precipitation event; it represented four times less precipitation than required in an unburned basin for a similar flood type, and with much higher flow velocity. Post-fire hydrogeomorphic hazards may consequently expose water resources to drastic quality and quantity changes that can impair downstream water supply of human and natural communities.

These post-fire impacts on the downstream water supply could potentially result in substantial economic costs (Emelko *et al.* 2011; Emelko and Sham 2014) and could possibly threatens human and ecosystem health, although the occurrence of many of the threats identified in the literature have so far never been reported and may even be highly hypothetical when it comes to drinking-water safety (Finlay *et al.* 2012; Writer and Murphy 2012). Potentially greater erosion rates in burned watershed can increase sedimentation in reservoirs regulating drinking-water provision (Moody and Martin 2004; Smith *et al.* 2011), therefore reducing their storage capacity and their lifespan. The possible increase in the concentration of dissolved organic carbon, often documented, may theoretically pose problem for water treatability as its combination with chlorine, a fundamental water disinfection chemical, may favor the formation of carcinogenic disinfection by-products (Writer *et al.* 2014). Other hazardous chemicals, such as lead or arsenic, could also accumulate downstream and reach concentrations levels that can be far greater than what is prescribed for drinking-water quality by the World Health Organization, although concentration values are usually taken before treatment (Tecle and Neary 2015). The trophic chain of riverine and lacustrine ecosystems can be highly disturbed by changes in turbidity and chemical element concentration (Tobergte and Curtis 2016) leading to decrease in ecosystem health with consequences on fisheries and recreational use of water (Tecle and Neary 2015). The water security of downstream human and natural communities may, therefore, be threatened, making them vulnerable to risk from wildfire (hereafter 'wildfire-water risk' [WWR]) (Thompson *et al.* 2013; Bladon *et al.* 2014; Robinne *et al.* 2016). As an emerging risk to coupled human-water systems, the WWR has been gaining in interest for the past decade. However, the threat it represents to global water security remains to be understood in a context of planetary change (Bogardi *et al.* 2012), in which extreme weather events, such as droughts and floods (Mann *et al.* 2017) are predicted to become increasingly common.

Global composite indexes are commonly used in water security assessment (Vörösmarty *et al.* 2010b; Garrick and Hall 2014), risk analysis (Dilley *et al.* 2005; Peduzzi *et al.* 2009; De Bono and Mora 2014), and other diverse environmental questions (Halpern *et al.* 2009; Freudenberger *et al.* 2012). Composite indexes are efficient tools to explore complex processes and to convey high-value information in an easily understandable manner (Gregory *et al.* 2013). They also help detect temporal and spatial trends in the evolution of a process, making them valuable to monitor policies effectiveness (OECD 2008). However, a robust composite index requires a well-structured analytical framework. The Driving forces-Pressure-State-Impact-Responses (DPSIR) framework (EEA 1999) simplifies complex causal relations between human and natural systems at several spatial scales (Bitterman et al., 2016; Freudenberger et al., 2010). It has been successfully applied to questions related to risk evaluation, water resources management, biodiversity protection, and economics (Halpern *et al.* 2009; Freudenberger *et al.* 2012). Meybeck (2003) contends the DPSIR framework as an appropriate tool for the analysis of global issues impacting freshwater quality and availability. The novelty of the WWR and its inherent complexity make it a good candidate for a DPSIR analysis. This framework is considered a problem structuring method that can help organising the numerous natural and social processes involved in the characterization of the risk and thus provide a tool to develop targeted policies (Gregory *et al.* 2013).

The present study adapts the DPSIR framework to produce the first global-scale assessment of the wildfire risks to water security. Our objective is threefold: 1) develop a reference WWR spatial analysis framework at a global scale, 2) understand the current geography of the WWR according to the different criteria involved, and 3) raise awareness of WWR issue to global water security challenges. To do so, we demonstrate the benefit of the DPSIR risk-based framework to creating a spatially explicit index. This index is then used to produce a global map showing the geography of the risk. We finally discuss the importance of our approach to the understanding of wildfire risks to water security and the questions posed by future global changes.

3.5 DATA AND METHODS

3.5.1 DATA

For clarity, we present hereafter the 33 datasets we used according to the five Drivers-Pressure-State-State-Impact categories, and we briefly explain their use as indicators (Tab.3-1). Although our application of the DPSIR framework deviates from that from the original EEA (1999), our adaptation of this approach remains similar to numerous other studies (Maxim *et al.* 2009). As no specific data depository representing the diversity of post-fire issues is currently available, we used a large panel of datasets available free of charge and able to approximate this diversity according to the literature.

3.5.1.1 Driving forces

The driving forces are those elements that trigger a chain of cascading events leading to the appearance of an environmental problem. For the WWR, post-fire effects are triggered by the combination of large and intense wildfire activity, high biomass load, extreme precipitation or snowmelt, and a steep terrain. We included seven variables in the analysis of the Driving Forces category.

We used the monthly average of the Build-Up Index (BUI) data (1998-2014 TRMM-3B42 version) from the Global Fire Weather Database (Field *et al.* 2015) as a proxy for fire severity. The BUI, based on the Canadian Fire Weather Index (FWI) System, is related to the amount of fuel available for combustion. The FWI System is a weather-based system and does not explicitly include vegetation type, structure and associated biomass loading in the calculation. Fire severity essentially affects soil chemical and physical properties—critical determinants of hillslope runoff generation (Neary *et al.* 2009) —by reducing the amount of above- and below-ground organic matter and exposes the soil to the erosive forces of rain, making excessive high-velocity runoff more likely to happen (Doerr *et al.* 2000). To approximate fire frequency, we used an aggregated sum of the yearly NASA MODIS fire counts for the period 2000-2010 (Giglio 2007), with higher values having a stronger negative effect. In addition, we integrated data on soil macrofauna diversity (Orgiazzi *et al.* 2015) (e.g. earthworms, arthropods, ants, and moles). According to recent studies, soil engineering capacities of those animals may limit fire occurrence and impacts (Henig-Sever *et al.* 2001; Hayward *et al.* 2016) and act as a buffer to post-fire runoff (Cerdà and Doerr 2010).

Fire activity across the world is highly correlated –either positively or negatively– with human pressure on landscapes (Bistinas *et al.* 2013). We used the human appropriation of net primary productivity (HANPP) (Imhoff and Bounoua 2006) as a proxy to human ignition capacity, considering that higher levels of HANPP would foster higher ignition frequency and density. The data, provided by the Socioeconomic Data and Application Center, is the ratio of available NPP to the human demand of NPP per capita, according to local water consumption patterns. Fire ignitions also naturally occur through lightning activity. Lightning fires have an important ecological role (Ramos-Neto and Pivello 2000) and they usually display different spatial and seasonal patterns compared to human-caused fires (Vazquez and Moreno 1998; Müller *et al.* 2013). They can also account for the largest burned areas in flammable forested ecosystems (Gralewicz *et al.* 2012). We used lightning flash density data on natural lands derived from the LIS and OTD sensors provided by the NASA Global Hydrology Resource Center (Cecil *et al.* 2014), although this data does not represent the density of ground strikes *per se* but the density of flashes.

Post-fire precipitation intensity is a paramount factor driving the occurrence and magnitude of post-fire hydrogeomorphic effects (Moody and Martin 2001a). Heavy precipitation events following large and severe wildfires can trigger destructive flash floods with unusually high streamflow and debris loads leading to catastrophic effects on downstream infrastructures (Jordan 2015) and water quality (Neary and Tecle 2015). To represent the occurrence of extreme precipitation, we used the maximum one-day precipitation amount from the CLIMDEX NCEP2 1979-2011 daily reanalysis data (Sillmann *et al.* 2013) created by the Expert Team on Climate Change Detection and Indices (Zhang *et al.* 2011). Although extreme precipitation is a strong driver of erosion is general, the literature also relates higher erosion rates and thus hydrogeomorphic effects with increasing slope steepness, although with variable responses depending on several environmental factors such as soil type or precipitation regime. From a post-fire perspective, Miller *et al.* (2011) pointed at the role of steeper slopes in post-fire sediment entrainment. To account for this effect in our framework, we used a global physiographic landform layer derived from the SRTM mission that relates complex terrains with mean slope gradients (Drăguţ and Eisank 2012).

3.5.1.2 Pressure

Indicators of the Pressure category of the DPSIR framework are proxies to the first order effects of wildfires on hydrosystems. In other words, these indicators inform on the direct hydrogeomorphic changes caused by the driving forces of fire severity and area burned that can eventually lead to downstream impacts. We included seven indicators in the estimate of the Pressure category.

The potential for an increase in the frequency and intensity of post-fire erosion-runoff events with potential impacts on water quality and quantity was approximated with soil variables from various global datasets (Batjes *et al.* 2009; Shangguan *et al.* 2014). Topsoil bulk density gives an idea of pre-fire soil structure and wettability (Neary *et al.* 2009), whereas topsoil sand content has been reported many times to be associated with post-fire hydrophobicity (DeBano 2000). Data on sediment deposit thickness (Pelletier *et al.* 2016) was added to approximate the potential for accumulation and remobilization of sediment-associated pollutants, based on adsorption processes making sediments a medium for chemicals to accumulate downstream (Ballantine *et al.* 2009; Murphy *et al.* 2015). We also retrieved soil moisture holding capacities from the IGBP soil database (Global Soil Data Task 2014) as a proxy for the potential for saturation-excess runoff. To account for the often-observed reduction in post-fire soil infiltration (USDA 2005; Certini 2005), we considered the hydrologic effect more deleterious in areas of higher pre-fire moisture, based on the fact that wet soils are better heat conductors, although this may be considered with caution as the hydrological response of soils to a burn is highly variable (Badía *et al.* 2017). We also included the erodibility factor K from a global RUSLE model (Naipal *et al.* 2015) to get a sense of preferential erosion areas.

As the deposition of smoke on surface freshwaters is believed to impact water quality (Spencer *et al.* 2003), we integrated this aspect using global smoke deposition estimates of 2.5 μm particulate matter derived from satellite observations of wildfire smoke emissions and global air mass modelling (Johnston *et al.* 2015). The mean annual runoff was retrieved from the Global Water System Project digital water atlas (GWSP, 2008) and was used to account for areas where post-fire overland flow is likely to enhance existing runoff values.

3.5.1.3 State

State indicators represent changes in quality and quantity of a phenomenon as a function of biotic and abiotic pressures. Applied to the WWR, those indicators approximate induced post-fire hazards as the various effects of post-fire hydrogeomorphic changes on nutrient concentration, flood occurrence, earlier peak flows, coarse woody debris flows, and ecosystems water retention capacities. We used eight indicators to represent the State category.

Frequent and severe wildfires can seriously deteriorate the soil microbiota (Prendergast-Miller *et al.* 2017), thereby increasing nutrients availability for leaching and soil instability after wildfire, as well as limiting vegetation recovery. We used a global representation of soil fungal taxonomic richness (Tedersoo *et al.* 2014) to account for this potential change in soil biotic capacities and the biodiversity it may impact. Higher levels of organic matter, represented by vegetation, litter, and humus in the soil may potentially favor the availability of labile chemical compounds transported by post-fire leaching and runoff, potentially leading to water treatment challenges, although it would be highly dependent on burning conditions (e.g., fire intensity or depth of burning) and post-fire environmental factors. We retrieved aboveground biomass data from the Geocarbon project (Avitabile *et al.* 2014) and we used several global soil datasets (Table 1) to account for belowground organic carbon, soil nitrogen, and soil phosphorus concentration derived from different soil properties databases based on field observations and geographic upscaling for global-scale mapping. As most of the fire effects on soils are detectable in the first centimeters of the soil profile (González-Pérez *et al.* 2004), we only used topsoil information (0–30cm) when available.

Wildfires, as disturbance agents of the hydrological cycle, may potentially favor the occurrence of floods triggered by storms or snowmelt (Rulli and Rosso 2007; Seibert *et al.* 2010). To account for those potential changes, we used snow-water equivalent data from the Global Land Data Assimilation System (GLDAS) (Rodell *et al.* 2004) and the flooded area fraction derived from a global 100-year river floods vulnerability model (Tanoue *et al.* 2016). Potential post-fire flash floods may also favor coarse woody debris recruitment, with a surge of material directly after the fire. Woody debris flows are themselves influenced by forest age, as older and disturbed forests tend to produce more debris (Sturtevant *et al.* 2011). We accounted for this hazard using global forest age data (Poulter et al., In Prep.), although to date no study has specifically addressed a potential forest age - coarse debris relationship.

3.5.1.4 Impacts

Indicators addressing impacts translate the effects of pressures and consequent changes on highly valued resource critical for the functioning of human and natural components of the system. In the WWR context, upstream hydrogeomorphic pressures cause changes to water quality and quantity, thereby threatening water supply capacity to downstream human communities and ecosystems. We used six indicators to represent these impacts.

As wildfires affect water flows and chemical balance, freshwater ecosystems might be exposed to disruptions of their environmental flows (Dahm *et al.* 2015). We used the global estimation of environmental water requirements developed by Smakhtin et al. (2004) based on a modeling approach providing information on multi-year available water and variability of runoff, from which low and high flow percentiles are determined for a given basin and its ecological water needs. Closely associated with environmental flows, freshwater biodiversity can be particularly sensitive to post-fire changes in water characteristics (Bixby *et al.* 2015), and we thus integrated a global estimation of freshwater biodiversity (Collen *et al.* 2014).

Impacts of wildfires to the water supply of human communities is a growing concern (Bladon *et al.* 2008; Hohner *et al.* 2017). We integrated several indicators approximating these effects on freshwater resources. We calculated domestic water withdrawal based on the ratio of water consumption per capita (FAO 2016) to average population density derived from 2000-2013 LANDSCAN data (UT-Battelle LLC 2013). As open water bodies can represent an important source of usable water for communities (Bakaic and Medeiros 2016), we also computed the global density of lakes using the Global Lakes and Wetlands Database levels 1 and 2 (Lehner and Döll 2004). Post-fire sediment exports can lead to higher sedimentation rates in reservoirs, a concerning effect that can reduce reservoir life expectancy (Moody and Martin 2004). This impact was represented by the use of potential sediment trapping data by large dams, based on an empirical model linking reservoir size, mean annual discharge, and reservoir sediment trapping efficiency (Vörösmarty *et al.* 2010b). We finally included the relative water stress index, as the ratio of total human water consumption to renewable water resources (Vörösmarty 2000). This last indicator informed on areas experiencing chronic water supply disruption that might be aggravated by cumulative impacts to freshwater resources from burned areas.

3.5.1.5 Response

The Response category illustrates crisis management options, as well as tools and methods for risk management that are available to societies and their capacities to deploy them. Applied to the WWR, improvements in fire prevention, firefighting techniques, post-fire watershed restoration, and post-fire risk mitigation seem to be adequate responses. The Response, therefore, defines the level of resiliency of a socio-hydrosystem to the wildfire-water risk. The absence of global data on wildfire management expenditures precludes the creation of the indicators identified in the DPSIR diagram. We used diverse datasets, five in total, to approximate this capacity.

We used Gross Domestic Product per capita (The World Bank 2016) as the best way to represent risk management capacity (Lerner-Lam 2007). We also used Investment Benefit Factor data (Vörösmarty *et al.* 2010b) as a proxy to the likelihood of society to maintain access to water following post-fire hazard occurrence. The number of hospital beds was used as a proxy for healthcare access (Horev *et al.* 2004) and the number of pupils as a proxy for risk education capacities (Izadkhah and Hosseini 2005). We considered those both indicators as critical aspects of social resilience to disaster. Data were retrieved from the Global Assessment Report on Disaster Risk Reduction (De Bono and Mora 2014). Finally, we incorporated travel time data (Nelson 2008) to account for the importance of the transportation network in response to a disaster, which here can be the capacity to fight a fire or site accessibility for restoration.

***Table 3-1**: List of the variables used to compute the WWR index. (I) specifies indicators whose values were inverted.*

Name	DPSIR	Unit	Temporal Coverage	Spatial resolution	Source	Proxy
Monthly mean Build-Up Index	D	Unitless	1990-2010	0.5° x 2/3°	Global Fire Weather Database (GWFED)	Potential for greater depth of burn and vegetation combustion
Fire counts	D	thermal anomalies/ yr	2001-2010	0.5°x0.5°	NASA Global Monthly Fire Product (MCD14ML)	Potential for fire susceptibility and soil impoverishment
Soil macrofauna diversity	D	# groups	2015	0.008°x0.008°	European Commission JRC	Potential for higher soil moisture and lower ground fuels, thus limiting fire ignition and spread (I)
Human appropriation of net primary productivity	D	g C/m2/yr	1995	0.25°x0.25°	NASA SEDAC	Potential for human ignition
Lightning flash density	D	Flashes/km2/yr	1995-2000	0.5°x 0.5°	NASA GHRC	Potential for natural ignition
Max 1-day precipitation	D	mm	1979-2011	2°x2. 5°	CLIMDEX NCEP2 Reanalysis	Potential for heavy rainstorm
Global topography	D	Unitless	2007-2012	0.008 °x0.008°	SCALA project	Potential for dangerous fire behaviour, flash flooding, and debris flow
Topsoil bulk density	P	Kg/dm3	2000	0.05°x0.05°	NASA HWSD	Potential for reduced post-fire infiltration
Topsoil sand content	P	% weight	2000	0.05°x0.05°	NASA HWSD	Potential for hydrophobicity
Sediment deposit thickness	P	meters	1900-2015	0.008°x0.008°	University of Arizona, USA	Potential for changes in post-fire turbidity and solid transport
Erodibility Factor K	P	t ha h/ha/MJ/mm	1995-2009	0.008°x0.008°	GTOPO-ETOPO-Max Planck Institute for Meteorology, Germany	Potential for postfire erosion susceptibility
Soil moisture holding capacity	P	mm	1995	0.08°x0.08°	IGBP_DIS	Potential for changes in soil water storage
Smoke deposition PM 2.5	P	µg/m3	1997-2006	2°x2.5°	University of Tasmania, Australia	Potential for water pollution from smoke deposition
Annual mean runoff	P	mm/yr	1950-2000	0.5°x0.5°	GWSP	Potential for post-fire effects accumulation (I)
Soil fungal diversity	S	# of taxons	1960-1990	0.33°x0.33°	University of Tartu, Estonia	Potential for changes in soil stability and vegetation regrowth
Above ground biomass (carbon)	S	Mg/ha	2000-2011	0.01°x0.01°	GeoCarbon project	Potential for the production of labile combustion by-products
Topsoil organic carbon content	S	% weight	2000	0.05°x0.05°	NASA HWSD	Potential for the production of labile combustion by-products
Soil phosphorus concentration	S	% weight	Multiple	0.08°x0.08°	GSDE	Potential for the production of labile combustion by-products

Soil nitrogen concentration	S	g/m²	1995	0.08°x0.08°	IGBP_DIS	Potential for the production of labile combustion by-products
Yearly mean snow-water equivalent	S	mm	2000-2010	0.25°x0.25°	NASA GLDAS	Potential for changes in flow seasonality
Flooded area fraction (100 years return interval)	S	% per area	1960-2013	0.25°x0.25°	University of Tokyo, Japan	Potential for catastrophic floods
Forest age	S	Age of dominant PFT	Multiple	0.5°x0.5°	Montana State University (Unpublished work), USA	Potential for changes in large woody debris production
Environmental water requirements	I	% total discharge	1961-1990	0.3°x0.3°	IWMI	Potential for water supply contamination
Freshwater biodiversity	I	Species richness	1994-2012	(vector)	Zoological Society of London, England	Potential for adverse effects of freshwater ecosystems
Domestic water withdrawal	I	m3/hab/yr	1900-2010	0.008°x0.008°	Aquastat - LANDSCAN	Potential for water supply contamination
Lake density	I	# lakes/km2 (weighted by size)	1992-1998	(vector)	WWF - GWLD	Potential for water supply contamination (I)
Sediment trapping by large dams	I	% land to ocean flux	2003	0.5°x0.5°	GWSP	Potential for the reduction of dams' lifetime
Water stress index	I	km3	1995-2002	0.5°x0.5°	WWRII-UNH	Potential for water supply contamination
Gross Domestic Product per cap.	R	Current US$	2016	-	World Bank	Potential for risk management
Investment benefit factor	R	Unitless	2010	0.3°x0.3°	Riverthreat.net	Potential for resilience
Healthcare access	R	# of hospital beds	2012	-	World Development Indicators	Potential access to health care
Risk education capacity	R	# of pupils	2012	-	UNESCO	Potential for prevention
Travel time	R	# hours	2008	0.008°x0.008°	European Commission JRC	Potential accessibility for intervention and restoration

3.5.2 INDICATOR DEVELOPMENT

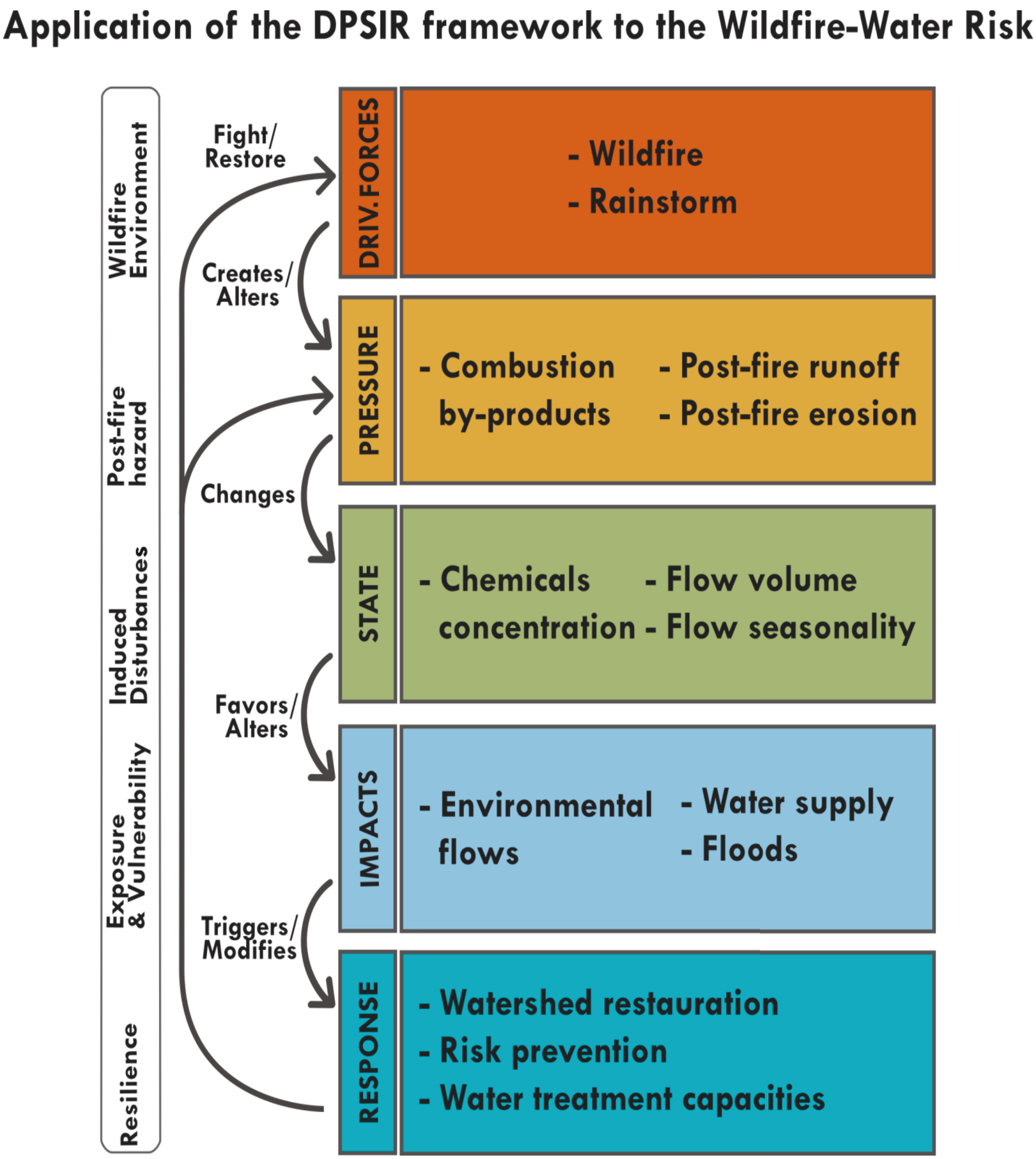


Figure 3-1: *Simplified version of the Driving Forces-Pressure-State-Impact-Response framework applied to the wildfire-water risk analysis (see Appendix 3). Each DPSIR category was paired with aspects of risk management: the wildfire environment as driving forces, potential post-fire hazards as pressures, induced post-fire disturbances triggered by post-fire hazards as states, exposure and vulnerability to those induced disturbances as impacts, and resilience capacities as the response. We applied an identical color scheme to symbolize each DPSIR category in the following figures. This is an idealized WWR framework and data to fully represent it in the index were often not available.*

The DPSIR framework offers both a widely validated environmental risk analysis method and a flexible design as to the potential range of indicators included and thus the complexity of the studied process (Tscherning *et al.* 2012). Based on cause-and-effects relationships, this framework provides a logical tool to identify the different variables involved in the evaluation of the post-fire hydrological risk, as well as a method to sort them into categories interconnected by environmental dynamics (Niemeijer and de Groot 2008; Maxim *et al.* 2009). Therefore, fitting our WWR approach to the DPSIR framework was the first critical step (Fig. 3-1). The DPSIR logic is driven by the significance of indicators, usually based on experts' opinion, in explaining the system under study (Bitterman *et al.* 2016). It means, and it is a very important point, that it is common that several indicators identified while adapting the framework may not be integrated in the final index if data to represent them are unavailable and that no adequate proxy can be found.

We used datasets that were publicly available or easily obtained and converted them all to the same raster format. Data were processed in ArcGIS 10.1(Environmental Systems Research Institute 2012) to produce a set of 34 indicators at 0.25 × 0.25 degrees spatial resolution in the WGS84 coordinate system. We did not keep small islands, Greenland, Antarctica, and areas with a runoff less than 10mm/year in the analysis thus applied on a final pool of 19,235 pixels. We adjusted (i.e., multiplied) the values of each raster data according to their probability of experiencing a fire (Moritz *et al.* 2012), for $p > 0.2$. This way, we emphasised the information contained in areas of higher probability as a potential source of post-fire hydrogeomorphic hazards or a preferential sink of exposure to post-fire hydrogeomorphic hazards. We then inverted the values of three variables –soil macrofauna, global runoff, and lake density– with a linear transformation so that the highest values correspond to the lowest values and thus account for the inverse relationship between these indicators and the inferred level of risk.

To simulate the propagation and the accumulation of those hazards downstream of drainage basins, we applied a routing function to all indicators in the D, P, and S categories. These indicators represented material that can be mobilized after a fire (e.g., runoff, debris, sediments, or nutrients) as well as processes and phenomena that mobilise this material (e.g., snowmelt, landslide). Based on a topological drainage network, upstream pixel values were added iteratively to downstream pixel values along a flow path (i.e., a network of contiguous pixels) from the source to the basin outlet, thereby mimicking downstream accumulation over the whole area. We applied the *accuflux* function available in PCRaster-Python Extension (van Rossum 1995; Karssenberg *et al.* 2010) onto the Dominant River Tracing network (Wu *et al.* 2012) for macroscale hydrological modelling. I and R categories were not routed.

Layers resulting from downstream-routing were then normalized (i.e. divided) by the global hydrologic discharge, which simply is the result of downstream-routing applied to the runoff (Vörösmarty *et al.* 2010b). This step acknowledges the adage 'the solution to pollution is dilution', which implies that the adverse effect of pollutant concentration is countered by higher volume of water available for dilution. We transformed routed and non-routed indicators using a base-10 logarithm function to obtain a standardized scale of values across all categories and to better account for contributing areas in the final index. We grouped these indicators according to their DPSIR category (Table 3-1), and we applied a principal component analysis (PCA) to each category, thereby collapsing the information spread across many indicators (see Appendix B). We only retained the first component of each category to create five global indicators, one for each category. We finally normalized the Response global indicator by the GDP per capita provided by the World Bank (The World Bank 2016), which penalized the locations with lower GDP values during the aggregation process.

We used Insensa-GIS (Biber *et al.* 2011), a software designed for the creation and verification of spatially-explicit composite indices, to create our WWR index. We first standardized our global indicators on a 0–100 scale using the following formula following notation standards prescribed by the OECD (2008):

$$I_q = \left(\frac{x_q - x_{q\,min}}{x_{q\,max} - x_{q\,min}}\right).100$$

Where I_q is a standardized global indicator and x_q the pixel values of the global indicator standardization is applied to. Then, we computed a composite indicator CI following a linear (i.e. additive) weighted aggregation method:

$$CI = \sum(I_{q,c}.w_{q,c})$$

With $I_{q,c}$ being the global indicator for each DPSIR category and $w_{q,c}$ the weight attributed to the global indicator. As proposed by Bitterman et al. (2016), the weights of each global indicator were assigned as a function of the number of in-out connections between categories (Tab.3-2), as displayed in the DPSIR flowchart (Fig.S1); categories with a higher number of connections were *de facto* attributed a higher weight. Finally, we performed a sensitivity analysis of the final WWR index, although limited for this study to jackknifing and low–high case scenarios (but see Freudenberger et al., 2013, 2012; Robinne et al., 2016), presented in Supplementary Material 3.

Table 3-2*: Number and direction of linkages among DPSIR categories and their respective final weights.*

Category	**Links in**	**Links out**	**Total Links**	**Weight**
Drivers	4	15	19	0.16
Pressure	15	14	29	0.25
State	12	12	24	0.20
Impact	12	14	26	0.22
Response	14	6	20	0.17

We produced a global map of the resulting composite index (Fig.3-2). After a general evaluation of the global risk pattern, an analysis was carried out using the global hydrobelt dataset created by Meybeck et al. (2013) to get a better sense of the WWR's regional patterns. Hydrobelts are defined as "global-scale delineations of the continental landmass

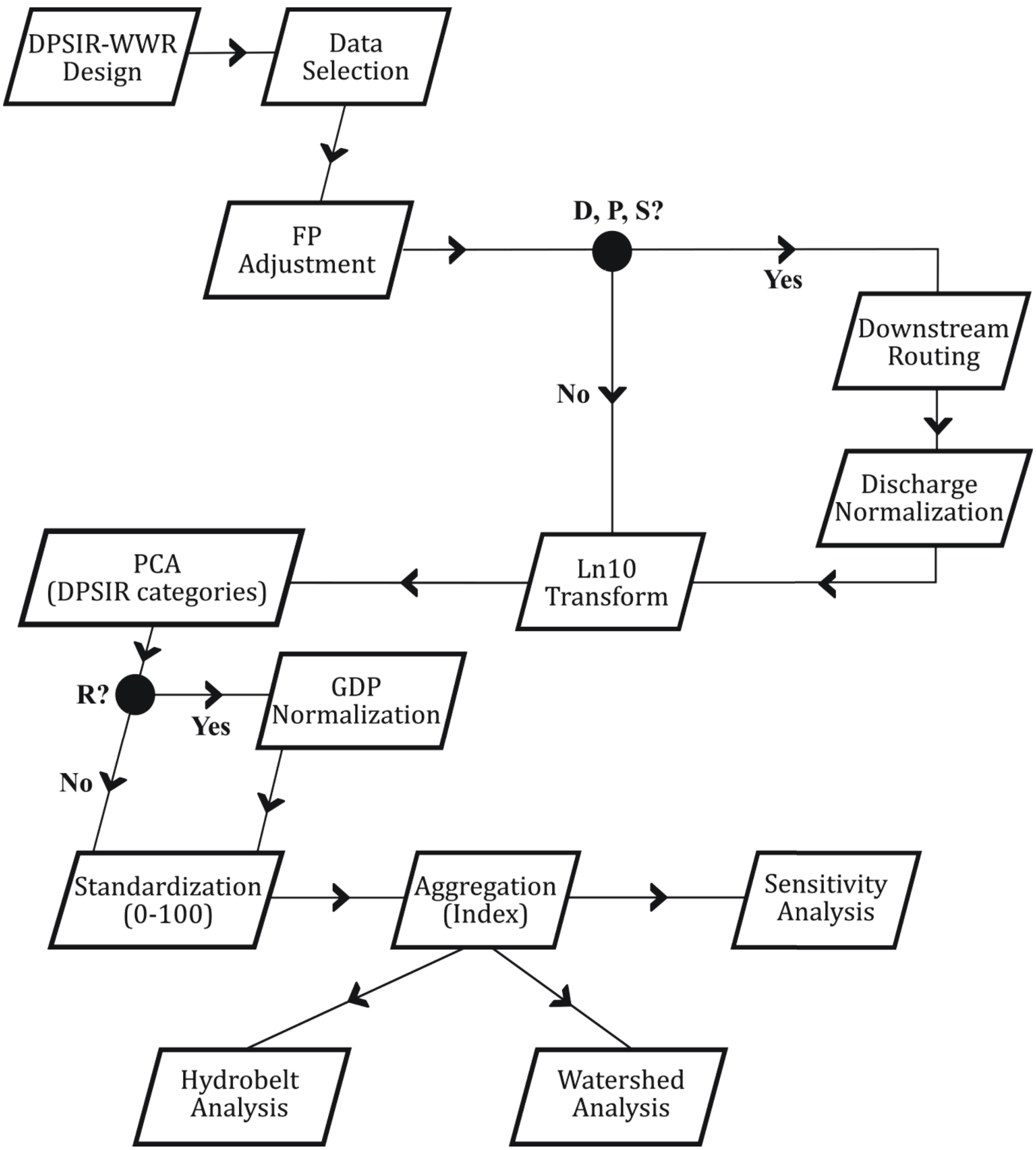


Figure 3-2: *Geoprocessing steps for the creation of the global WWR index. 'FP' stands for fire probability, 'Ln10' for logarithm base-10, and 'PCA' for principal component analysis. If indicators were part of the Driving Forces, Pressure, or State categories they were processed through the downstream routing and river discharge normalization steps.*

into homogeneous hydrological regions", based on the merging of non-glaciated continental river basins showing a similar hydroclimatic regime (Meybeck *et al.* 2013). We extracted the weighted raster values of our global indicators within each hydrobelt to examine the individual contribution of each DPSIR category to the final index scores, thus comparing the controls of the risk within and between hydrobelts. We applied the same extraction method to the watersheds of 16 arbitrarily selected cities whose surroundings are regularly affected by wildfires. Those cities represented a diversified sample of environmental conditions (human and natural) found around the world. Watershed boundaries were derived from the Aqueduct 2.1 dataset developed by the World Resource Institute (Gassert *et al.* 2014).

3.6 RESULTS

Values of the global composite index of the wildfire-water risk range from 0.25 to 77.27, with a mean of 18.11 and a standard deviation of 12. A closer look indicates that ~3.5% (Score >= 40) of the global area is at a substantially greater risk from wildfire impacts on water than other regions of the world. However, approximately 45.5% of the terrestrial area of the earth is at a moderate risk (Score = 20-40), while ~51% is at a relatively low risk (Score < 20). Greater risk scores are mostly found in the continental parts of the Northern hemisphere around the Great Plains of North America and Interior Alaska; Central Asia; North-Eastern China; Mongolia; in the Yakutsk basin in eastern Russia (Fig.3-2). The Iberian Peninsula, Eastern Europe and Anatolia, and a few clusters in Africa, South America, India and Australia show greater wildfire-water risk values. The index shows that moderate risk is common at tropical and intertropical latitudes, as well as in Eastern North-America, in western and northern Europe, and the large continental plains of Eurasia. The majority of low risk scores are found in the Equatorial belt, especially in the Amazonian forest, with scores between 10 and 20; in large portions of the circumboreal forest with scores falling under 10 when approaching the Arctic Circle, and many mountain ranges such as the Alps or the Carpathian mountains. Patagonia and northernmost tundra steppes of North America and Siberia show a score lower than one. The coast of the Gulf of Alaska, the Plateau of Tibet, and a large part of Central Asia also show similarly low levels of risk.

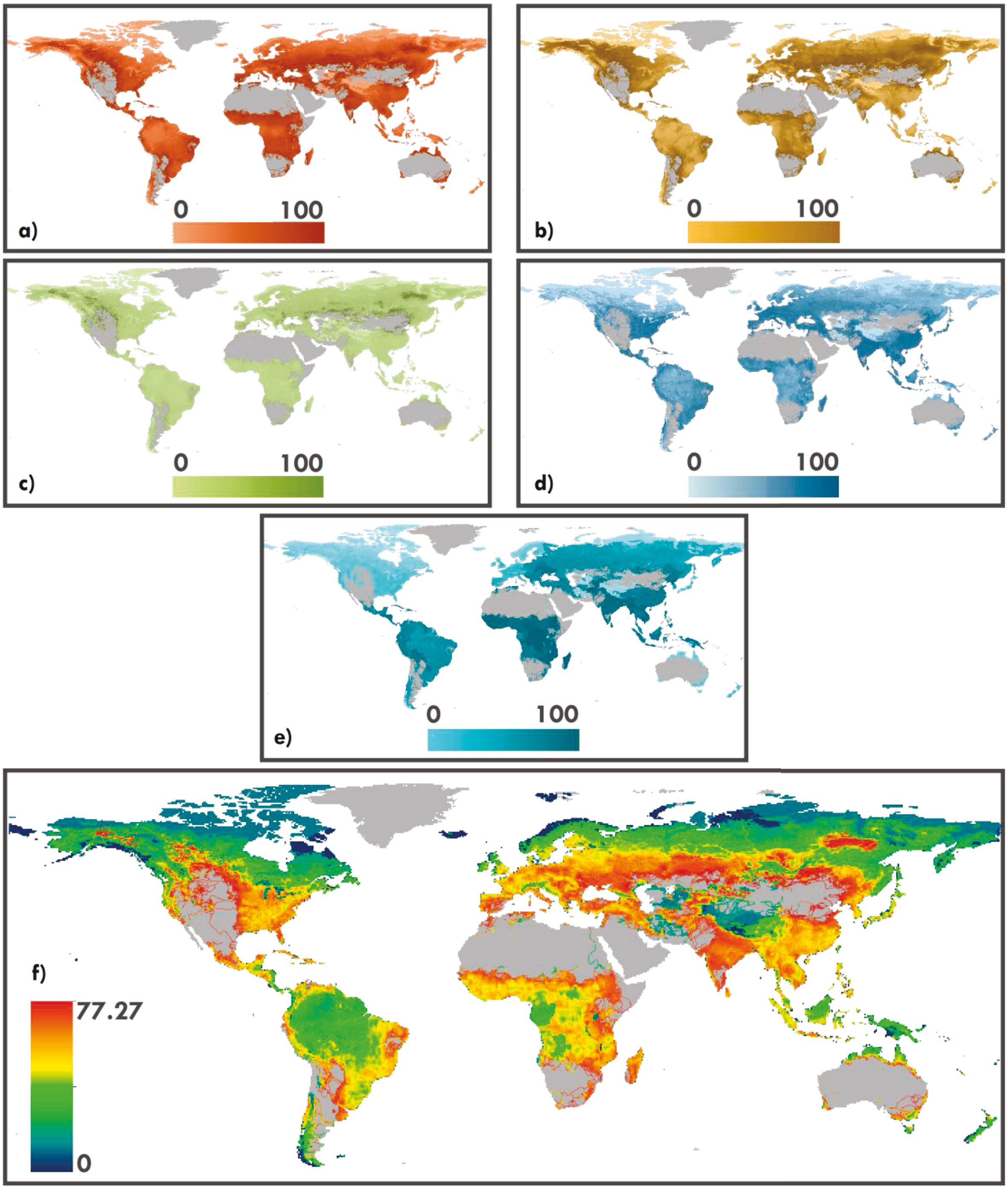


Figure 3-3; *Maps of the standardized global indicators, with: a) Drivers, b) Pressure, c) State, d) Impact, e) Response; and f) the final WWR index (unitless) resulting from the weighted sum of the global indicators. The color palette used for maps a) through e) is the same as Fig1. The color scheme applied to the final index follows an equalization stretch of the histogram to enhance the contrast between scores.*

A closer look at the WWR index scores by hydrologic belt (Fig.3-4) reveals the controls of the risk in the different regions of the world. The boreal (BOR) belt shows an important variability in the final score values, although scores remain less than 20). The individual contribution of DPSIR global indicators is larger for Impacts and Pressure. The

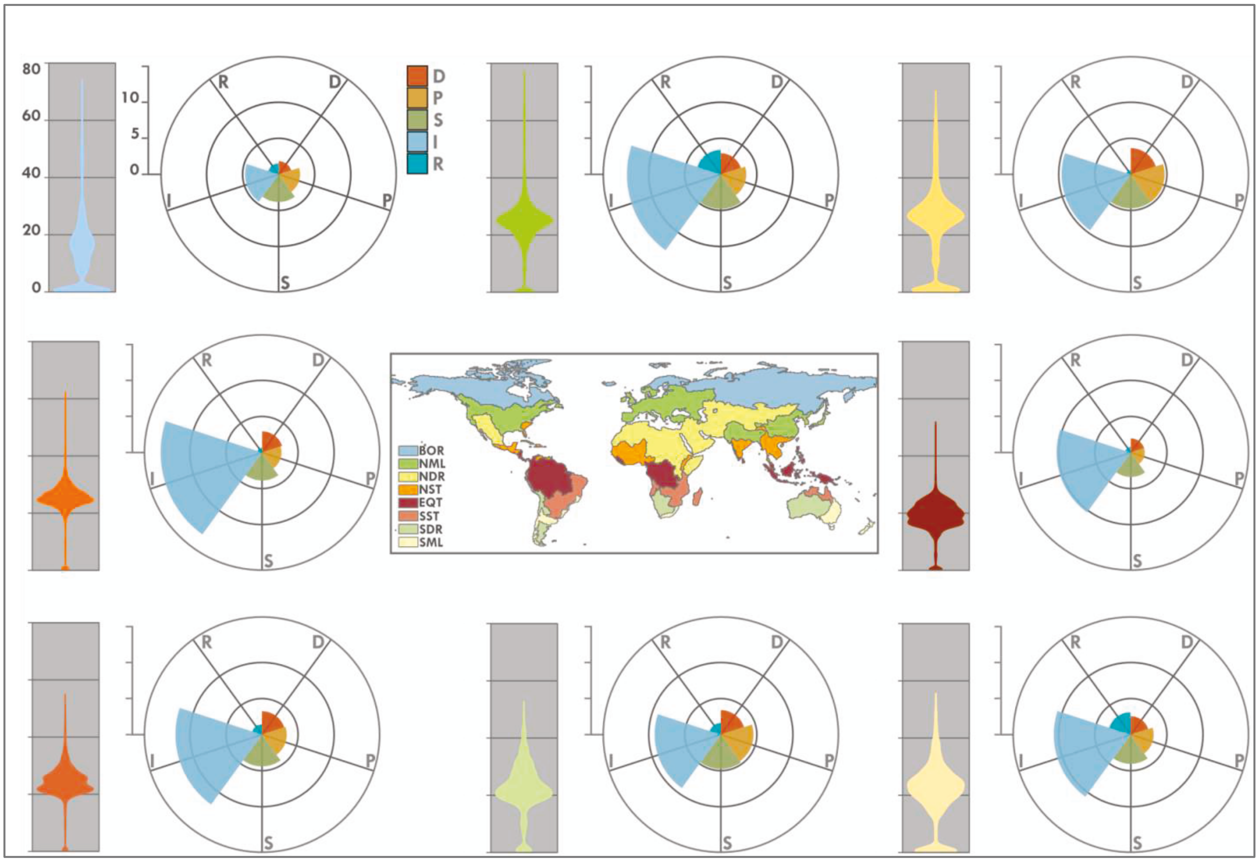


Figure 3-4: *Details of the WWR per hydrobelt, as shown in the centre map. Violin plots show the distribution of the WWR index scores for a hydrobelt, whereas polar plots show the average contribution of each DPSIR category to the final score for the same hydrobelt. The color palette used for each category is the same as Fig.1. For readability purpose, the Response category presented here has been GDP-adjusted (multiplied), so higher values show higher response capacities, whereas the final index score uses GDP-normalized values (see Methods). The hydrobelts are: BOR=Boreal; NML=North Mid-Latitude; NDR=North Dry; NST=North Subtropical; EQT=Equatorial; SST=South Subtropical; SDR=South Dry; SML=South Mid-Latitude.*

North and South mid-latitude (NML and SML) and the South subtropical (SST) belts show a similar pattern in the contribution of global indicators, with an overarching dominance of the Impact category (average scores are ~12.5, ~10, and ~12 respectively), followed by the State category (average score is ~5 in all three regions). The Response category remains well represented for NML and SML (average score is ~2.5) but is lower than 2.5 for SST. However, NML shows a larger variability in final scores whereas SML and SST show lower final scores in general. The North dry (NDR), North subtropical (NST), and Equatorial (EQT) belts are all characterised by a quasi-absence of response values and a dominance of the Impact category, especially for NST and EQT (average score are ~12.5 and ~10 respectively). However, the distribution of index scores is highly variable between these three belts, with NDR showing in general higher scores. The Southern dry (SDR) shows a singular pattern with the impact and response categories having an equal contribution to the final score (average score is ~7) and final risk values clustered around 20.

The fine-scale analysis of 16 watersheds (Fig.3-5) in regions known for their fire activity confirms the pattern shown at the hydrobelt level, underlining the importance of the impact and resilience categories in the control of final risk scores. Haifa, Marseille, Melbourne, and San Francisco, despite high levels of impacts, see their final score diminished because of their response capacity, whereas Guadalajara, Istanbul, Pune, Palangkaraya, and Quito do not show this response capacity for similar levels of impact. The cases of Denver, Fort McMurray, and Yakutsk show a different pattern where the drivers, pressure, and state categories account for a greater role in the final score, although the low response capacity coupled with higher Pressure levels in Yakutsk probably explains the final highest score (46.2).

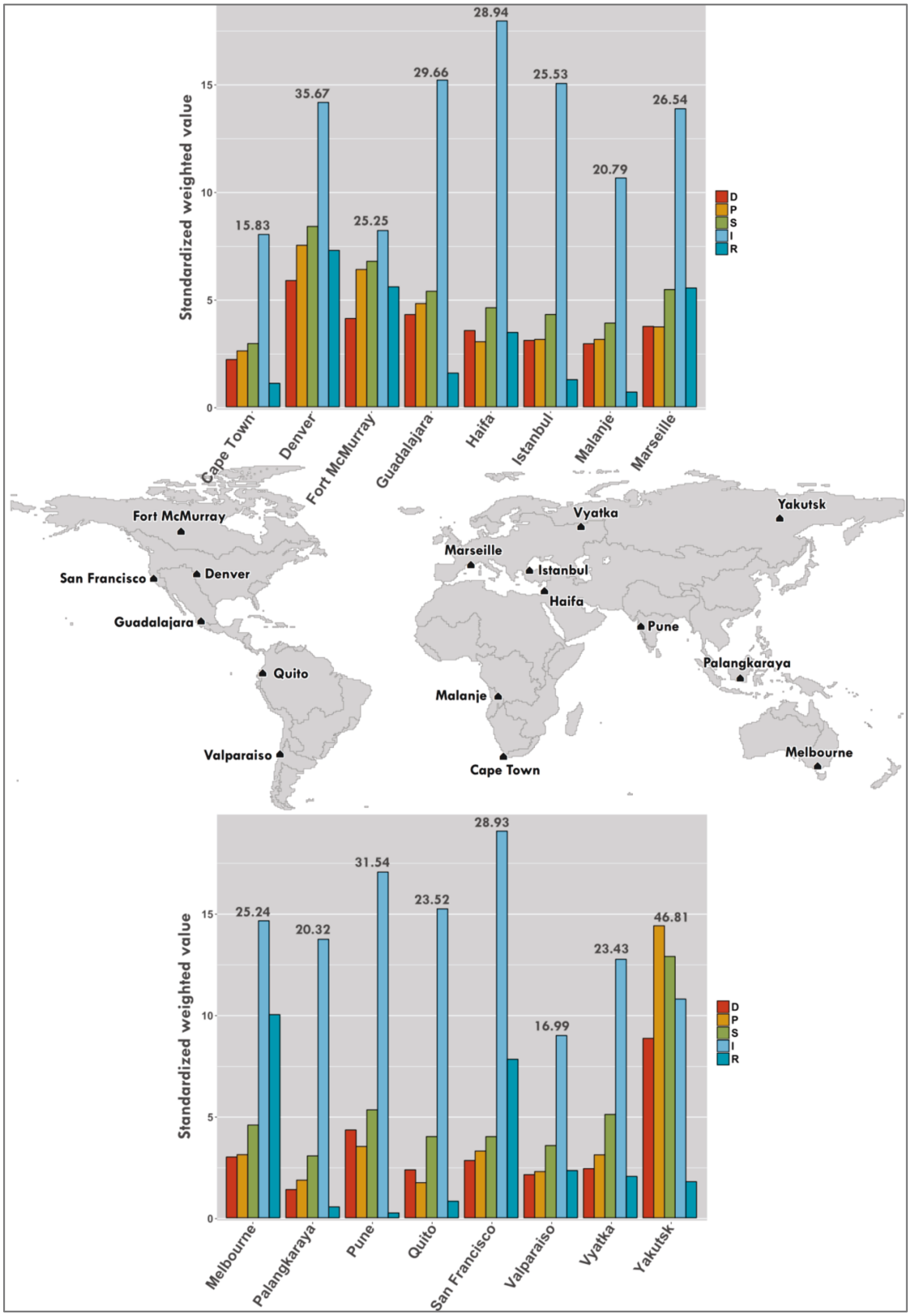


Figure 3-5: *Detail of global indicators' values for 16 selected watersheds across the world. The X-axis shows the mean value of each DPSIR global indicator for each watershed. The color palette used for each category is the same as Fig.1. The number at the top of each graph shows the average risk score for the watershed. For readability purpose, the Response category presented here has been GDP-adjusted (i.e. multiplied), so higher values show higher response capacities, whereas the final index score uses GDP-normalized values (see Method).*

3.7 DISCUSSION

The creation of a spatial index showing the geography of wildfire-water risks to water security was motivated by three objectives: to create a robust framework, to study the WWR's geography, and to raise awareness about the WWR. We believe that the DPSIR framework for the analysis of the WWR is robust, in a sense that it offers the possibility to capture a large range of processes, though with an inherent flexibility that make it adaptable to the amount of information available, which is in line with other studies presenting the DPSIR as a useful tool for the development of environmental indicators, from a global (Freudenberger *et al.* 2010) to a city scale (Jago-on *et al.* 2009). The global geography of the WWR displays similar risk levels among many regions of the world, important information that suggests there are opportunities to transfer skills and technologies from WWR-prepared countries, like USA or Australia, to unprepared countries. However, such transfers would require adaptation to regional and local socio-ecological settings. Indeed, according to our results, the top-down controls of the risk result in important geographic discrepancies driven by three major aspects: the size of the exposed population and the magnitude of other values at risk, the capacity of exposed human and natural communities to face the risk and respond to a disaster situation, and the gravity of post-fire hazards. In this respect, the index shows commonalities with other studies related to water security and wildfire risk, in which regionally strong population growth and deficient economies drive the exposure and the vulnerability to the risk (Chuvieco *et al.* 2014; Gain *et al.* 2016; Veldkamp *et al.* 2016). The potential impacts on freshwater ecosystems could also endanger critical food sources for more than 150 million people around the world, a majority of them living in developing countries (McIntyre *et al.* 2016). Nonetheless, our results also suggest that developed regions, such as North America and Europe, may not be immune to the hydrogeomorphic consequences of wildfires, as shown by Emelko et al. (2015) in Canada and White et al. (2006) in Australia, although the inference space of our index remains limited. Nonetheless, those possible consequences may exacerbate existing water challenges linked to the increasing water demand or the ongoing degradation of freshwater resource quality (McDonald *et al.* 2014; Schewe *et al.* 2014; Green *et al.* 2015). Our work underlines the connections between fire, water, and soils at a global scale, adding to the threats to rivers systems listed by Vörösmarty et al. (2010) and Ceola et al. (2015). Combined with other decision-support tools, the application of our framework adapted to the WWR can feed further thinking on the integration of wildfire risks into water security governance (Bell 2012; Tscherning *et al.* 2012).

More than a half of human population now lives in urban areas, following a global urbanization trend that is expected to continue (Seto *et al.* 2011). Meanwhile, there is an increasing concern as to the vulnerability of ever-growing cities to natural hazards and water supply disruption (Jackson 2006; Hoekstra and Mekonnen 2012). Several recent initiatives, such as Global Forest Watch-Water from the World Resource Institute (Qin *et al.* 2016), Urban Water Blueprint from the Nature Conservancy (McDonald and Shemie 2014), and 100 Resilient Cities (http://www.100resilientcities.org) have identified cities whose watersheds are exposed to wildfires. Our results confirm the risk posed by wildfires in several water basins supplying surface water resources to large urban areas, which emphasise the importance of considering the WWR in enhancing city resilience (Kinoshita *et al.* 2016; Martin 2016). The historical fire season experienced by Chile in early 2017 provides further demonstration of existing interconnections between wildfire activity and water security. As the fire was spreading through a scorched countryside, it damaged numerous water distribution facilities, consequently limiting the water supply to firefighters already challenged by water shortages due to non-reliable water distribution systems. Subsequent rainstorms in the widely burned Maipo River watershed, supplying Santiago, caused landslides and floods that further disrupted water supply to 5 million people.

The protection of watershed's natural capital has been overwhelmingly supported to assure the long-term provision of freshwater ecosystem services in urban areas (Muning *et al.* 2011; Andersson *et al.* 2014). Emelko et al. (2011) emphasize the need for source water protection from severe wildfire events, a statement enforced by the recent report 'Beyond the source' by the Nature Conservancy (The Nature Conservancy 2017) which places wildfires as a critical threat to freshwater services. However, the expansion of human population in natural areas leads to a multiplication of wildland-society interfaces that favors increased wildfire activity (Le Page *et al.* 2010), especially in developing countries (Aldersley *et al.* 2011). Burke et al. (2013) also point to the post-fire water pollution from interface fires in urbanized areas where anthropogenic pollutants, such as heavy metals, can substantially leach. Our results advocate for an enhanced protection or restoration of ecosystem services in watersheds supplying critical freshwater resources using finer-scale spatial modeling, as well as support the development of drainage-basin-scale hydrological studies integrating fire activity as a potentially important control of discharge and sediment flow.

Our study raises questions about the future of wildfire-water risks to water security in a context of global environmental change. A growing number of studies document the vulnerability of coupled human-and-natural systems to these changes (Rockström *et al.* 2009) and the future associated challenges of water security they will have to overcome (Bogardi *et al.* 2012). The predicted alteration of climate, land use, and human demographics will affect fire activity (Flannigan, Stocks, *et al.* 2009), natural habitat health (Seto *et al.* 2012), soil properties (Hicks Pries *et al.* 2017), and water availability (Van Vliet *et al.* 2013). These ongoing modifications (Dodds *et al.* 2013; Jolly *et al.* 2015) will certainly change the nature of the WWR (Bladon *et al.* 2014). Developing countries, which are already the most vulnerable according to our results, will probably experience even more constant and pervasive effects of climate change (Harrington *et al.* 2016). In this respect, the WWR should be seen as an emerging risk, whose identification is one of the priorities of the 2015-2030 Sendai Framework for Global Disaster Risk Reduction (UNISDR 2015). Therefore, our results provide a baseline for scenario-based exploration of future global changes, and the DPSIR framework provides a ready-to-use tool for benchmarking the fate of the WWR to water security using enhanced or updated information layers showing predicted changes as a function of policy evolution or global environmental change.

Although the weighting scheme is a common drawback of indexation work (Vörösmarty *et al.* 2010b; Gain *et al.* 2016), We strived to keep our scheme as objective as the available information would allow. Further expert validation would help to integrate other indicators and to combine them in a different fashion. For instance, indicators only appear once, but many post-fire hydrogeomorphic phenomena and their consequences can overlap several DPSIR categories and thus could be integrated several times. Moreover, any global approach depends on the use of proxies, and therefore is subject to interpretation as to the relevance of any indicator. We chose to represent the main post-fire dynamics and water security constraints as presented in the literature, but this information can vary widely among locations and therefore must be kept in mind when interpreting the results presented here. Furthermore, this information is lacking in many parts of the world, a critical aspect that highly limit the ability of our proxies to provide a complete and accurate picture of the likely consequences –or absence of consequences– of wildfires on renewable freshwater resources. Therefore the inference space of our index remains fairly narrow as to the water provision challenges that may be imputed to wildfire activity and its subsequent implication for people water supply. Our results should therefore be interpreted with caution and only be used for communication and information purposes in their current form, as they are based on a generalization of post-fire phenomenon that can display important regional-to-local variability, and thus are not design for active management purposes.

The accessibility to adequate data is another classic limitation of such work (De Bono and Mora 2014). Despite an increasing availability of data at a global scale, many datasets relevant to this study were produced one to two decades ago. Many data are available only in an aggregated fashion (i.e. country-scale) and cannot be used directly in spatially explicit approaches. Other variables, such as firefighting expenditures, technological water-treatment capacities, or human and economic losses specifically related to the WWR are simply nonexistent at a global scale. It is interesting to note that in the "big data" world, we still lack spatially explicit (i.e. pixel-based) global datasets that are regularly updated, especially those representing social indicators, and that many parts of the world suffer from a deficit in scientific information (Leidig *et al.* 2016). The science of post-fire hydrogeomorphology is well developed, but there is a need for a global database that maps key WWR measures. In this regard, the database of post-fire debris flows in the Mediterranean region started by Parise and Cannon (2003) is an encouraging effort.

3.8 CONCLUSION

The work presented here gives a global overview of the wildfire-water risk to water security. As any global index, its primary aim resides in giving an overall perspective of an issue that could affect most parts of the globe. In line with the Sendai Framework for Disaster Risk Reduction, we used the DPSIR analysis framework to provide new insights and raise awareness about this emerging risk. Although actual risk management actions take place at finer scales, a global view of this growing concern offers a new facet to consider in the governance of water-related risks. Our spatial index may help further investigate hydrological systems where the water supply is already under pressure because of urban development, ecosystem degradation, or climate change. As indexes are geared toward environmental performance improvement, our framework introduces a tool for long-term monitoring of actions towards the reduction of post-fire threats to water security. Our work could also help to reconsider the place of fire in the landscape and to foster the use of “good fires” as a means to preserve water-related ecosystem services. We believe that our results represent an important contribution to the current knowledge of the global geography of risk, as well as provide new insights for the achievement of global water security.

3.9 ACKNOWLEDGEMENTS

We want to thank the numerous researchers who provided data for this analysis: Ben Collen from the Zoological Society of London; Ben Poulter from Montana State University; Fay Johnston from University of Tasmania; Leho Tedersoo from University of Tartu; Huan Wu from University of Maryland; Yukiko Hirabayashi from University of Tokyo; Victoria Naipal from the Max Planck Institute for Meteorology.

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CHAPTER 4- ANTHROPOGENIC INFLUENCE ON WILDFIRE ACTIVITY IN ALBERTA, CANADA

4.1 ABSTRACT

The boreal forest of Alberta, Canada, is under pressure from a rapid expansion of the wildland–human interface driven by natural resources exploitation. The specific impact of these changes on area burned remains poorly understood. We addressed this issue by modelling area burned for the 1980–2010 period using variables accounting for various anthropogenic effects. We hypothesize that an ecological frontier exists in the areas of intermediate to low human influence in Northern Alberta, which implies a new influx of human-caused ignitions coinciding with continuous flammable vegetation, hence promoting area burned. Using a statistical control approach, we assessed the importance of each anthropogenic variable by adding them to a biophysical regression model. Our results show that there is a diversity of responses of area burned to the different anthropogenic factors considered. Distance to transportation network, human footprint and density of the energy network significantly improved the model predictions. The area burned in the ecological frontier showed clusters of higher predictions by anthropogenic models, which supports our hypothesis of an ecological frontier and suggests that human and natural ignitions have an additive, albeit temporary, effect on landscape fire susceptibility.

4.2 INTRODUCTION

The boreal forest of western Canada is currently subjected to increasing pressure from large-scale anthropogenic activities related to natural resources exploitation (NRTEE 2005; Bogdanski 2008; Government of Canada 2009). In parallel, urban development and extensive agriculture are still occurring in many areas of the country (www.statcan.gc.ca, accessed 7 October 2015), mainly at the southern fringe of the boreal forest. In areas of profound anthropogenic landscape modifications, wilderness and human features overlap, which applies pressure from different types of human-caused disturbances. Among the numerous ecological consequences of these mutations, changes in the natural Boreal fire regime are today fully acknowledged, yet not well understood (Johnson *et al.* 1998; Gralewicz *et al.* 2006, 2012; Parisien *et al.* 2011; Butsic *et al.* 2015). These modifications usually translate into a higher ignition density at the vicinity of human activities yet a lower area burned as anthropogenic landscape fragmentation (e.g., roads) limit fire spread and makes fire detection and fire suppression more effective (Stocks *et al.* 2002; Syphard *et al.* 2007; Gralewicz *et al.* 2012). However, this general rule of thumb might hide a diversity of spatial patterns, depending on the type and intensity of human activities.

The province of Alberta, in western Canada, has experienced an important industrial development linked to the extraction of large oil and gas reserves and the exploitation of an abundant timber resource over the last few decades. This development has been rapidly extending northward, leading to the appearance of an ecological frontier (Héritier 2007). Based on the geographic concept of the pioneer front, an ecological frontier has a regional operational scale – usually hundreds of kilometers deep and thousands of kilometers long – with fuzzy limits, and is defined as a natural space under extensive but regulated industrial development. It is characterized by a low but growing population density, a penetrating low-density road network, an intensification of natural resource exploitation, tensions between stakeholders, and a large range of ecological consequences (Schneider *et al.* 2003; Alberta Sustainable Resource Development 2008a; Guyot 2009; Héritier 2009). Relating the existence of an ecological frontier to changes in wildfire activity seems like a logical statement, yet no study addressing this specific relationship has been undertaken to this day in the province. In addition, it appears that fire activity is not always understood within this spatial range (Calef *et al.* 2008; Wang and Anderson 2010; Gralewicz *et al.* 2012).

The traditional use of the wildland-urban interface (WUI) notion, which primarily relies on housing location and density (US Departement of Agriculture - Forest Service 2001), is not appropriate for the proposed study area, which is characterized by the entanglement of extensive wildland-industrial interfaces, wildland-urban-rural interfaces (Gonzalez-Caban and Omi 1999), and wildland-infrastructure interfaces (e.g. roads). In this study, we introduce the notion of wildland-human interface (WHI), simply defined as zones of contact between the wilderness and any human-made feature. We can therefore integrate human influence in fire-prone areas that are mostly not urban, and where the impacts on fire regimes and the consequence on humans are thus different than many of those associated with the WUI (Radeloff *et al.* 2005; Theobald and Romme 2007). Though general, this notion appears more flexible than the WUI. It can potentially improve fire modeling efforts by including more anthropogenic effects (Mann *et al.* 2016), at multiple spatio-temporal scales, and thus yield better prediction of changes in wildfire regimes in areas of increasing industrial activity (Alberta Sustainable Resource Development 2008a; Pickell *et al.* 2015).

Our study investigates the potential effect of the anthropogenic development of Alberta on the provincial wildland fire area burned. Based on the apparent existence of an ecological frontier extending northward, with a widespread but low-density anthropogenic footprint amongst large patches of continuous vegetation, we hypothesized that this transitional space should promote an increased area burned. This hypothesis would demonstrate that frontier dynamics result in non-monotonic shifts in area burned, as opposed to the more common monotonic decrease observed elsewhere when anthropogenic dynamics are involved. Specific objectives of the study consist of: (1) modeling and predicting area burned from 1980-2010 in Alberta, (2) evaluating the role of several anthropogenic variables in predicting area burned, and (3) spatially assess predictions of total area burned to detect changes linked to the existence of an ecological frontier. We assessed the influence of human factors on the Total Area Burned (TAB) for the 1980-2010 time period using a statistical approach with which we control for the effect of climate variables. We discuss how natural fire regimes may shift toward anthropogenic fire regimes in the context of the extensive current industrial development of the Boreal forest of western Canada.

4.3 MATERIALS AND METHODS

4.3.1 STUDY AREA

The study area covers the province of Alberta (total area 661,848 km²), in western Canada (Fig.4-1 a). This zone is characterized by a continental climate with short and warm summers and long cold winters. For our study period, the mean winter temperature was -

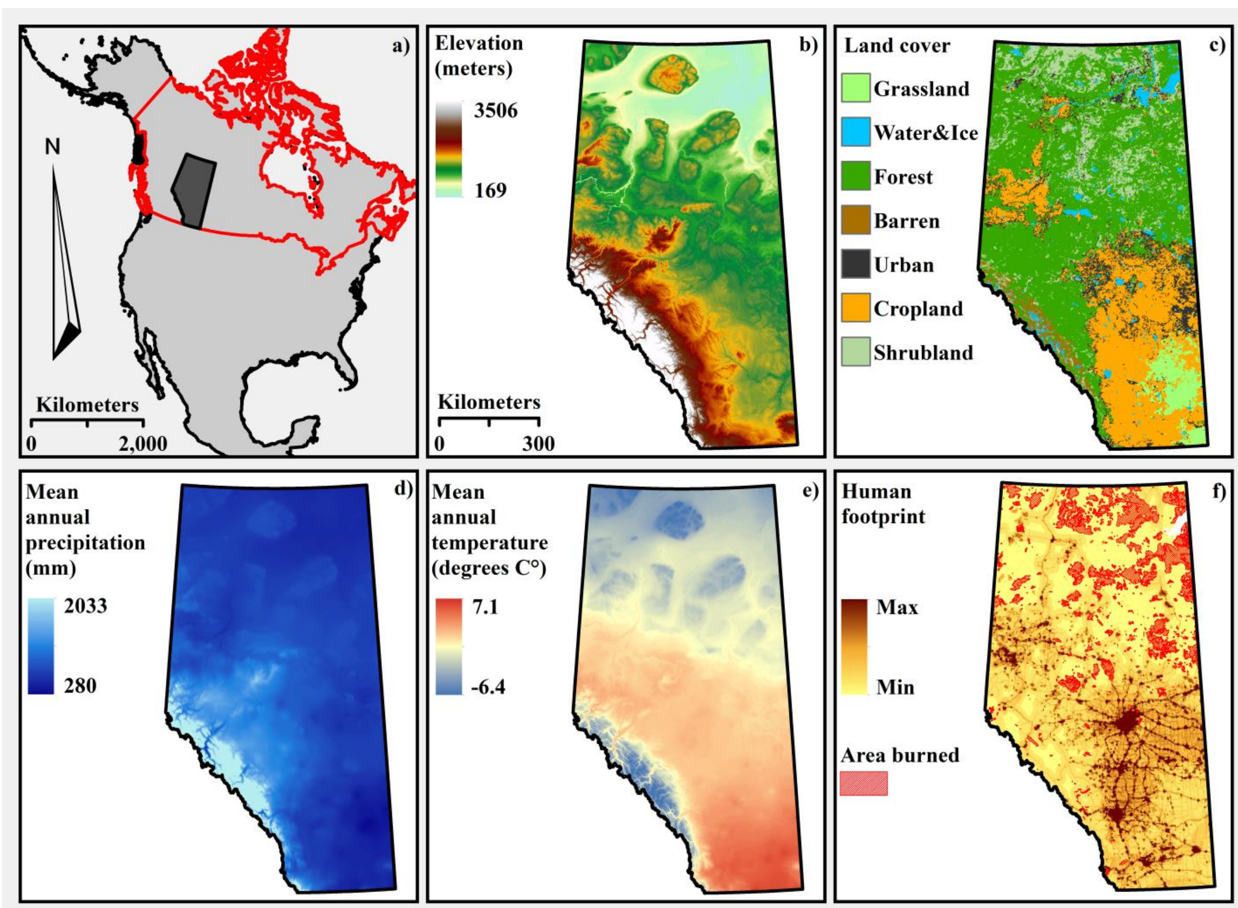

Figure 4-1: *Characterization of the study area. a) The location of Alberta in western Canada, with b) Elevation, c) Simplified land cover, d) Mean annual precipitations, e) Mean annual temperature, f) Human Footprint Index and area burned (1980-2010).*

12.7 °C and the mean summer temperature was 14.6 °C. Mean winter and summer precipitations were 63 and 204 mm, respectively. There are strong topographic and latitudinal climatic gradients across the study area (Fig.4-1 b-e), characterized by a transition from low and flat elevations in the north (lowest: 169 m) toward the hilly slopes of the foothills, and finally the high peaks of the Rocky Mountains (highest: 3506 m) in the southwest. The study area is mostly covered by a mosaic of coniferous and mixedwood forests, with white spruce (*Picea glauca*), black spruce (*Picea mariana*), jack pine (*Pinus banksiana*), lodgepole pine (*Pinus contorta*), and trembling aspen (*Populus tremuloides*) as dominant species.

The fire season is fairly long, starting early-April and ending late-September. Over the 1980-2010 period, 69% of recorded fires over 200 hectares were naturally ignited, 25.5% are human-caused, and the rest were of unknown origin (Natural Resources Canada 2015). The total area burned was 6,542,747 hectares, with 4,389,913 hectares (67%) from lightning-caused fires. This said, there is spatial variability in the distribution of area burned, with the northern forests experiencing most of the fire activity, whereas the foothills and the Rocky Mountains have little activity in comparison (Tymstra *et al.* 2005).

The province of Alberta has been experiencing major economic growth since the early 1990s, mainly due to the development of the oil-and-gas industry, though forestry and agriculture also play a role in Alberta's economy. This growth has been accompanied by a sizeable increase in the human population (almost 2%/year), thus enlarging existing settlement and industrial fronts in forested areas (Lee *et al.* 2009; Pickell *et al.* 2014). With a current population of 4 million, concentrated in the southern half of the province, demographic projections estimate a maximum of 7.3 million people by the year 2041 (Alberta Treasury Board and Finance 2014). Hence, this spatial dynamic is likely to progress in the coming decades, with the potential for generating an extensive and pervasive amount of human-related impacts to natural areas (Fig.4-1 e).

4.3.2 DATA

We built multivariable regression models based on the TAB over the 1980-2010 period, notwithstanding of fire causes. Based on the literature, we selected several geospatial datasets to derive a suite of biophysical and anthropogenic explanatory variables (Table 4-1). All explanatory variables were at a 1-km spatial resolution raster format and projected using the Alberta 10° Transverse Mercator (Forest) coordinate system.

For the purpose of this study, a 50 km-width (216,506 ha) hexagon vector grid (hereafter hexels) was generated using the tool 'Repeating shapes for ArcGIS' (Jenness 2012). At the scale of an ecological frontier – i.e. regional – studies have shown that such resolution represents a good compromise between capturing sufficient environmental variability and avoiding too much loss of information due to averaging (Parisien *et al.* 2011), while limiting spatial autocorrelation in model residuals (Fortin 1999; Dungan *et al.* 2002). For each hexel, we computed the total area burned and the average value of each explanatory variable. Spatial data preparation was conducted using ArcGIS 10.1 (Environmental Systems Research Institute 2012) and statistical analysis was carried out using R 3.1.1 (R Core Team 2014).

Table 4-1*: The variables used to model Total Area Burned in Alberta, 1980-2010. All anthropogenic variables have been tested one by one using a statistical control approach.*

Variable	**Type**	**Units**	**Mean (range)**
Temperature	Biophysical	°C	17.6 (1.6-23.3)
Diurnal range	Biophysical	°C	11.7 (3.9-15.2)
Climate moisture index	Biophysical	mm	-1.2 (-7.4-18.9)
Precipitation	Biophysical	mm	60.7 (38.2-141.1)
Distance to temporary (daily) fire attack bases	Anthropogenic	km	37.6 (11-156.9)
Distance to primary fire attack bases	Anthropogenic	km	76.9 (18.4-19)
Distance to secondary fire attack bases	Anthropogenic	km	14.4 (17.7-46.7)

Distance to fire attack bases	Anthropogenic	km	34 (11-15.6)
Distance to buildings	Anthropogenic	km	63.52 (7.4-31.8)
Distance to mines	Anthropogenic	km	54 (0-284.2)
Distance to cutblocks	Anthropogenic	km	53.3 (0-310)
Distance to transportation network	Anthropogenic	km	4.6 (0-88.5)
Transportation network density	Anthropogenic	m/ha	1.16 (0-13.6)
Energy network density	Anthropogenic	m/ha	0.54 (0-4.2)
Seismic line density	Anthropogenic	m/ha	1.35 (0-17.37)
Population per square kilometer	Anthropogenic	People/km^2	4.32 (0-1310.38)
Anthropogenic non-fuel	Anthropogenic	%	31 (0-100)
Human Footprint Index	Anthropogenic	Unitless	N-A

Fire. Fire data used to produce our dependent variable, TAB, were obtained from the Canadian National Fire Database (Natural Resources Canada 2016). We focused our study on the forested natural subregions of the province where most fires occur, which encompasses the Boreal, the Foothills, and the Rocky Mountains natural sub-regions (Downing and Pettapiece 2006). We excluded fires <200 hectares, as they are inconsistently reported; however, in Boreal Canada fires larger than this size account for only ~3% of all fires yet are responsible for ~97% of the total area burned (Stocks *et al.* 2002). In order to limit the influence of inconsistencies in burned-area mapping (e.g., discrepancies in fire perimeters precision and location, absence of unburned islands), the TAB does not consider areas burned more than once. Given the low proportion of re-burned areas in our database (<5%), it is unlikely to affect the results of this study.

Biophysical. We selected a set of four climatic variables known to influence area burned at intermediate-to-large spatial scales and computed their average over 30 years (1981-2010). For the fire season, the maximum temperature (Parisien *et al.* 2011), the average precipitation (Whitman *et al.* 2015), as well as the climate-moisture index (i.e., precipitation minus potential evapotranspiration) (Hogg 1994) as a proxy for drought conditions (Flannigan and Harrington 1988; Wotton *et al.* 2003; Littell *et al.* 2009) were computed using ClimateWNA (Wang, Hamann, *et al.* 2012). We also included the diurnal range of temperatures as a yearly average based on data provided by Natural Resources Canada (McKenney 2006), as a proxy for continentality.

Anthropogenic. We wanted to account for human influence on the area burned by including both anthropogenic variables that can, in principle, limit fire activity, favor it, or have a dual (i.e. nonlinear) role (Parisien *et al.* 2012). The location of fire suppression attack bases, a wildfire limiting agent, was provided by the government of Alberta. From these locations we derived several distance rasters representing different aspects of the suppression network: primary attack bases and secondary (permanent–semipermanent) attack bases, and temporary bases, which are those that are deployed to the sites of campaign fires. Population density was extracted from Landscan data (UT-Battelle LLC 2013) ranging from 2000 to 2010. We also extracted Human Footprint Index values, a measure of human influence on ecosystems (Sanderson *et al.* 2002), using the Global Human Footprint V2, a regional-scale measure of landscape anthropization, released by the Center for International Earth Science Information Network (Wildlife Conservation Society and Center for International Earth Science Information Network - CIESIN - Columbia 2005). We finally used the Long Term Satellite Data Records project (Latifovic *et al.* 2009) from 1985 to 2005 to derive anthropogenic permanent non-fuel data (e.g. water bodies, barren lands, glaciers, crops, urban areas).

Using the Canadian Vector database (Natural Resources of Canada 2013), we computed the density of the transportation network (roads and railways) using a moving window of 10^5 km to account for regional scale influence. We also considered the distance to both urban and industrial buildings. Using the Alberta Biodiversity Monitoring Institute (2012) wall-to-wall land cover inventory data, which was derived from high-resolution satellite imagery, we calculated the density of seismic lines, the distance to cut-blocks, and the distance to mining activities. Seismic lines, cut-blocks and mines are characteristic anthropogenic disturbances of many Alberta forests, resulting from resource prospection and extraction (Pickell *et al.* 2015), although detailed information about the age of these disturbances, and hence the stage of vegetation recovery, remain fragmented.

4.3.3 STATISTICAL MODELING

Model building. We created a biophysical generalized linear model (GLM, Gaussian family) for TAB, based on climatic variables that are known to influence fire activity at the spatial extent of the study area, hereafter called biophysical model. Our approach assumes a simple functional form, with area burned increasing for warmer and drier climatic conditions (Flannigan and Harrington 1988; Littell *et al.* 2009), which are represented in our model by changes in average temperature, climate moisture index and precipitation, whereas including the average diurnal range of temperature gives a measure of continentality (Eqn 1).

$$\sqrt{TAB} = \beta 0 + \beta 1\,Temp + \beta 2\,Temp^2 + \beta 3\,CMI + \beta 4\,Precip + \beta 5\log(DR) + \varepsilon \qquad (1)$$

With *Temp* being the temperature, *CMI* being the climate moisture index, *Precip* being the precipitation, log(*DR*) being the natural logarithm of the diurnal range, and ε the error term. The area burned was transformed using a square root function to homogenize the variation in residuals. We selected fires that occurred between 1980 and 2010, and analyzed them according to their cause (Table 4-2). Starting with 283 observations, we dropped hexels that had an area <10% within the study area, yielding 271 hexels that were used for model building. We used the original 283 hexels to develop predictions in order to have a full coverage of the study area. We explored regression models of area burned and ignition frequency stratified by ignition cause. However, these models did not produce results satisfying enough (see appendices 1-2) for our purpose. Therefore, we used a model of area burned accounting for all ignition causes, which provides more robust results, assuming that most anthropogenic variables may affect all fire, regardless of the cause of ignition.

Table 4-2: *Fire activity for 1980-2010 as reported by the Canadian National Fire Database for fires >200 hectares.*

Cause	Total Area Burned (ha)	Total Ignition Number
Human	677,284	315
Natural	4,389,913	851
Unknown	1,475,550	66
All	6,542,747	1232

We then analyzed TAB as a function of biophysical plus individual anthropogenic variables, acknowledging the multiple effects of human on fire activity, hereafter called anthropogenic models, but in proportions that have not been analyzed in Alberta. Therefore, each anthropogenic variable is included one by one to the existing biophysical model, testing for decrease in TAB with increase in seismic line density for instance (Eqn 2).

$$\sqrt{TAB} = \beta 0 + \beta 1\, Temp + \beta 2\, Temp^2 + \beta 3\, CMI + \beta 4\, Precip + \beta 5 \log(DR) + \beta 6\, AV + \varepsilon \quad (2)$$

With *AV* being the anthropogenic variable. We adopted a statistical control approach to evaluate the influence of each anthropogenic variable on TAB by controlling for the effect of the model's biophysical (i.e., non-anthropogenic) variables. Human variables can improve model fitting by either having a positive, negative, or nonlinear (e.g., humped) influence on area burned, and several of them were log- or square root-transformed when their distribution were strongly asymmetrical (Table 4-3). We opted for three performance metrics to evaluate the effect of anthropogenic variables on the regression: the deviance explained –the amount of variation in original observations accounted for by a GLM regression– by the model for this particular variable, the gain in deviance explained by the model due to the addition of the anthropogenic variable of interest, and the significance (*p*-value) of this particular variable in the model.

Evaluation of anthropogenic models and importance of variables. We performed a partial dependence analysis to evaluate the relationship between TAB and individual anthropogenic variables using the plotmo package (Milborrow 2012). This method removes the effect of other variables and thus allows for a better understanding of variable of interest's marginal effects.

Spatial effects of anthropogenic variables. We kept the three most valuable anthropogenic models, based on the gain in deviance explained, to produce predictive maps of TAB. We then produced maps showing the difference between biophysical predictions minus anthropogenic predictions to examine the distribution of potential human repercussions in the geographic space. Three sets of maps were created: one showing the original anthropogenic variable, the second showing predictions according to this variable, and a final one showing the difference in predictions (positive or negative). The classification scheme for the predictions map are based on a geometric classification, which is appropriate for continuous skewed data, whereas difference maps have a classification scheme based on standard deviation, which is more appropriate to assess positive and negative variations in model predictions.

In order to spatially assess the effect of the ecological frontier on area burned, we classified our hexels in three classes representing their level of human influence based on the distance to transportation, considered a key spatial marker of the evolution of an ecological frontier. We initially discretized our distance data in three classes –low, intermediate, and high– applying an equal-interval classification scheme to its distribution. The ecological frontier was driven by intermediate distance to transportation. Finally, we computed the residuals from the biophysical model (i.e., observations – models predictions) and we overlaid them on top of human influence to identify potential patterns in under- and over-predictions that may be influenced by the level of anthropogenic imprint on the landscape and could not be captured by biophysical variables only.

4.4 RESULTS

Importance of anthropogenic variables. The biophysical model of TAB gives us a deviance explained of 39.66%. Table 3 shows the effect of individual anthropogenic variable on model explanatory power. The distance to the transportation network, the Human Footprint Index, the density of the energy network, and the distance to logging areas are all highly significant and add to the model's explanatory power by more than 5% (p <0.001), with almost 8% for the distance to the transportation network. The distance to buildings, to mines, and the density of the transportation network are highly significant as well but with a gain around 4% (p <0.001). Percent non-fuel adds more than 4% to the model power but is only moderately significant (p <0.05). The variables accounting for the distance to firefighting bases are of a limited influence on the model's explanatory power. The distance to daily bases add a gain of 1.34% in deviance explained. The density of seismic lines is the variable with the lowest effect on the model, with only 0.28% gain.

Table 4-3**: Generalized Linear Model performance for each anthropogenic variable predicting Total Area Burned. (L) stands for log transform, (S) stands for square root transform. Significance codes: '' 0.001; '**' 0.01; '*' 0.05; '.' 0.1; 'ns' 1.*

Variable	**Biophysical Deviance Explained = 39.66**		
	Deviance Explained (%)	**Gain (%)**	**|p|**
(L) Distance to temporary (daily) fire attack bases	40.70	1.34	*
Distance to primary fire attack bases	40.46	0.80	.
(L) Distance to secondary fire attack bases	40.56	0.90	.
Distance to fire attack bases	40.52	0.86	.
Distance to buildings	43.41	3.75	***
Distance to mines	42.98	3.32	***
Distance to cutblocks	45.13	5.47	***

(S) Distance to transportation network	47.57	7.86	***
(L) Transportation network density	43.76	4.10	***
(L) Energy network density	44.71	5.50	***
(S) Seismic line density	39.94	0.28	**ns**
Population per square kilometer	40.77	1.11	**ns**
(S) Anthropogenic non-fuel	43.80	4.14	*
(S) Human Footprint Index	46.20	6.49	***

The partial dependence plots (Fig.4-2) presents the relationship of TAB to several anthropogenic factors. Distance to logging, buildings, and mining activities shows a pronounced non-linear relationship, with TAB increasing within a distance up to 70 to 150km and then regularly decreasing. Distances to the transportation network display a similar pattern, but within a shorter distance range, with a sharp increase in TAB within the first 20-km. The density of energy and transportation networks shows a sharp decrease in TAB for low-density levels, although it stabilizes around 0.5 m/ha for the density of energy whereas it keeps decreasing for the density of transportation. We also observe a sharp non-linear decrease in TAB for Human Footprint Index values under 10.

The distance to temporary attack bases and the percent of non-fuel in land cover shows a slight non-linear decrease in TAB and then a stabilization in the predicted area burned. A linear decrease in TAB is observed with an increasing distance to primary and secondary attack bases, and the distance to all bases show a slight non-linear relationship, with a decrease in TAB within a 75 km range and then an increase until 150 km. Densities of seismic lines and population are almost unrelated to TAB.

Spatial effects of anthropogenic variables. Figure 4-3 shows the results for the three most important anthropogenic variables. There is a general south to northeast regional pattern, with a gradient going from low predicted TAB in highly developed areas to the highest predictions in remote areas of the Boreal forest, where wildland-urban interfaces are rare or non-existent. The predictions related to transportation and energy networks show a broad band of intermediate-value predictions when transitioning from developed to remote areas, whereas this pattern is less clear for the Human Footprint Index, with a higher variability in the spatial pattern of predicted values of area burned. The residual maps confirm the ability of anthropogenic variables to better predict TAB in areas influenced by humans, as shown by the low residuals values in those locations. Conversely, areas of lower human influence present more variability for the three anthropogenic variables, but also display several clusters of under and over-estimations.

Figure 4-4 shows a regional landscape classification based on the level of human influence, displaying the theoretical ecological frontier in orange. Areas of high human influence, i.e., dense urban areas, mostly present 5000 to 20,000 ha of over-estimations by the biophysical model, underlining the role of human activities in regulating fire activity. The north-east area of low human influence, i.e., the dense homogeneous Boreal forest, generally shows large clusters of underestimated TAB, many over 50,000 ha and more than 150,000 ha in some locations. The most northeastern part of the province shows a cluster of over-estimations where ignitions might be limited. Conversely, the Rocky Mountains at the southwestern edge of the province mostly show model over-predictions, yet these are quite limited in size. Finally, areas of intermediate human influence, defined as the ecological frontier, mostly display low to medium levels of over-estimations, quantitatively limited to ~21,000 ha or less and evenly distributed inside the ecological frontier, where human influence has already spread substantially in the landscape. However, several clusters of important under-predictions appear at the east, extreme north, and within a north-south corridor in central Alberta. Those clusters tend to saddle on the edge of the ecological frontier and the remote areas of the northern Boreal forest. These edges are often characterized by sparse but penetrating transportation and energy features within densely forested areas. Overall, it is ~876,000 ha of underpredicted area burned in the ecological frontier, versus ~449,000ha overpredicted.

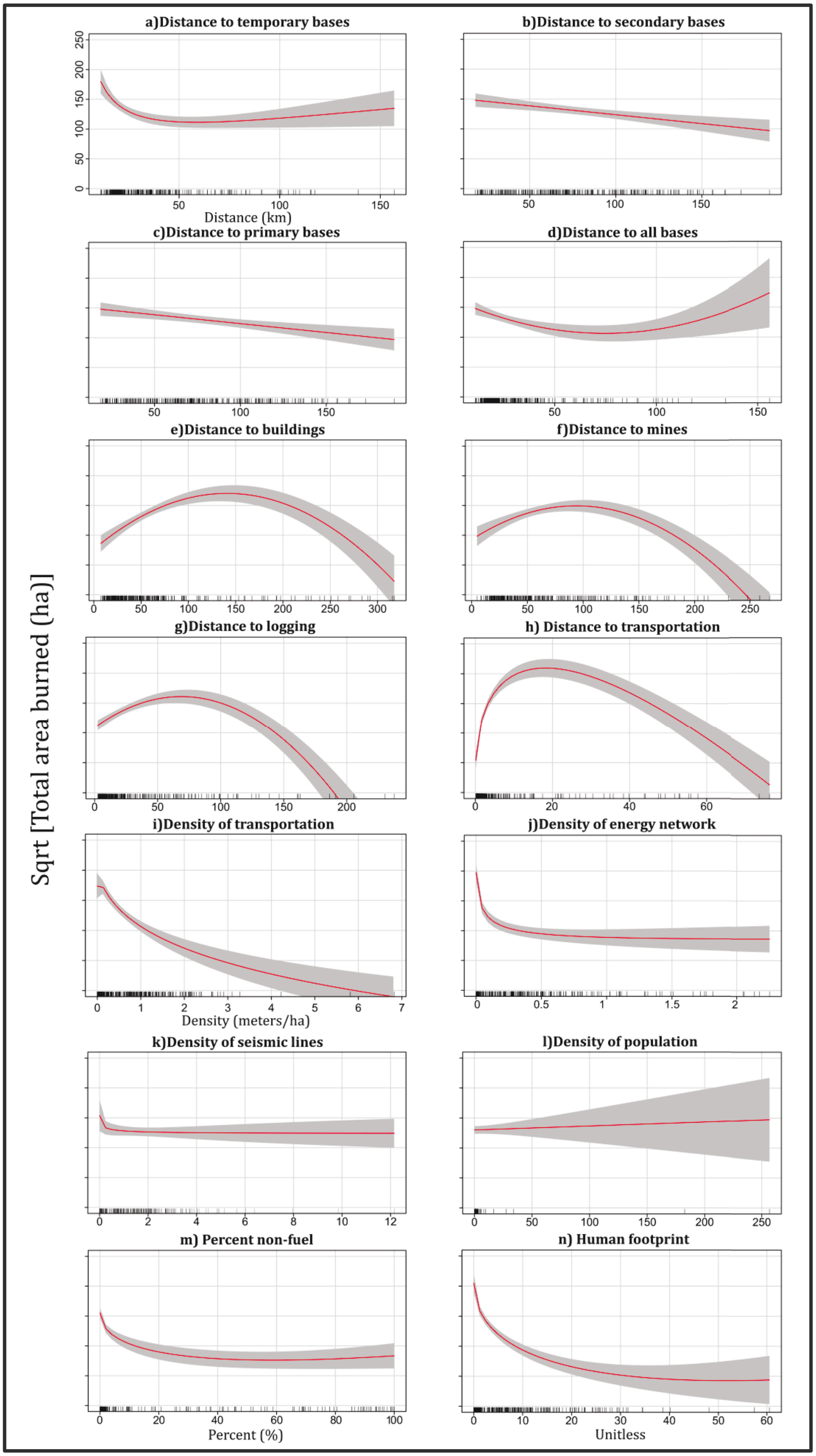

Figure 4-2: *Partial dependence plots showing the response of the total area burned (TAB) to each anthropogenic variable when accounting for the biophysical effect. The rugs correspond to the 283 observations used. Each graph displays a confidence interval of 95%.*

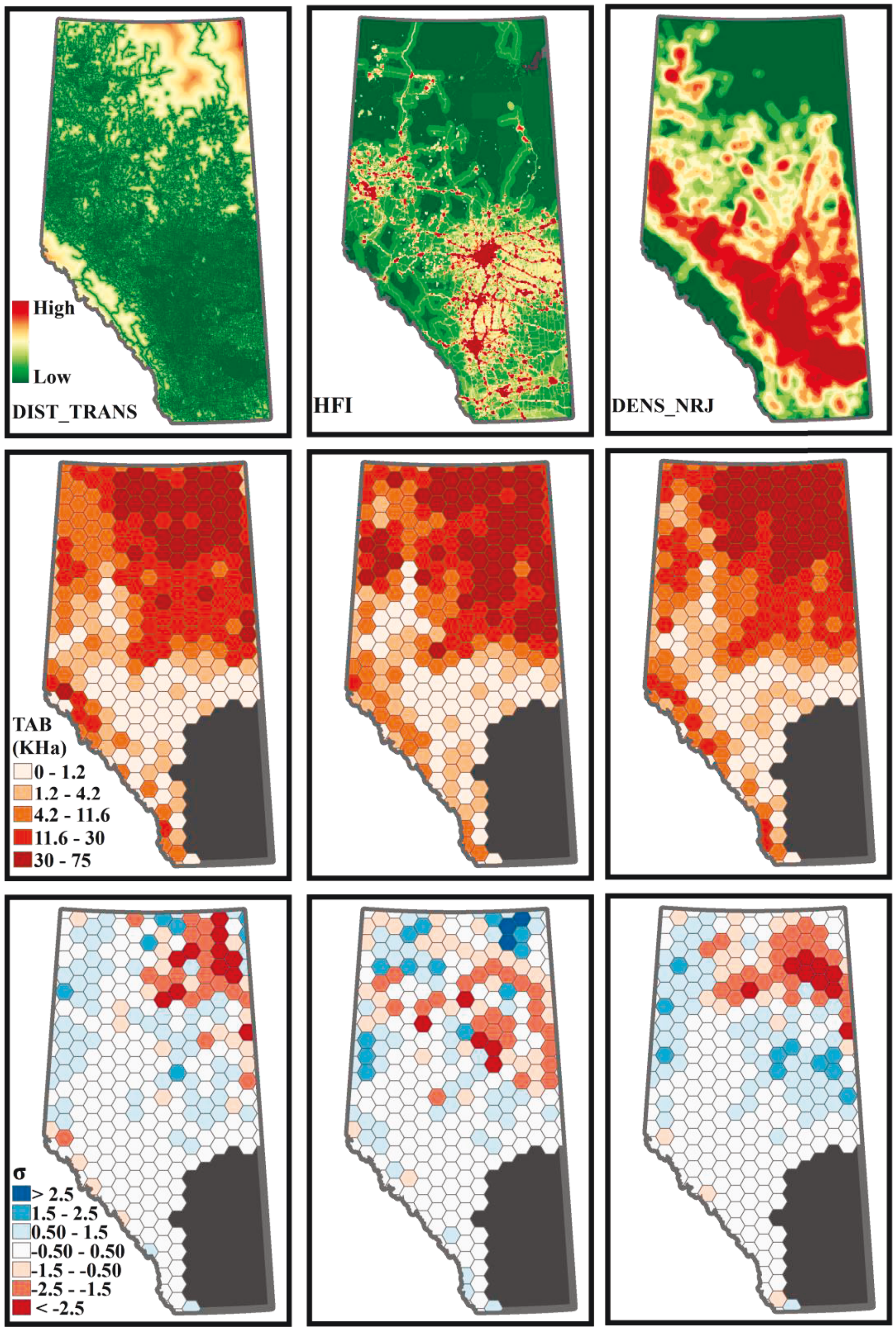

Figure 4-3: *Map sets comparing the spatial effects of the three most important anthropogenic variables: distance to transportation network (DIS_TRANS), Human Footprint Index (HFI), and energy network density (DEN_NRJ). Rows present a) the original variable, b) the prediction in area burned (thousands of hectares), and c) the amount of change compared to the biophysical model (in standard deviation units). Areas in blue are locations where the model under-predicts (positive residuals) area burned when anthropogenic are included in the biophysical model, whereas red areas represent over-predictions (negative residuals), and white areas are locations of minor changes. See Table 4-1 for variable description.*

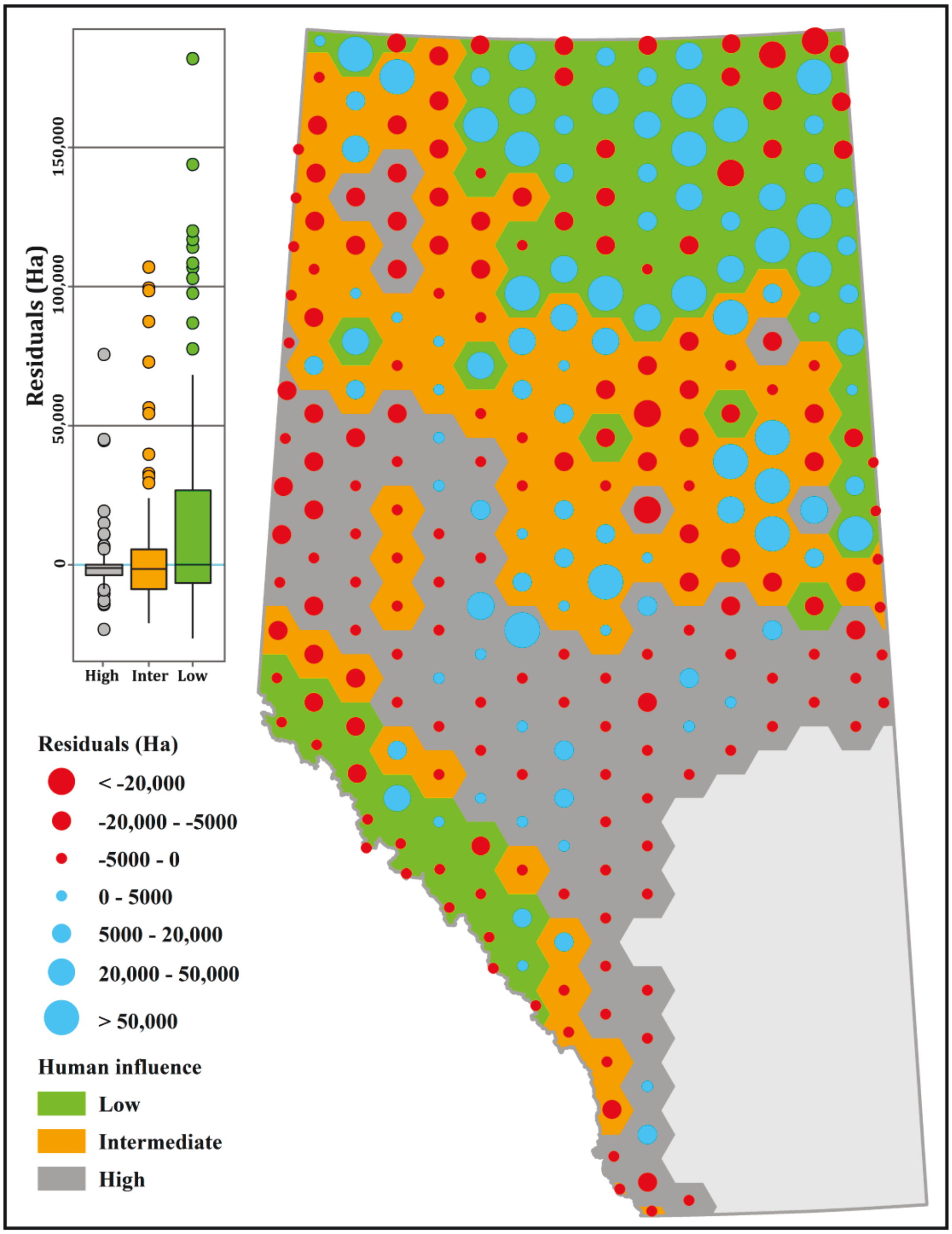

Figure 4-4: *Map of the classified model residuals according to the regional human influence levels, determined as a function of distance to road. The ecological frontier is essentially represented by the orange band extending northward, with a northwest-southeast general orientation. Blue dots represent model under-predictions, whereas red dots represent over-predictions. The boxplot provides an overview of the statistical distribution of residuals according to each human influence level.*

4.5 DISCUSSION

Our study explored how humans influence area burned in the context of wide spread industrial development of the forest of Alberta. We fulfilled our objectives, showing that incorporating large-scale wildland-human interfaces provides a diversity of responses on area burned, rather than the generally accepted decrease when human activity increases. Several anthropogenic factors show non-linear relationships with area burned, supporting the claim that human-fire relationship in North America may be complex and non-stationary (Syphard *et al.* 2007; Parisien *et al.* 2012; Gralewicz *et al.* 2012). Our spatial results also show that the impeding effect of human on fire is the most pronounced in areas with a high concentration of WHIs, which are the zones of contact between natural areas and anthropogenic features, as better detection and defensibility reduce response time and increases suppression effectiveness (Arienti *et al.* 2006). Conversely, the existence of the northern ecological frontier, a result of the ongoing industrial exploration (Héritier 2007, 2009), shows higher than expected area burned due to the combination of low concentration of WHIs with high forest connectivity, offering new insights on natural-to-anthropogenic transition of fire regimes in the Boreal forest.

Many studies focusing on the human influence of wildfire activity use a limited but commonly accepted set of anthropogenic variables. Only a few studies have accounted for an extended set of anthropogenic variables to include area-specific influence of humans in their modeling approach (Cardille and Ventura 2001; Martínez *et al.* 2009; Moreira *et al.* 2011). In line with those studies, we included factors more representative of the industrial context of Alberta, such as logging or seismic lines. The most influential human variables in Alberta are consistent with other studies showing the importance of land use and transportation network in fire activity modeling (Yang *et al.* 2008; Badia *et al.* 2011; Silvestrini and Soares-Filho 2011). Nonetheless, anthropogenic variables commonly used to assess the human influence on area burned, such as distance to roads or population density, were not sufficient for this purpose in Alberta, where the anthropogenic footprint is strongly linked to natural resource exploitation. Although roads and population variables are easily transferrable to different study areas, they may fail at capturing fire-environment specificities that confer an area its fire susceptibility (Miller 2003; Hardy 2005). Similarly, integrative variables such as the Human Footprint Index (Sanderson *et al.* 2002) and population density (Bistinas *et al.* 2013) can show a strong and coherent relationship with area burned, yet they may mask the behavior of the various factors they encompass (e.g. initial attack bases, energy network). Our study shows that many different human impacts can be folded within these variables.

Mining, logging and oil and gas activities occur across virtually all of Alberta outside of protected areas. The non-linear relationships between area burned and the distance to industrial activities is surprising at first, as the increase in area burned with distance countered our expectations. This may be explained by the extensive vegetation management in the vicinity of related infrastructures (e.g. pipelines) and their limiting influence on ignition and spread probability (Alberta Sustainable Resource Development 2008b), although this aspect deserves further investigation. Clearcuts also favor landscape fragmentation (Wulder *et al.* 2008) and we logically observed an increased TAB with distance to logging (Ryu *et al.* 2007; Krawchuk and Cumming 2011). In contrast, the effect of the seismic line network linked to oil-and-gas exploration on areas burned is unclear. Given the extremely high density of these features in parts of Alberta, many fire behavior specialists would expect them to have a non-negligible effect – whether positive or negative – on fire spread, given the greater grass cover in these areas, acting as “flashy” fuels in which fires can ignite and spread rapidly. However, at present, their role on fire activity is not well understood (Arienti *et al.* 2006; Krawchuk, Cumming, *et al.* 2009). The coarse qualitative resolution of the original data may be responsible for this lack of signal, as attributes about line age, maintenance status, and vegetation recovery were limited or non-existent.

Somewhat surprisingly, a higher area burned close to daily attack bases was reported. After closer inspection and discussion with fire managers, this effect could be easily explained: temporary bases that are deliberately located close to large fires—in other words, they track fire activity. In spite of logical relationships between area burned and fire-suppression variables, these had low predictive power; further investigation is thus required to fully understand the long-term consequences of fire suppression in Alberta.

The transportation network seems to be a key factor of change in area burned at the provincial scale, which supports other large-scale findings (Calef *et al.* 2008; Parisien *et al.* 2012; Gralewicz *et al.* 2012; Hawbaker 2013). Moreover, advances in road ecology show that ecological effects of road networks are rather complex (i.e., non-linear and indirect) and occur at multiple-scales through geographic structures identified as "roadsheds" (Lugo and Gucinski 2000). Similar to the watershed concept, a roadshed assumes a specific and consistent distribution of roads, as well as a specific influence on the geographic space it overlaps. The ecological frontier of Alberta may be defined by a characteristic roadshed, made of a low density but penetrating network favoring industrial exploitation, favoring greater area burned in a homogenous forested landscape. This theory is supported by Narayanaraj and Wimberly (2012), who showed that the extension of forest roads necessary for resource exploitation is a vector of higher fire activity when the network remains sparse. Our results also provide quantitative support to the claim that, although human ignitions close to transportation corridors may be numerous (Syphard *et al.* 2007; Gralewicz *et al.* 2012), in Canada the largest fires tend to burn where there is little human access.

Historical changes in fire activity where anthropogenic frontier dynamics are involved have been reported elsewhere in the circumboreal biome (Niklasson and Granström 2000; Achard *et al.* 2008; Liu *et al.* 2012; Clear *et al.* 2014), although these studies do not specifically address the role of industrial development on the area burned. The most common modern example of an ecological frontier resides in the Amazonian rainforest where recent studies show that both ignition rates and area burned increase following a period of human expansion (Cochrane 2003; Fearnside 2005; Morton *et al.* 2008), which is in line with our findings. Our results displaying increases in area burned at the edge of the ecological frontier, yet spatially clustered, imply the existence of a spatial gradient in the anthropogenic alteration of the fire regime of Alberta. In this regard, the ecological frontier must be seen as a transitional – and typically short-lasting – area favoring landscape fire susceptibility in its initial phase of establishment, when human-caused and natural-caused ignitions overlap. We believe this effect is spatially represented by lower-than-predicted values at the edge of the ecological frontier, although this pattern may be enforced by spatial discrepancies in observed area burned between different levels of human influence, as well as the relative simplicity of our model and the limited number of biophysical variables. Then, fire likelihood would slowly decreases as human pressure increases, as shown by low levels of overpredictions spreading across the ecological frontier, which is the general observed trend (Guyette *et al.* 2002; Moreira *et al.* 2010). We argue the first stage of this dynamic would combine a higher ignition rate (i.e., human + natural) with high landscape connectivity, thereby favoring both fire initiation and spread (Haight *et al.* 2004; Lacroix *et al.* 2006; Moreira *et al.* 2010). Although applied to the agricultural context, Weir and Johnson (1998) showed that escaped fires in the early 1900s could widely spread northward of the settlement frontier, where the Boreal forest cover was homogeneous.

While our approach allowed us to meet our objectives, the datasets used for modeling add uncertainty to this study's results. Changes in fire-suppression policies, fire management inside and outside protected areas, firefighting techniques, and variations in the year-to-year fire season severity can induce variability in fire data that may be masked over a 31-year period. That being said, earlier findings suggest that the consequences of such changes would mainly be noticeable when working with a 10-year time scale (Cumming 2005). With the last major firefighting policy change in Alberta being in 1983 (i.e., so at the beginning of our period of interest), we believe that three decades of observation is an acceptable compromise: it smooths the impact of temporal changes while preserving the main information regarding fire activity (Syphard *et al.* 2007). Many of the anthropogenic variables may also have varied within the study's aggregated 30-year period. The inclusion of temporal variables accounting for human expansion, the use of spatio-temporal models, or working at a different temporal scale (e.g. decadal) could reduce uncertainties and strengthen the approach; however, at present finding annually-resolved data representing human influence remains a critical difficulty in fire modeling in Alberta and elsewhere (Turner 1990; Hardy 2005; Syphard *et al.* 2007; Thompson and Calkin 2011). Moreover, several datasets, such as the Human Footprint Index or the land cover derived from satellite imagery, do not include information for the end of our study period. The population density only covers 10 years of the study period. In a region of dynamic and large scale development, these data gaps might impact the accuracy of our results, an issue that could be partially fixed with more integrative data such as the regional Human Threat Index created by Global Forest Watch (Lee *et al.* 2009, see appendix 3).

Area burned remains a very important metric of fire activity for the Boreal forest, and consequently an important issue for large-scale fire modeling in North America (Littell *et al.* 2009; Parisien and Parks 2014; Parks *et al.* 2015). Nonetheless, natural and anthropogenic fire activity show different characteristics (Guyette *et al.* 2002), and area burned differs by ignition cause. Not accounting for it is a limitation of our approach, although explained by an issue of data sparseness, as there were few human-ignited fires compared to naturally-ignited fires for our period of interest. However, using a GLM encompassing all ignition causes, along with climate variables, has been shown to be efficient for large scale area burned assessment (Littell *et al.* 2009). In addition, most of the anthropogenic variables will affect lightning-caused fires as much as they may affect human-caused fires.

Under the current policy of fire monitoring and fire suppression, managing the ecological frontier in Alberta requires special attention and could warrant the creation of a

HWI classification to identify specific natural-anthropogenic feature configurations. For instance, which configuration of the road network combined with forest harvesting might favor fire ignition and spread? Some studies in the Mediterranean area, where land managers have long-acknowledged the human dimension of fire regimes, could provide guidance in this respect (Lampin-Maillet *et al.* 2010; Chas-Amil *et al.* 2013) and help mitigate the negative impacts of the anthropization of our Boreal wildlands after some adaptation to the geographic context. To this goal, our results exploring the consequences of natural resource exploitation should be strengthened by complementary studies addressing local effects.

Our findings provide useful insights in the understanding of fire activity in future ecological frontiers that will undoubtedly appear as part of the projected industrial development of the Boreal forest during the 21st century. The magnitude and speed at which humans are expanding their range in Alberta, and hence altering natural fire regimes, may occur in other parts of Boreal North America that are on the verge of similar magnitude changes according to planned economic development and model simulations (Schneider *et al.* 2003). Our results outline the importance of incorporating human influence in future fire predictions in areas where rapid increases in anthropogenic activity are expected, as the response of fire regimes may not be monotonic and therefore impossible to extrapolate from other geographic areas (Parisien *et al.* 2016). We also provide insights to explain how natural fire regimes may shift to anthropogenic fire regimes, adding more complexity in the human-fire relationship, a complexity which is likely to increase with expected climate change effects on wildland fire activity in Alberta (Tymstra *et al.* 2007).

4.6 ACKNOWLEDGEMENTS

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CHAPTER 5 - UPDATING A GLOBAL WATER RISK INDEX USING CURRENT AND FUTURE FIRE ACTIVITY DATA

5.1 ABSTRACT

The fate of global freshwater resources is currently at the centre of world governance discussions. Based on the findings from many studies, the pressure on freshwater resources is predicted to grow sharply because of climate change, as well as population growth and the increase of wealth. However, emerging risks triggered by the global environmental change are getting increasingly more attention, adding new pressures to water resources. In this respect, the apparent increase in wildfire occurrence around the world has raised concerns. Recent large and severe fire events that happened in water basins supplying drinking water to downstream communities have fed worries as to the reliability of the water supply for human consumption and overall watershed health. As those events seem to become less isolated, it appears important that wildland fire hazards become included in water risk assessments. The present study relies on a public water risk assessment database, Aqueduct, to update and compare changes in global water risk levels when indicators of current and future fire activity are combined. My results show that a "fire-enhanced" index displays a different pattern compared to the original one, especially in areas supposedly preserved from considerable water pressures. However, future changes in the pattern of water risks remain limited, which implies that potential future threats from post-fire hydrogeomorphic hazards might be offset by population controls on fire activity and water consumption.

5.2 INTRODUCTION

Global environmental change currently occupies the forefront of many scientific research efforts working towards the long-term sustainability of anthropogenic development (Foley *et al.* 2005; Rockström *et al.* 2009; Verburg *et al.* 2013; Costanza *et al.* 2014).

Among the multitude of potential impacts on coupled human-and-natural systems that have been advanced in the literature, the consequences of global change on freshwater resources have raised particular concerns. Documented increases in the occurrence of climate anomalies have intensified the hydrological cycle (Huntington 2006), causing greater variability and intensity of hydroclimatic events and thus more extreme and unanticipated hazards such as floods (Hirabayashi *et al.* 2013) and droughts (Dai 2012). In parallel, worldwide changes in land use and land cover due to anthropogenic activities have altered the natural function of the planetary water system (Sterling *et al.* 2012). Such dynamics invariably interact with climate change (Murray *et al.* 2012) and further expose populations to extreme hydroclimatic hazards and water pollution (Schwarzenbach *et al.* 2010; Kundzewicz and Matczak 2015). These ongoing alterations in the functioning of the global hydrological system pose serious questions as to the long-term security of the water supply (Hope and Rouse 2013), along with the potential emergence of new threats to freshwater resources (Myers and Patz 2009).

As evidenced by a recent scientific focus, wildland fires represent a growing danger, as several fire events that have happened in the past decade have unveiled concerns as to the protection of the water supply (e.g., 2011 New Mexico Las Conchas Fire, 2013 California Rim Fire, 2016 Alberta Horse Creek Fire, 2017 Chile fires). Although wildland fires play an important part in the functioning of the global water budget (Li and Lawrence 2016), the aftermath of a massive and severe blaze can profoundly disturb local and regional hydrological systems (Shakesby and Doerr 2006). Indeed, the absence of vegetation coupled with changes in soil structure may potentially lead to shorter concentration time and higher peak flows during rainstorms (Moody and Martin 2001a), higher flows during the dry season (Kinoshita and Hogue 2011), and an annual water yield that may eventually be higher in burned watersheds (Hallema *et al.* 2016, 2017). A likely larger runoff may also favor erosion (Mayor *et al.* 2007), leading to greater loads of sediments and ashes and thus increased turbidity (Townsend and Douglas 2000; Silins *et al.* 2009), higher concentration of dissolved chemical compounds such as phosphorus, organic carbon, or nitrogen (McEachern *et al.* 2001; Bladon *et al.* 2008; Brown *et al.* 2015), and accumulation of woody debris (Bendix and Cowell 2010). These post-fire hydrological effects, given the proper post-fire environmental conditions, can potentially last up to several decades (Kuczera 1987; Meixner and Wohlgemuth 2003) and reach downstream locations as far as 50km (Dahm *et al.* 2015). As global change studies have revealed an increasing trend in wildfire activity (Flannigan *et al.* 2009; Flannigan *et al.* 2013; Jolly *et al.* 2015), concerns regarding post-fire effects on the water supply of populations have been logically growing (Smith *et al.* 2011; Emelko and Sham 2014; Kinoshita *et al.* 2016). As evident interactions amongst water resources, socio-economic stability, and wildfires have emerged (Martin 2016), there is an urgent need for reflection about the future consequences of wildland fires on water security (Bladon *et al.* 2014).

Most studies of post-fire risks to water resources have been done at a rather small spatial scale (e.g., from watershed to large river basin) (Moody and Martin 2004; Smith *et al.* 2011), though further efforts to recognize wildfires as a global source of water-related risks remain necessary. However, the complexity of post-fire hydrological phenomenon would be difficult to model at a global scale; a higher level of abstraction is therefore appropriate. Spatially explicit indexes have been widely used to address a diversity of questions related to global environmental issues, especially water security (Vörösmarty *et al.* 2010b; Ceola *et al.* 2015; Green *et al.* 2015; Veldkamp *et al.* 2016). Such indexes are useful decision-support tools, as they aggregate multiple sources of relevant variables into a final result relatively easy to understand and to reproduce, making them an invaluable support for benchmarking policy implementations in a simple, yet efficient, way (OECD 2008). In parallel, the fast development of web-GIS technologies now offers governmental agencies and research entities the capability to massively share scientific knowledge (Haklay *et al.* 2008). The multiplication of online geographic data platforms accessible to the public, such as the Global Forest Watch (World Resource Institute 2014) or the Global Flood Awareness System (Alfieri *et al.* 2013), represents an effective vector for spreading rich spatial content on current environmental issues and raise awareness of the future of the Earth system. The Aqueduct 2.1 water risk index, available through a web-GIS platform maintained by the World Resource Institute (WRI) (Reig *et al.* 2013; Gassert *et al.* 2014) has attracted attention, including in the scientific community (Bierkens 2015; Gleick 2015; Mueller *et al.* 2015; Quinteiro *et al.* 2017). The open access geodatabase provides information on water quantity and quality constraints for more than 20,000 water basins worldwide. As the indexation method is reproducible and the result provided has been scientifically endorsed, the Aqueduct water risk index represents a sound basis for the integration of wildfire information and the assessment of fire effects on the global water risk.

The study presented hereafter has two main objectives: 1) updating the Aqueduct database with current and future global wildland fire activity information, and 2) revealing change patterns in the levels of risk across the globe due to current and future fire activity. I hypothesized that the inclusion of wildland fire information as a proxy to the occurrence of post-fire hydrogeomorphic hazards should increase water risk levels in a majority of hydrologic basins for both periods. As global environmental change is likely to decrease the reliability of freshwater resources (Vörösmarty *et al.* 2007; Van Vliet *et al.* 2013; Arnell and Lloyd-Hughes 2014), I believe this approach points at the necessity to systematically integrate wildfire data in the global assessment of risks to the water supply.

5.3 DATA AND METHODS

To test my hypothesis, I created two indicators of wildland fire activity known to affect water quantity and water quantity. I then calculated two “fire-enhanced” water risk indexes (i.e. current and future periods) updated with my fire hazard indicators, following the methodology published in the WRI. Finally, I computed the difference between indexes to reveal the degree of change in the spatial pattern of the global water risk index.

5.3.1 DATA

I used the Aqueduct 2.1 Dataset version 2015, downloadable free of charge from the eponymous WRI web-GIS application (Tab.5-1). The creation of the database was initially motivated by the necessity of several international companies to spatialize their potential exposure to water-related risks as a function of their industrial sector (e.g., oil & gas, micro-conductors) (CDP 2016), although the WRI has also promoted the use of Aqueduct for multi-sectoral purposes. The data offered water-related risk information for 21,688 hydrological basins covering the global landmass. Twelve indicators provided proxies to several factors known to influence the sustainability of freshwater resources. Seven indicators, such as seasonal flow variability or the presence of upstream reservoir storage, pertained to the water quantity risk theme. Two indicators, the baseflow return rate and the amount of upstream protected lands, pertained to the water quality risk theme; the former measures the ratio of available water that has already been used upstream, whereas the latter accounts for the importance of undisturbed natural areas in the provision of hydrological services. Finally, three indicators, such as the amount of media coverage and the general access to water, pertained to the reputational risk theme. Those 12 indicators were aggregated using a weighted mean (Gassert *et al.* 2014). More information on the creation of the original indicators as used in this study is available in the WRI Working Papers for the Aqueduct project (Gassert *et al.* 2013, 2014; Reig *et al.* 2013; Luck *et al.* 2015).

Noticeable changes in water flow regimes of hydrological basins have been linked to the upstream area affected by disturbances (Wine and Cadol 2016). I, therefore, used the area burned (AB), expressed as the fraction of a pixel, to approximate potential increases in annual water yield and the potential for higher peak flows. I relied on current (1997-2005) and future (2091-2100) area burned data modelled at a 1.875°×1.875° by the SPITFIRE global mechanistic fire model (Thonicke *et al.* 2010) and constrained by the CMIP5 representative concentration pathway(RCP) 8.5 (Kloster and Lasslop 2017). As part of the Max Plank Institute Earth System Model, fire simulations also took into account future changes in fire occurrence based on projected population density as a control of area burned (Bistinas *et al.* 2013; Lasslop *et al.* 2014). Although the model tended to underestimate current patterns of area burned when compared to the Global Fire Emission Database (GFED) (Giglio *et al.* 2013), it allowed me to conserve homogeneity in the source of data for present and future area burned indicators.

Fire severity (FS) is one of the primary controls of post-fire impacts on water quality, as it triggers large release of chemical compounds and sediments due to vegetation combustion and changes in soil protective coverage (González-Pérez *et al.* 2004; Hosseini *et al.* 2016). The Build-Up Index (BUI) is a weather-based index used as a component of the Fire Weather Index (FWI) (Van Wagner 1987) that provides an approximation of the amount of fuel available for consumption and thus a proxy to fire severity. However, the BUI does not explicitly consider fuel characteristics (e.g., vegetation type or structure). Therefore, values in dry and hot areas (e.g., Sahara, Namib) reach the highest fire danger scores, as weather conditions are extremely fire-conducive, but the absence of vegetation prevents fire from occuring (Pausas and Ribeiro 2013). I used Net Primary Productivity (NPP) data to explicitly account for fuel availability by adjusting (i.e. multiplying) them according to BUI values; low-to-null NPP values in desertic areas were able to give more realistic FS values than BUI alone. BUI data came from a global FWI database (Flannigan *et al.* 2013) developed for current (1995-2010) and future (2091-2100) periods, the latter being based on the SRES 2 carbon emission scenario A2 (Ipcc 2000) as simulated by the

Hadley Centre global circulation model. It comes as a 2.5°×2.5° resolution point mesh. NPP data came from a long-term NASA MODIS archive spanning 2000 to 2010 (Zhao and Running 2010) at a 1km×1km pixel resolution. However, no future projections were available; therefore, NPP values were held constant.

I refer hereafter to AB and FS as fire hazard indicators, not fire risk, as they do not include any notion of asset vulnerability and resilience.

Table 5-1: Table 1: Details about the data used in this study.

Dataset	**Unit**	**Date**	**Source**
Aqueduct 2.1	Multiple	2015	World Resource Institute
Area Burned	%/pixel	Present: 1997-2005 Future: 2090-2100	Max Plank Institute for Meteorology
BUI	Unitless	Present: 1995-2010 Future: 2090-2100	Canadian Forest Service
Net Primary Productivity	g/cm²/year	2000-2014	NASA – Numerical Terradynamic Group, University of Montana

5.3.2 METHOD

All data were projected to the Winkel Tripel coordinate system, which is the best compromise between the preservation of shape and area at a global scale, so proportions are closer to reality (Goldberg and Gott 2006). Raster data were reprocessed to reach a 60km pixel resolution, equivalent to a 0.5×0.5° resolution at the equator in the WGS84 geographic coordinate system. AB and BUI data were first converted to points and then interpolated using a spline with tension method in ArcGIS 10.1 (Environmental Systems Research Institute 2012), also referred to as a thin plate method and commonly apply to FWI index calculations (Price *et al.* 2000). The same method was applied to both current and future period data. The shape of the distribution for the reprocessed data remained equivalent to the original one, although several output values were negative due to the interpolation process and were reclassified as 0. The final FS raster layer was obtained by multiplying BUI by NPP.

Before the integration of the fire indicators into Aqueduct, I improved the database geometry to remove topological errors. I first converted all water basins to single polygon entities, resulting in a change from ~25,000 polygons to 75,000. I did so to remove all polygons less than 100km², as they mostly were artefacts inherited from the intersection of different data sources to create the original dataset (see (Gassert *et al.* 2014). After removal of those "micro" water basins, this intermediate dataset contained ~50,000 polygons. For each of them, I extracted the mean pixel value of the fire hazard indicators. When a water basin was too small to compute an average (i.e., only covered by one pixel), I directly extracted the indicator's value for this polygon. Finally, I dissolved this modified Aqueduct layer to recreate a polygon database made of 20186 entities, Greenland excluded.

Following the original WRI methodology (Reig *et al.* 2013), fire hazard indicators were standardized between 0 and 5 in order to assign them a hazard level. 0 stands for an absence of hazard; 1 low; 2 low to medium; 3 medium to high; 4 high; 5 very high (Tab.5-2). To my knowledge, no global-scale classification of fire hazard either based on area burned or fire severity is currently available. I therefore applied a classification scheme resulting in a pattern representative of the burned area fraction and the biomass consumption displayed in the Global Fire Emission Database (GFED) 4 for the current period. Noteworthy the fact that it is impossible to reproduce the exact same pattern due to inherent differences in data creation processes and spatial resolution between the raster data sources, as AB and BUI come from a modelling effort, whereas GFED is based on direct satellite observations (Giglio *et al.* 2013; Lasslop *et al.* 2014). Following the original WRI methodology (Reig *et al.* 2013), fire hazard indicators were classified into six levels: 0 (no hazard); 1 (low); 2 (low to medium); 3 (medium to high); 4 (high); 5 (very high) (Tab.5-2). I then applied the same class ranges to the indicators of future fire hazard.

Table 5-2: Hazard classification applied to fire indicators' values.

Hazard class	Fraction area burned (%) (1997-2005)	Fire severity
0:0.99 – No hazard	< 0.08	< 0.75
1:1.99 – Low	>= 0.08 - < 1	>= 0.75 - < 3
2:2.99 – Low to medium	>= 1 - < 2.5	>= 3 - < 15
3:3.99 – Medium to high	>= 2.5 - < 15	>= 15 - < 35
4:4.99 – High	>= 15 - < 30	>= 35 - < 100
>= 5 – Very high	> 30	> 100 - 1330

Scaling the indicators values to the hazard classes was carried out using a continuous function based on the following formula:

$$f(x) = \frac{(b-a)(x-min)}{max-min} + a$$

With b and a being the upper and lower bounds of the class, respectively (e.g., 1.99 and 1 for the risk class 1), x is any value falling into this range, max and min being the highest and lowest values of the original indicator for this class, respectively. This function was different from the one originally used by the WRI (Reig *et al.* 2013), but it was more adapted to the range and the distribution of my data.

Once the fire indicators were assigned a hazard class value from 0 to 5, I joined them to the first 12 water-related risk indicators available in Aqueduct to obtain only one dataset. The original Aqueduct index was the result of a linear aggregation by weighted mean, in which all indicators were assigned a default weight according to their importance regarding water supply security as defined by a group of water experts (Gassert *et al.* 2013). I reproduced this scheme for the initial water indicators, and I assigned the maximum weight, or descriptor of importance, to my fire indicators (Tab.5-3). This choice was driven by the central aim of this study to emphasize the potential impact of fire activity on water resources. The same aggregation method was carried out using the projected fire hazard indicators while keeping the water indicators constant. The resulting indexes for the present and the future were mapped to illustrate the difference between the current water risk with and without fire indicators, then between current and future "fire-enhanced" water risk indexes.

Table 5-3: Descriptor of importance applied to the "fire-enhanced" Aqueduct water risk index. For more information on the indicators used in this study, see the WRI working papers by Gassert *et al.*(2013,2014).

Water-risk Indicator	Risk theme	Default weight
Baseline water stress	Water quantity	× 4
Inter-annual variability		× 1
Seasonal variability		× 0.5
Flood occurrence		× 1
Drought severity		× 1
Upstream storage		× 2
Groundwater stress		× 2
Area burned		**× 4**
Return flow ratio	Water quality	× 1
Upstream protected land		× 0.5
Fire severity		**× 4**
Media coverage	Water regulation	× 1
Threatened amphibians		× 0.5
Access to water		× 2

5.4 RESULTS

Figure 5-1 shows the AB fire indicator per water basin for the current period (1997-2005). 4984 basins are in low hazard, 4966 low to medium, 2341 medium to high, 5221 high, and 2676 are classified in very high hazard. For the 2091-2100 period based on the

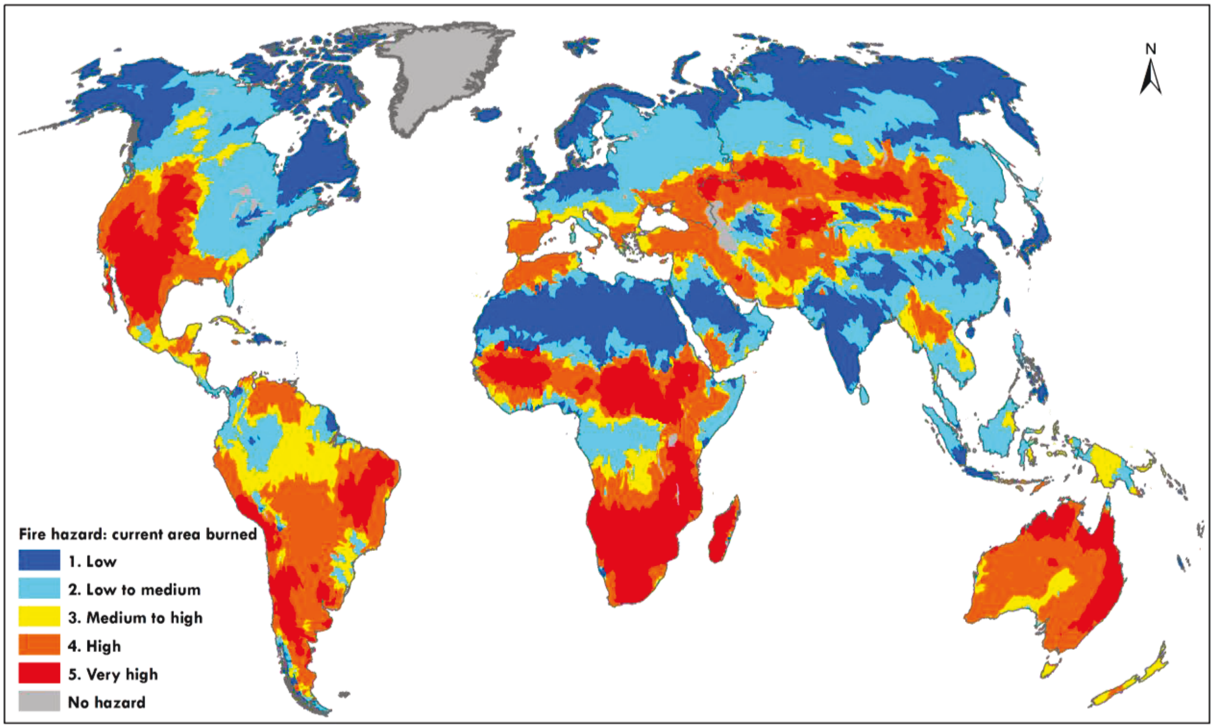

Figure 5-1: *Area burned hazard indicator for present days (1997-2005) according to the water basins defined by the WRI.*

RCP 8.5 scenario (Fig.5-2), 4135 basins are classified as low hazard, 6297 as low to medium, 2514 as medium to high, 4756 as high, and 2486 as very high. Low and low-to-medium hazard categories cover greater areas, especially in the North hemisphere, whereas the extent of high and very high categories tends to diminish. Figure 5-3 shows the FS fire indicator for the present days (1971-2000), with 2861 water basins classified as low hazard, 2825 as low-to-medium, 6600 as medium-to-high, 4236 as high, and 3666 as very high. Figure 5-4 shows the fire severity hazard for the future period 2091-2100, with 2561 basins in low hazard, 1936 in low-to-medium, 5077 in medium to high, 4648 in high, and 5966 classified in very high hazard. Although medium-to-high hazard tends to extend northward, it is worth notifying the expansion of high and very high hazard, specifically in Amazonia, Western North America, and Western and Central Eurasia.

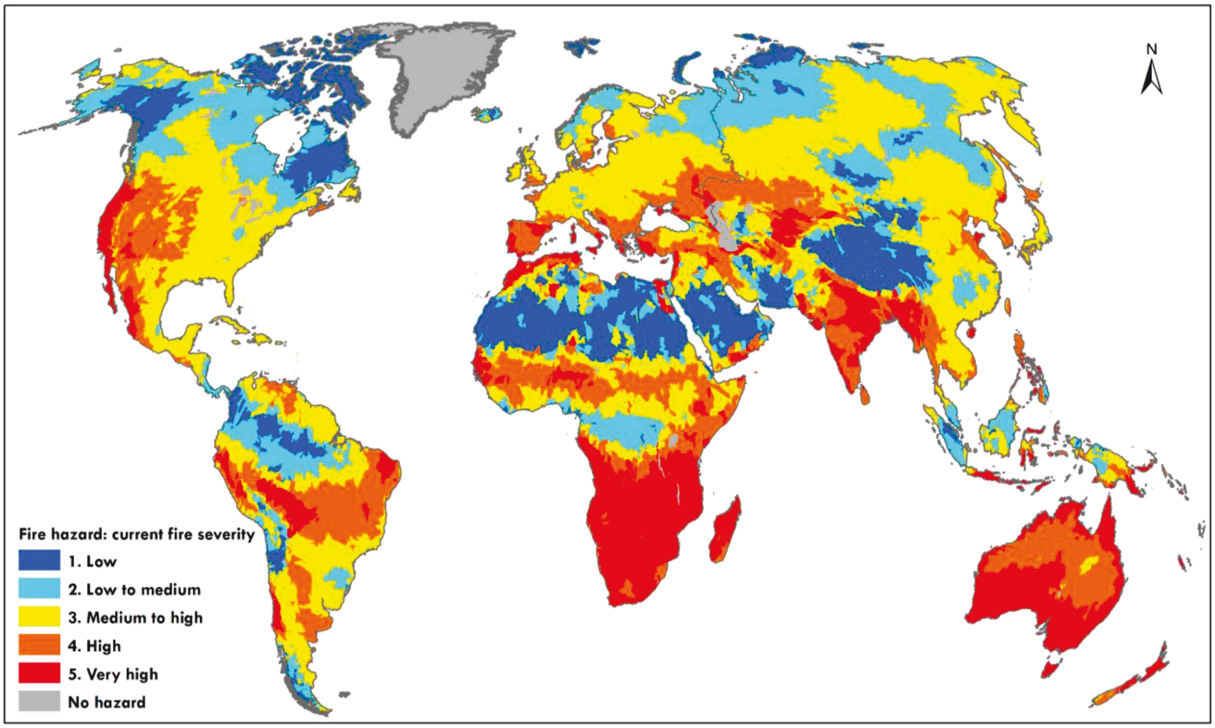

Figure 5-2: *Fire severity hazard indicator for the current period (1995-2010), according to water basins boundaries defined by the WRI.*

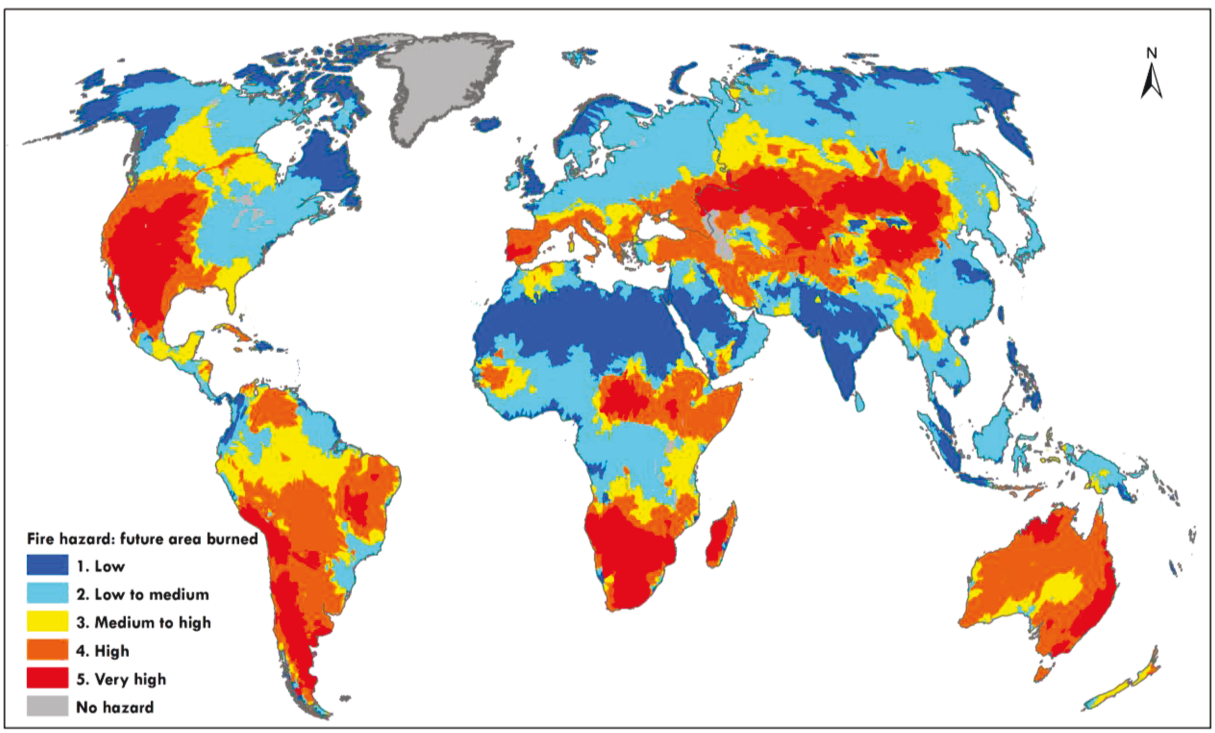

Figure 5-3: *Area burned hazard indicator for the period 2091-2100, according to the water basins defined by the WRI.*

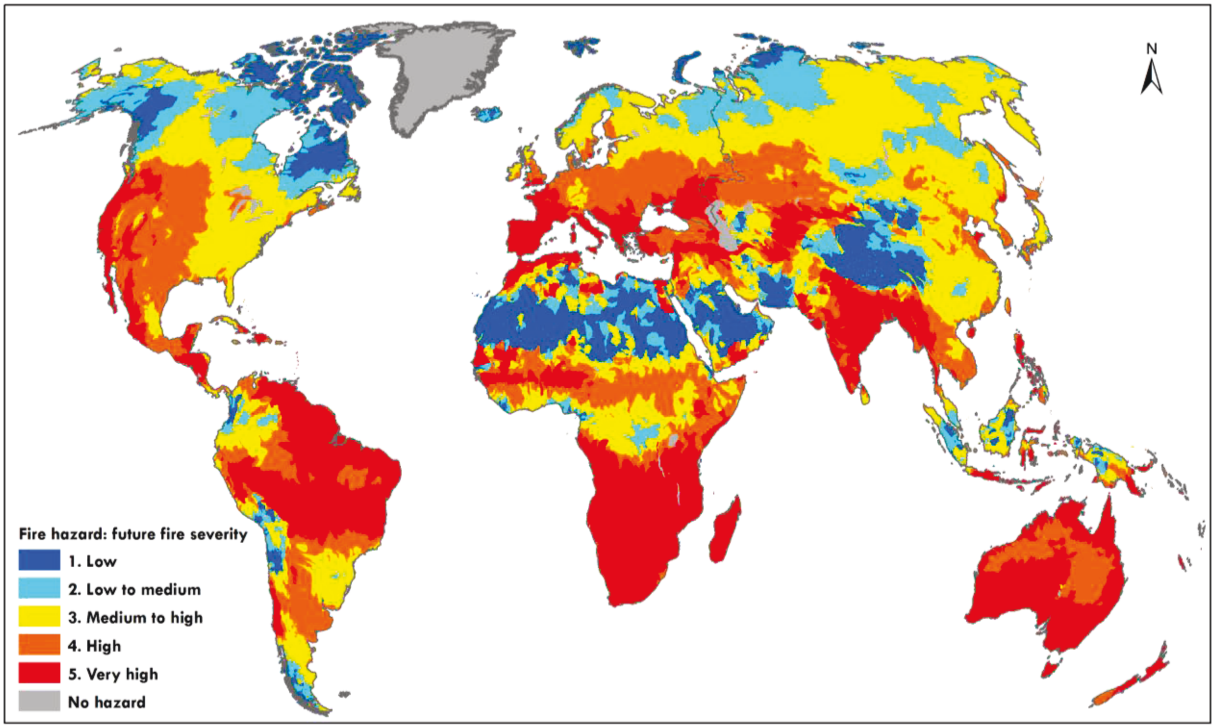

Figure 5-4: *Fire severity hazard indicator for the future period 2091-2100, according to water basins boundaries defined by the WRI.*

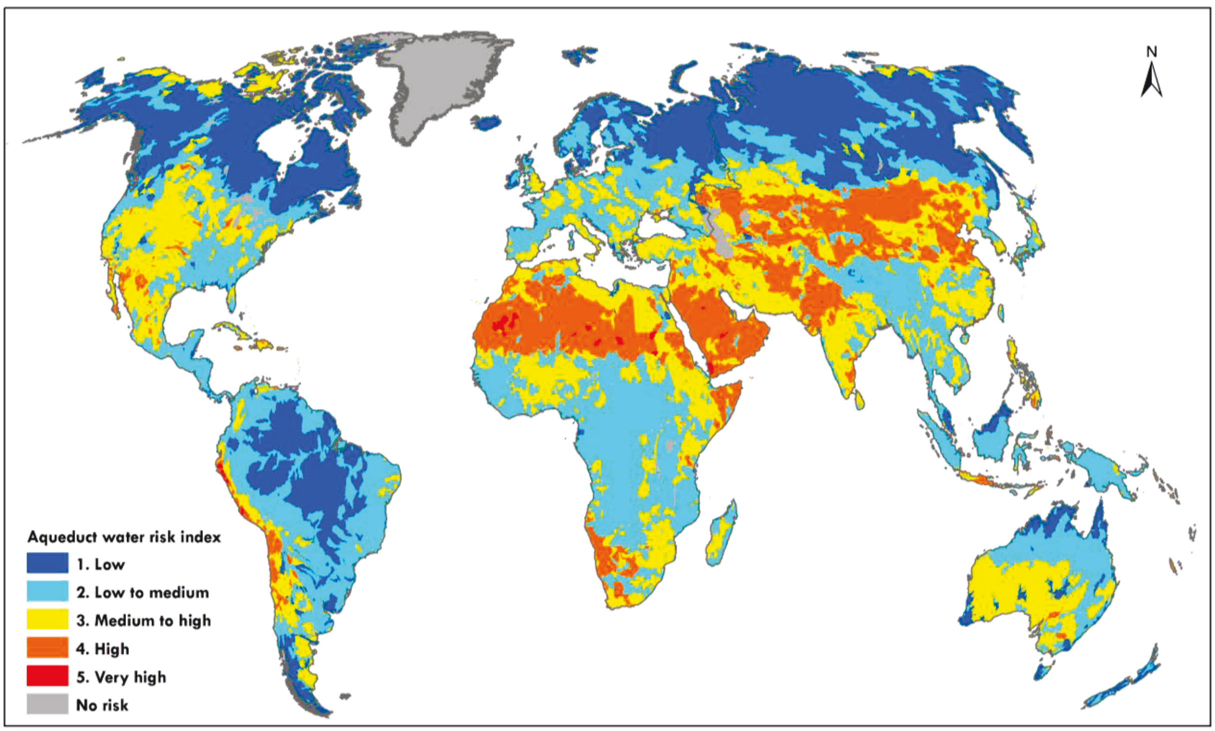

Figure 5-5: *Original Aqueduct water risk index*

Figures 5-5, 5-6, and 5-7 show the primary Aqueduct water risk index with 12 water risk indicators, the “fire-enhanced” index that includes AB and FS fire hazard indicators, and the difference in the final risk classification between the both versions of the index, respectively. 857 basins, mainly located in Amazonia and Australia, experience an increase in two levels of risk, 6625 a one-level risk increase, 9058 remain unchanged, 3143 decrease by one risk level, 494 by two risk levels, and nine decrease by 3 levels of risk. Most of the areas where the risk decreases are located in dry areas, where environmental constraints prevent the growth of dense vegetation limiting fuel availability and thus the occurrence of severe fires (Pan *et al.* 2013; Pausas and Ribeiro 2013). That being said, this is the same environmental constraints that put dry locations at high to very high water-related risk (Hoekstra *et al.* 2012).

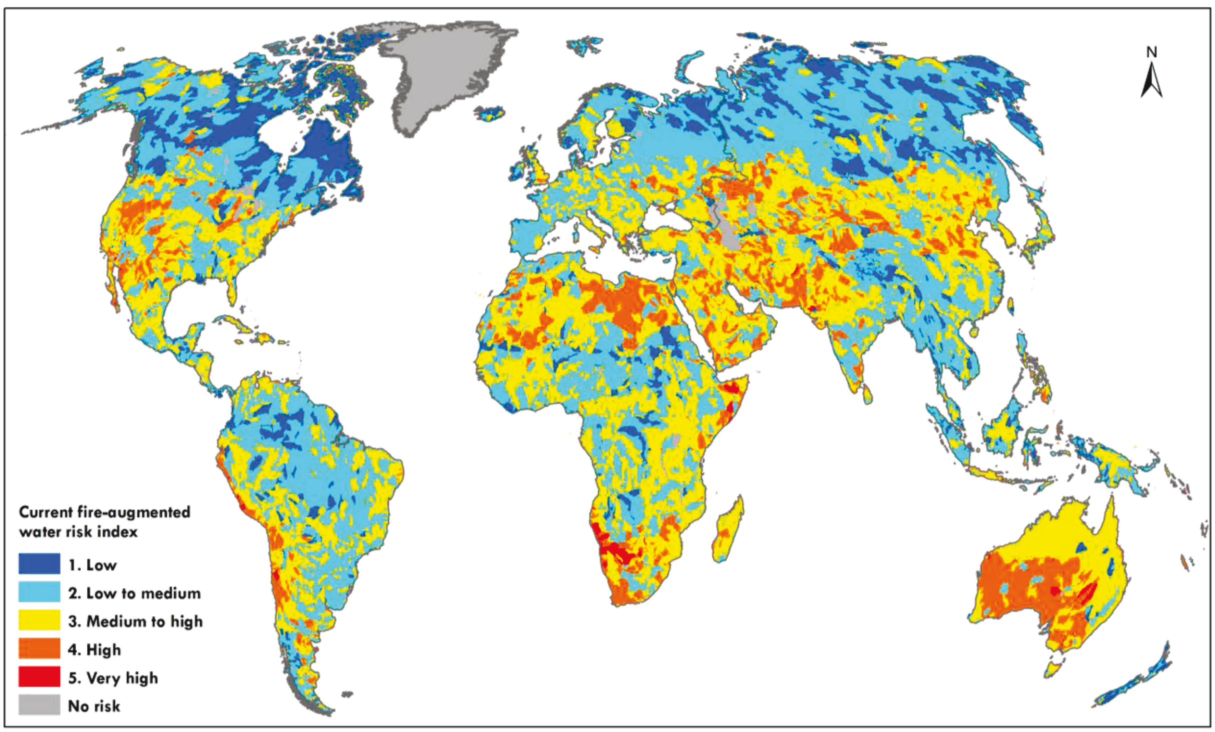

Figure 5-6: *“Fire-enhanced” Aqueduct water risk index for the current period*

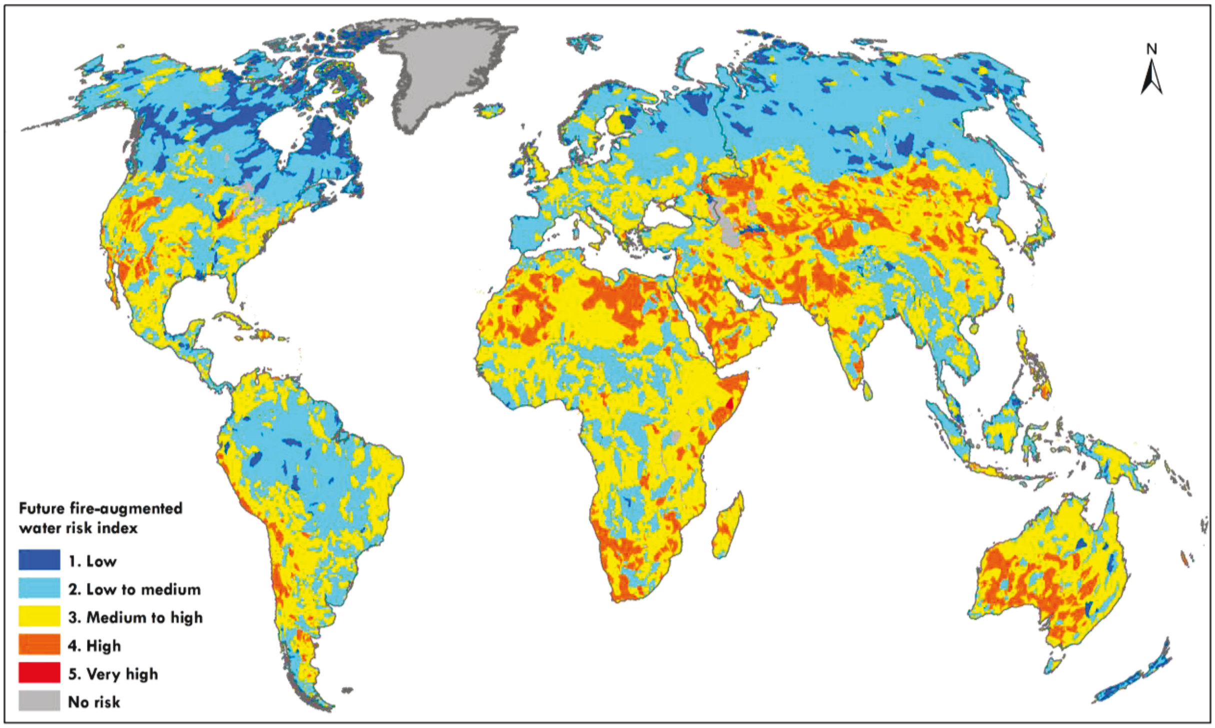

Figure 5-7: *Future "Fire-enhanced" water risk index*

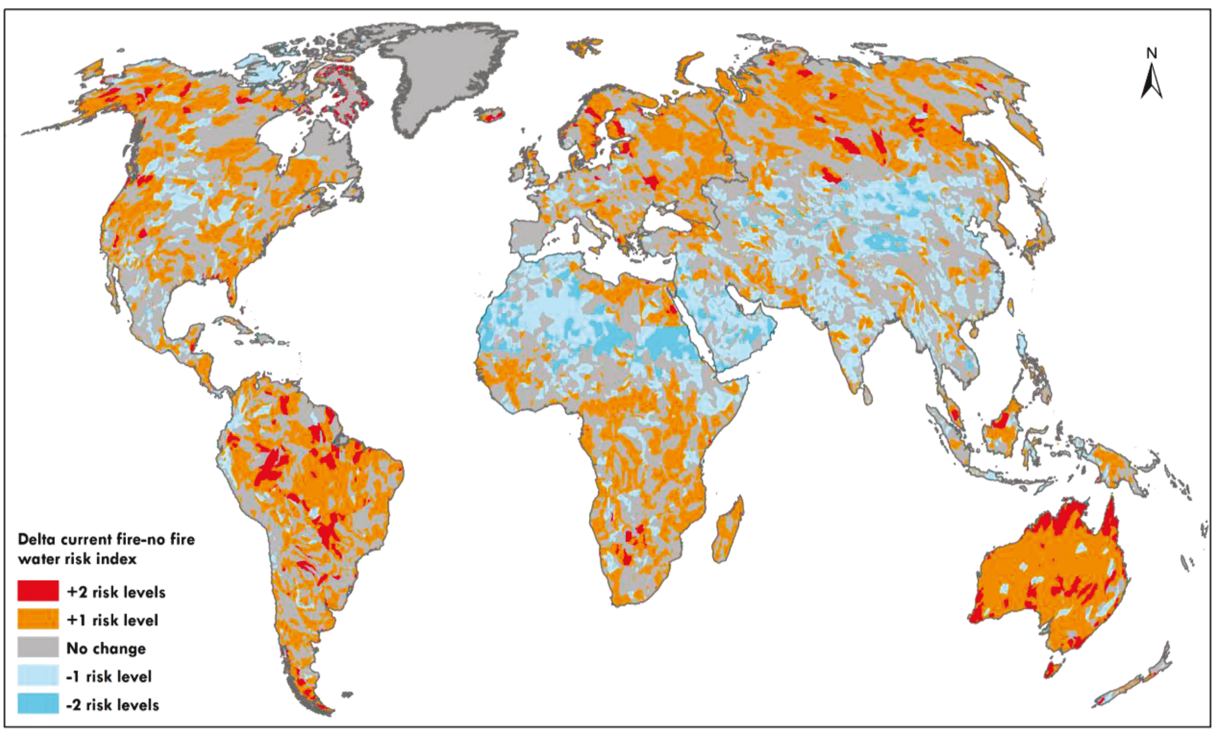

Figure 5-8: *Difference between the "fire-enhanced" and the original Aqueduct water risk indexes.*

Figures 5-8 and 5-9 show the future "fire-enhanced" risk and the difference in the final risk classification between the both "fire-enhanced" versions of the index, respectively. 1400 basins are in low level of risk, 7862 in low-to-medium, 8611 in medium-to-high, 2309 in high, and four in very high water risk. The difference in risk level classifies two water basins with an increase of two risk level, 3274 with a level increase, 15094 remain unchanged, 1812 have lowered their risk by one level and four by two levels of risk. Noticeable increases in future risk levels in sparsely vegetated areas are probably due to the higher FS scores in those areas, as this fire indicator is mainly driven by climate variables that are expected to be more fire-conducive in the future (Flannigan *et al.* 2013).

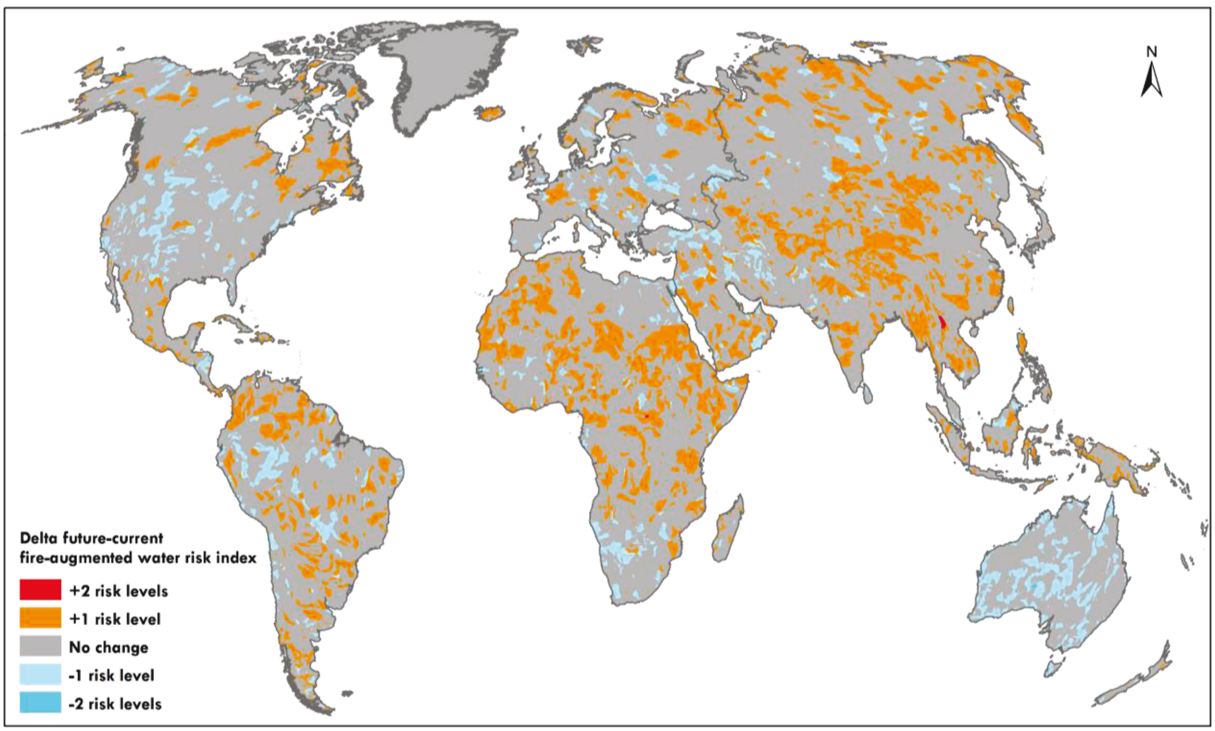

Figure 5-9: *Difference between future and current "fire-enhanced" water risk indexes.*

5.5 DISCUSSION

My study focuses on the integration of wildfire activity information in an existing global water risk analysis framework. The "fire-enhanced" water risk index provides a result that partly confirms my hypothesis: the spatial pattern of the risk changes significantly, reaffirming the need to include fire activity in global water assessments (Robinne *et al.* 2016). Including wildfire information highlights locations where post-fire hydrogeomorphic impacts to freshwater resources have been documented: in the Western USA (Miller *et al.* 2011), in the boreal forest of Canada (Prepas *et al.* 2003), in the Amazonian forest (Cochrane 2003), in Northern UK (Brown *et al.* 2015), in Japan (Seidel *et al.* 2017), and in Australia (Townsend and Douglas 2004). Noticeable consequences in such a number of locations displaying different socio-environmental settings raise concerns as to the reliability of the water supply to communities and ecosystems; disruptions to drinking-water distribution and environmental flows seem like a logical threat emerging from burned areas, although there is no documented case of the former (Smith *et al.* 2011; Bixby *et al.* 2015; Neary and Tecle 2015). This result suggests that natural or human-caused wildfires in those areas can be an important determinant of water-related risks. According to my results, it also suggests that other locations across the world, in South-America, in Africa, and in Northern Eurasia, depend on freshwater resources vulnerable to post-fire hydrogeomorphic disturbances. However, the quasi-absence of change, or even the reduction in risk levels in many parts of the Mediterranean area, the Indian sub-continent, the Northern African savannas, and the steppes of Eastern Asia, though highly fire-prone, are dominated by other drivers leading to water-related risks. In most cases these drivers are most likely related to physical and economic water scarcity and water pollution problems (Döll *et al.* 2009; Wada *et al.* 2011; Veldkamp *et al.* 2016).

The lack of change in future water risk levels when fire activity is included somewhat countered my expectation. This fact could be linked to the different directions of change taken by both fire indicators. Increases in risk level seems to mainly happened in dry water basins in which future climate conditions could likely become more conducive to fire, although those areas could also show a future state quasi-perpetual drought preventing vegetation growth and, thus, would rather experience a decrease in the risk linked to fire activity (Pan *et al.* 2013). This is a trend that is likely to be enhanced in the course of the century (Dai 2012; Flannigan *et al.* 2013). Despite a general global expansion of fire hazard levels for the 2091-2100 period, the trend does not seem to be sufficient to worsen fire-induced risks to water resources significantly. Water basins showing a decrease in their risk score appear to display lower area burned, which is a fire indicator that is influenced by the expansion of global human population in the future (Jones and Neill 2016; Kloster and Lasslop 2017). This phenomenon is linked to the fact that SPITFIRE echoes anthropogenic land use and land cover changes as a limiting factor of fire activity as part of an Earth System model. The increase of low and medium fire hazard could mainly be associated to climate change, whereas human influence would probably cause a reduction of high and very high hazard, a scheme already documented at the global scale (Bowman *et al.* 2011; Archibald *et al.* 2013).

The predicted expansion of a wealthier human population will also multiply the density of human-wildland interfaces (Theobald and Romme 2007; Knorr *et al.* 2016; Robinne *et al.* 2016), which will lower the size of wildfire events but will also increase the probability of ignition and the number of interface fires (Radeloff *et al.* 2005; Syphard *et al.* 2007). Fires in urban fringes can trigger very particular post-fire hydrogeomorphic issues showing a greater concentration of human-made pollutants (Burke *et al.* 2013), although such level of risk would be complicated to represent at a global scale. However, it lines up with many modeling results showing the future importance of global human population dynamics in controlling the fate of water resources and fire occurrence (Vörösmarty *et al.* 2007; Wada and Bierkens 2014; Knorr *et al.* 2016).

Many existing global water risk assessments, whether considering water scarcity, water consumption, or water security as a whole, usually place natural areas with plentiful water resources as low-risk locations. However, Vörösmarty et al. (2010) have shown that global patterns of water security risks can also depend on the diversity of threats taken into account, as well as the capacity to prevent and respond to water-related disasters. My results underline the growing vulnerability of the water supply to wildfires in a world where freshwater resources are a critical vector of social stability (Kinoshita *et al.* 2016; Martin 2016). As the Aqueduct project was created with corporate needs in mind, such results may encourage the industrial sector to evaluate potential water disruption due to fire activity as part of their risk management plans (Larson *et al.* 2012). It may also encourage governments and administration agencies to evaluate the potential indirect economic impact of post-fire hydrological risks. To my knowledge, such information is currently non-existent, a fact that highlights potential limitations in the applicability of risk management plans in areas of high fire hazard, exposed water supply, and vulnerable downstream businesses. My results have important implications for global water security matters, as they show wildfires as a source of enhanced water-related risk, which can contribute to a better understanding of the world water system (Alcamo *et al.* 2008). There are few examples where the importance of wildfires on water security has been recognized in management and policy. The US Forest to Faucets initiative is the only large-scale water risk assessment that explicitly incorporates fire hazards as a threat to national water security (Weidner and Todd 2011); Melbourne Water in Australia is, to my knowledge, the only example of a water basin management entity being responsible for fire management and firefighting in its jurisdiction (Melbourne Water 2017). Based on the results of this study, I argue that integrating vegetation fires as a water security stressor should be part of the reflection in any water resource risk analysis.

My approach suffers from some limitations. The primary data source, the Aqueduct geodatabase, needs improvements to harmonize the resulting water basin boundaries, as the first fusion of several datasets created geometric artefacts that have no hydrological relevance. The suppression of polygons less than 100km² should be completed by a precise control of remaining polygons to ensure that the hydrological connectivity is conserved. Moreover, the small size of many water basins creates a mismatch with the resolution of the raster layers used to create the fire indicators. These were too coarse to introduce inter and intra-basin variability, which is a problem when numerous small water basins are clustered on top of a few pixels of similar values. Fire hazard indicators should be included using higher resolution rasters, although information for future time periods may not be available, as current Earth-system models still operate at a coarse resolution. Taking into account future changes in water availability would also represent a significant improvement, as provided by a recent evolution of the Aqueduct database (Luck *et al.* 2015). This study uses projections of fire activity for 2091-2100 only, and only for the worst case RCP and SRES scenarios. Covering a larger time span combined with a higher diversity of global change outcomes could provide more insights as to the evolution rate of the "fire-enhanced" water risk around the world. Moreover, this study relies, on purpose, on a weighting scheme that underlines the potential effect of fire on freshwater resources; a parallel scheme based on the expert opinion of fire hydrologists would provide more realistic results. In relation to the weighting, the choice of indicators themselves is open to debate. One will never underline enough the use of indicators as proxies lining-up with a rationale that may not be shared by peers, a fact that can however help the indexation framework evolving. Nevertheless, no matter the number of indicators and the weighting scheme, users must limit the inferences they may want to draw from the results presented here.

5.6 CONCLUSION

Many studies on post-fire hydrogeomorphic changes have pointed to the potentially pervasive effects of wildfires on freshwater resources and a growing number of studies specifically have addressed wildfire risks to the drinking water supply. The study presented above aimed at enhancing the Aqueduct global water risk index, a widely used geospatial database, with wildfire hazard information. I showed that the inclusion of wildfire hazard indicators changes the global pattern of water-related risks, increasing risk levels in locations traditionally not identified as particularly exposed to such issues. However, future wildfire hazard levels did not trigger a noticeable change in the pattern of the risk compared to the current period, suggesting that the future of water-related risks is mainly tied to global population dynamics driving water consumption habits and land-cover modifications more than to changes in wildland fire activity.

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CHAPTER 6 - SYNTHESIS

This doctoral work explored the emergence of wildfire risks to water security and the threat they might embody in the current context of global environmental change. According to the rationale exposed in the introductory chapter, to my research question, and based on the results presented in this document, it is paramount to make a move on the systematic integration of wildfire risk assessment in the design of current and future policies targeting the achievement or the durability of global water security. Despite a plethora of multiscale and multi-environment research in the field of hydrology and water security, this research also highlights serious knowledge gaps of the scientific community when it comes to wildfire hazard and post-fire hydrology outside of countries experienced in wildfire risk management. In this respect, the presentation of the wildfire-water risk framework seems to offer a sound structure for the study of this emerging problem in any socio-environmental settings and any spatial scale –especially when the use of mathematical modeling is complicated or impossible– granted that the choice of representative variables and their relationship leading to a potential risk are supported by science, or at least local knowledge. This latter point underlines the importance of involving experts in a risk analysis based on indexation.

Chapters two and three were based on an indexation modelling approach, a method commonly used to shed light on new environmental issues, to share scientific findings in a simple yet meaningful way, and to benchmark the effect of implemented policies on the state of the environment. The significant number of publications and operational indices currently in use –for many decades in cases– across the world (e.g., Environmental Performance Index, Environmental Sustainability Index, Human Development Index, World Risk Index) clearly speak for their robustness as information and governance tools. My collaborators and I adapted published methodologies widely used in global water security evaluations to illustrate the wildfire-water risk concept, showing that the global exposure of water resources to wildfires and the heavy dependence of downstream ecosystems and human communities to upstream fire-prone areas was a problem worth a growing research focus. Chapters three and four built on this framework and the results it fetched to introduce measures of global environmental change, exploring both the effects of regional landscape anthropization and planetary climate change effects on the response of wildfire-water risks. My colleagues and I showed that the combination of a spatially extended wildfire activity with growing pressures on renewable freshwater resources would likely enforce wildfire-water risks in the future, although following different trajectories depending on future population patterns. Those results help to foster the importance of this emerging danger in the achievement and long-term protection of global water security.

Despite providing a valuable contribution to the fields of hydrological science and pyrogeography, this work also points at three main limitations. The first constraint of such work at a global scale lays in the strong cultural, socio-economic, and natural disparities displayed by the Earth system, which suggests that proper management of the wildfire-water risk would be better achieved with a systematic combination of social and natural sciences at a regional scale, if not local. However, if the scientific interest in wildfire impacts on watershed hydrology and geomorphology has been vivid for decades, the extension of this knowledge specifically to post-fire risks to water security has only been emerging for the past decade or so, and the volume of detailed knowledge is thus inherently limited, making any enactment of general wildfire-water risk management standards a challenging exercise. Similarly, a critical lack of data on post-fire hydrogeomorphic hazards and their consequences on socio-hydrosystems creates a second significant restraint. As 60% of world's population relies on surface freshwaters, it is reasonable to hypothesize that the quasi-ubiquitous nature of wildfires could have fostered or will foster potentially acute post-fire risks such as floods or water contamination episodes, including in countries in which scientific research on this matter is for now limited. In the light of ongoing environmental changes, it is, therefore, crucial to extend the research network on this problematic and to maximize the use of cutting-edge technological capabilities to collect, store, analyze, and share data. The final limitation pertains to the use of proxy variables, or indicators, which are by definition an abstraction of the reality and sensitive to interpretation. Index producers and users must never forget this inherent and important limitation in the creation process, the interpretation effort, and finally the potential use of resulting maps when it comes to it comes to translating the framework and the maps into the governance space. The amount of inferences that can be drawn from such work is rather limited, as an index, particularly at this scale, is not meant to designing management actions. That being said, the importance of global-scale indices must not be devaluated, as when used with care, they remain an important source of information to be linked with the proper communication tools towards the achievements of sustainable development goals.

I believe this exploration of the wildfire risk to water security offers a new avenue for the development of research projects focusing on the socioeconomic evaluation of ecosystem services. This work indeed points at the growing interests of preserving, or restoring, natural fire regimes in headwaters to ensure proper ecosystems' health. Maintaining the functioning of our natural capital, and the water services it provides implies a collective understanding and acknowledgement of the irreplaceable role of natural fires in the maintenance of the long-term resilience of socio-hydrosystems. Protecting water with fire, here is a major paradox that deserves immediate attention if we are to secure the renewable freshwater resources of a changing world. Although many aspects of the interacting dynamics between pyrogeography and hydrology remain to be understood, the multiplication of research efforts towards the recognition, the prevention, and the management of wildfire risks to socio-hydrological systems is highly encouraging. I hope this large-scale research will enlarge the scope of ongoing scientific endeavor to more fully include better those parts of the world where ongoing pressures on the blue gold deserve priority actions and will help to promote ecosystem services protection as one of the last resort to slow down global environmental change.

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APPENDICES

APPENDIX 1

Generalized Linear Model performance for each anthropogenic variable predicting Total Area Burned per ignition cause. (L) stands for log transform, (S) stands for square root transform. Significance codes: '*' 0.001; '**' 0.01; '*' 0.05; '.' 0.1; ' ' 1.**

Variable	TAB_ALL			TAB_HUM			TAB_NAT			TAB_UNK		
	DE = 39.66			DE = 4.13			DE = 35.45			DE = 9.8		
	DE	Gain	$\lvert p \rvert$	DE	Gain	$\lvert p \rvert$	DE	Gain	$\lvert p \rvert$	DE	Gain	$\lvert p \rvert$
(L) Distance to temporary (daily) fire attack bases	40.7	1.04	*	5.33	1.2	.	41.5	6.02	***	18.07	8.27	***
Distance to primary fire attack bases	40.46	0.8	.	5.71	1.58	*	36.9	1.49	*	11.09	1.29	.
(L) Distance to secondary fire attack bases	40.56	0.9	.	4.13	0		36.6	1.16	*	13.33	3.53	**
Distance to fire attack bases	40.52	0.86	.	6.39	2.26	*	39.4	3.98	***	18.12	8.32	***
Distance to buildings	43.76	4.1	***	5.81	1.68	.	38.7	3.21	***	20.02	10.2	***
Distance to mines	44.71	5.05	***	4.54	0.41		36.9	1.42		19.38	9.58	***
Distance to cutblocks	43.41	3.75	***	5.34	1.21	.	41.2	5.73	***	22.73	12.9	***
(S) Distance to transportation network	39.94	0.28		5.32	1.19	.	36.8	1.35	*	25.15	15.4	***
(L) Transportation network density	47.53	7.87	***	6.18	2.05	.	40.8	5.31	***	19.65	9.85	***
(L) Energy network density	42.98	3.32	***	4.86	0.73		40.4	4.99	***	11.82	2.02	*
(S) Seismic line density	45.16	5.5	***	4.75	0.62		39.9	4.41	**	18.66	8.86	***
Population per square kilometer	40.77	1.11		4.72	0.59		35.5	0		9.8	0	.
(S) Anthropogenic non-fuel	46.2	6.54	*	6.17	2.04	.	43	7.58	***	10.32	0.52	
(S) Human Footprint	43.8	4.14	*	5.45	1.32		38	2.58	**	13.48	3.68	**
(L) Distance to temporary (daily) fire attack bases	47.57	7.91	***	4.78	0.65		39.6	4.11	**	22.8	13	***

The model only accounting for area burned by human-caused fire ≥200ha has low statistical power (4.13% of deviance explained), and does have any significant anthropogenic variable. This is probably due to the under-representation of large human-caused fire in the sample used. Conversely, modeling natural causes only show many similarities with the model considering all causes. Linked to the previous point, lightning fires are comparatively over-represented in our sample, probably explaining these commonalities between both models.

The model for unknown causes is slightly more powerful, but still under 10% DE, although many anthropogenic variables are significant when added to the calculation, with a 15.4% gain in DE when the distance to transportation network is tested, and several other variables adding 8% to 10% in DE: Distance to fire attack bases, Distance to daily attack bases, Distance to buildings, Transportation network density, Seismic line density. This fact suggests that human causes may be hidden behind most of 'unknown' fire causes.

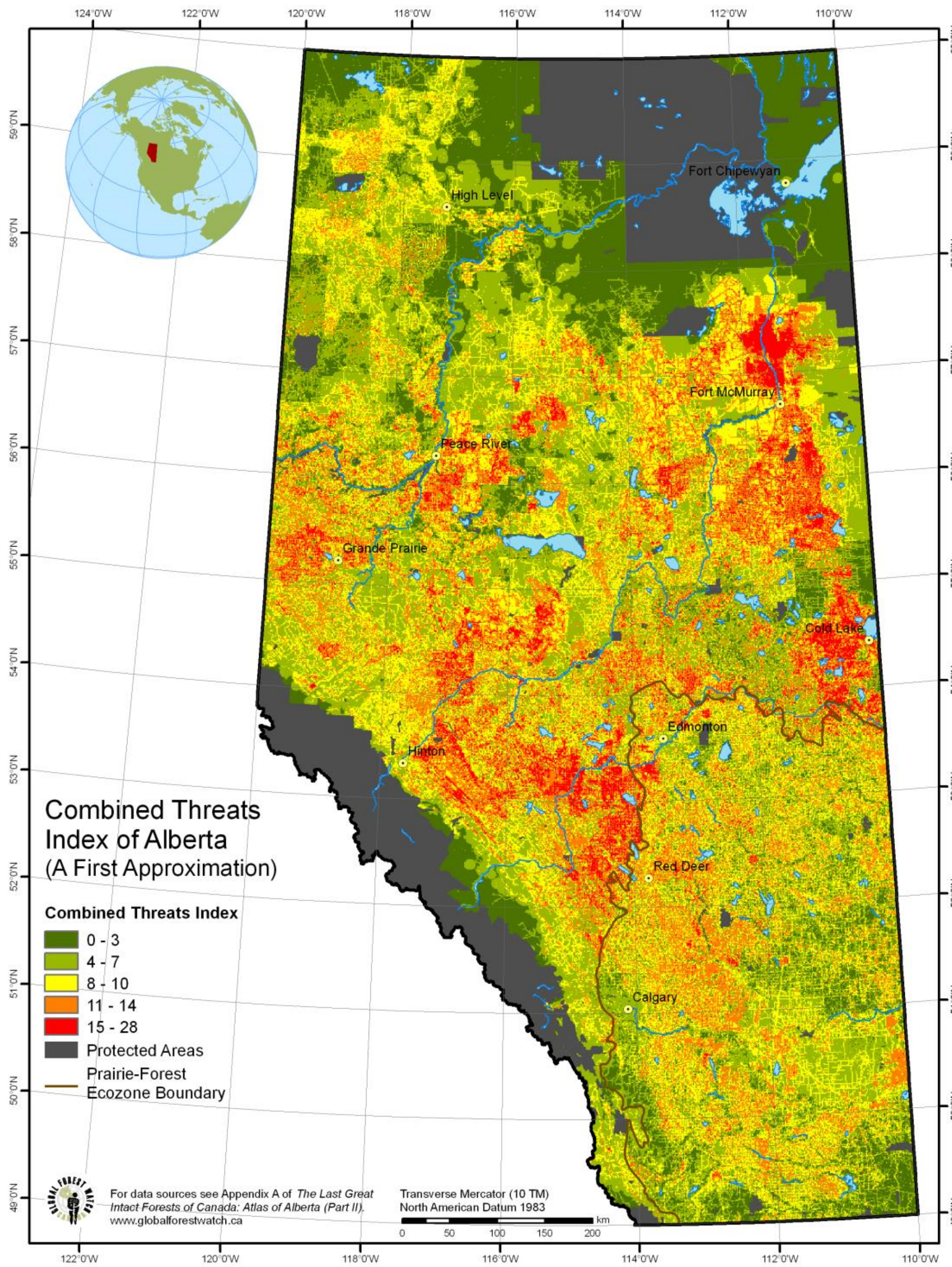
APPENDIX 2

Generalized Linear Model performance for each anthropogenic variable predicting Total Number of Ignition per ignition cause. (L) stands for log transform, (S) stands for square root transform. Significance codes: '*' 0.001; '**' 0.01; '*' 0.05; '.' 0.1; ' ' 1.**

Variable	TIN_ALL			TIN_HUM			TIN_NAT			TIN_UNK		
	DE = 28.19			DE = 20.69			DE = 48.82			DE = 5.84		
	DE	Gain	|*p*|	DE	Gain	|*p*|	DE	Gain	|*p*|	DE	Gain	|*p*|
(L) Distance to temporary (daily) fire attack bases	29.71	1.52	.	21.5	0.84		49.6	0.79	*	7.65	1.81	.
Distance to primary fire attack bases	29.9	1.71	**	25.4	4.68	***	49.2	0.39		6.13	0.29	
(L) Distance to secondary fire attack bases	28.79	0.6		21.3	0.61		49.1	0.29		6.41	0.57	
Distance to fire attack bases	29.71	1.52		24.3	3.61	**	49.5	0.69		7.94	2.1	.
Distance to buildings	30.46	2.27	**	22.9	2.2	*	50.1	1.26	*	7.75	1.91	.
Distance to mines	29.09	0.9	.	24.2	3.53	**	49.7	0.9	.	6.72	0.88	
Distance to cutblocks	29.21	1.02	.	24.2	3.52	**	52.9	4.03	***	6.53	0.69	
(S) Distance to transportation network	29.27	1.08	.	22.4	1.74		49.4	0.59	.	9.89	4.05	**
(L) Transportation network density	30.97	2.78	**	22.5	1.78	.	52.1	3.23	***	10.21	4.37	*
(L) Energy network density	28.55	0.36		23.4	2.74	*	51.6	2.74	***	6.74	0.9	
(S) Seismic line density	28.71	0.52		22.4	1.66	.	52	3.21	***	7.14	1.3	
Population per square kilometer	28.54	0.35		21.2	0.51		53.3	4.46	***	6.63	0.79	
(S) Anthropogenic non-fuel	31.43	3.24	***	30.6	9.87	***	61	12.2	***	7.47	1.63	
(S) Human Footprint	31.55	3.36	**	28	7.35	***	66.5	17.7	***	10.31	4.47	*
(L) Distance to temporary (daily) fire attack bases	29.17	0.98	.	21.6	0.91		50.2	1.35	**	7.41	1.57	

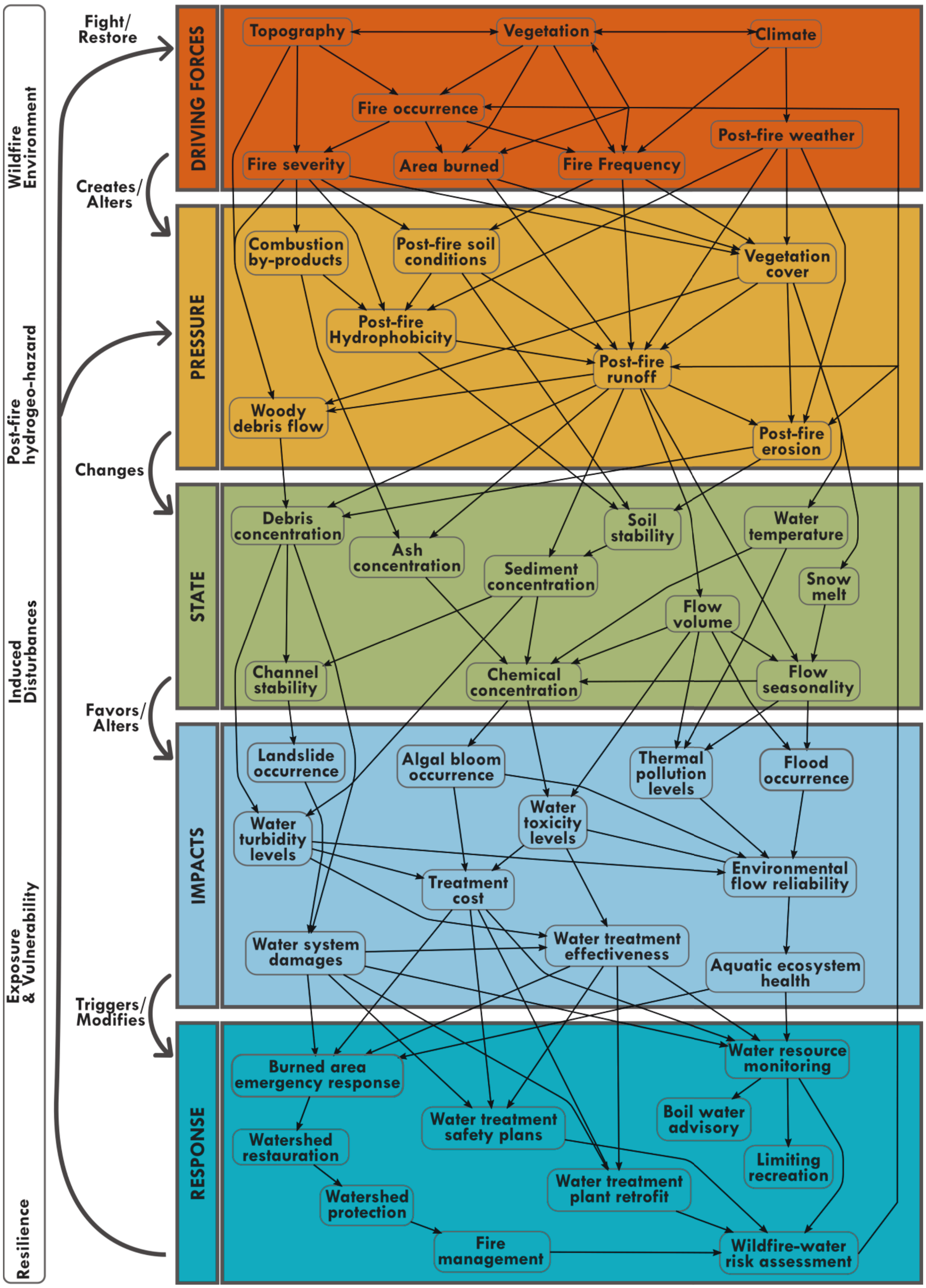
Modeling unknown-caused fire does not suggest any satisfying relationship. Modeling the total number of ignitions for fires >200ha is better achieved using natural causes only, probably for the same reason advanced in appendix one, as lightning fire represent most of our sample. Several anthropogenic variables add to the model statistical power, especially the human footprint and the non-fuel variables. The model considering a human ignition reaches only 21% deviance explained, potentially because of the relative independence of human-caused ignition and weather. It shows a fairly different set of significant anthropogenic variables, although non-fuel and human footprint are the most significant. The same scheme is observed for the model including all causes. This may suggest that the general human influence on the landscape remains the principal driver of human versus natural ignition.

APPENDIX 3



Study area and combined threats index. This index has been computed based on anthropogenic footprint and its impact on the natural landscape. Reproduced with the authorization of Global Forest Watch Canada (Lee et al. 2009).

APPENDIX 4



Wildfire Environment
Post-fire hydrogeo-hazard
Induced Disturbances
Exposure & Vulnerability
Resilience
Fight/ Restore
Creates/ Alters
Changes
Favors/ Alters
Triggers/ Modifies
DRIVING FORCES
Topography
Vegetation
Climate
Fire occurrence
Post-fire weather
Fire severity
Area burned
Fire Frequency
PRESSURE
Combustion by-products
Post-fire soil conditions
Vegetation cover
Post-fire Hydrophobicity
Post-fire runoff
Woody debris flow
Post-fire erosion
STATE
Debris concentration
Ash concentration
Soil stability
Water temperature
Sediment concentration
Snow melt
Flow volume
Channel stability
Chemical concentration
Flow seasonality
IMPACTS
Landslide occurrence
Algal bloom occurrence
Thermal pollution levels
Flood occurrence
Water turbidity levels
Water toxicity levels
Environmental flow reliability
Treatment cost
Water system damages
Water treatment effectiveness
Aquatic ecosystem health
RESPONSE
Burned area emergency response
Water resource monitoring
Boil water advisory
Water treatment safety plans
Watershed restauration
Limiting recreation
Water treatment plant retrofit
Watershed protection
Fire management
Wildfire-water risk assessment

Detailed version of the Driving Forces-Pressure-State-Impact-Response framework applied to the wildfire-water risk analysis, as shown in Fig.3-1.

APPENDIX 5

CREATION OF THE GLOBAL INDICATORS

Each the DPSIR category was assigned a certain number of indicators approximating the different processes leading to the wildfire-water risk. As it is not practical and not desirable to aggregate 34 indicators, we collapsed the information respective to each category using a Principal Component Analysis. The tables resulting from this process – covariance matrix, correlation matrix, eigenvalues, and accumulative eigenvalues– are provided hereafter for each category.

COVARIANCE MATRIX

# Layer	1	2	3	4	5	6	7	8
1	0.48977	0.46443	0.37099	0.35879	0.28205	0.26733	0.37299	0.49920
2	0.46443	0.50037	0.40081	0.38432	0.28302	0.27461	0.37315	0.48521
3	0.37099	0.40081	0.42711	0.29838	0.22461	0.22799	0.29554	0.39930
4	0.35879	0.38432	0.29838	0.40124	0.24058	0.22732	0.30162	0.37213
5	0.28205	0.28302	0.22461	0.24058	0.22303	0.20642	0.25853	0.26976
6	0.26733	0.27461	0.22799	0.22732	0.20642	0.19970	0.24364	0.26662
7	0.37299	0.37315	0.29554	0.30162	0.25853	0.24364	0.32570	0.36696
8	0.49920	0.48521	0.39930	0.37213	0.26976	0.26662	0.36696	0.67811

==

CORRELATION MATRIX

# Layer	1	2	3	4	5	6	7	8
1	1.00000	0.93817	0.81115	0.80937	0.85338	0.85479	0.93389	0.86622
2	0.93817	1.00000	0.86701	0.85771	0.84722	0.86875	0.92435	0.83298
3	0.81115	0.86701	1.00000	0.72077	0.72776	0.78067	0.79239	0.74197
4	0.80937	0.85771	0.72077	1.00000	0.80421	0.80308	0.83436	0.71341
5	0.85338	0.84722	0.72776	0.80421	1.00000	0.97812	0.95924	0.69367
6	0.85479	0.86875	0.78067	0.80308	0.97812	1.00000	0.95535	0.72454
7	0.93389	0.92435	0.79239	0.83436	0.95924	0.95535	1.00000	0.78084
8	0.86622	0.83298	0.74197	0.71341	0.69367	0.72454	0.78084	1.00000

==

EIGENVALUES AND EIGENVECTORS

# Number of Input Layers	Number of Principal Component Layers
8	8

# PC Layer	1	2	3	4	5	6	7	8
# Eigenvalues								
	2.75580	0.19353	0.12139	0.09243	0.04950	0.02279	0.00628	0.00328
# Eigenvectors								
# Input Layer								
1	0.40724	-0.05894	-0.10728	0.21658	0.61041	0.57158	0.26172	0.06688

2	0.41425	0.07653	0.13388	-0.09511	0.45411	-0.75545	0.11558	-0.07276
3	0.34566	0.01220	0.87935	-0.06196	-0.22520	0.21577	-0.05552	-0.05416
4	0.33552	0.31652	-0.27142	-0.81278	-0.12726	0.18887	-0.01289	0.03043
5	0.25502	0.33853	-0.19614	0.32489	-0.32219	0.01079	0.24128	-0.71708
6	0.24598	0.27327	-0.08189	0.28586	-0.38411	-0.12253	0.39996	0.67371
7	0.32861	0.25179	-0.15489	0.30338	0.00863	0.00504	-0.83383	0.13119
8	0.44382	-0.79854	-0.23341	-0.01700	-0.32073	-0.07220	-0.03796	-0.03276

==

PERCENT AND ACCUMULATIVE EIGENVALUES

# PC Layer	EigenValue	Percent of EigenValues	Accumulative of EigenValues
1	2.75580	84.9243	84.9243
2	0.19353	5.9641	90.8883
3	0.12139	3.7408	94.6292
4	0.09243	2.8485	97.4777
5	0.04950	1.5253	99.0030
6	0.02279	0.7024	99.7054
7	0.00628	0.1934	99.8988
8	0.00328	0.1012	100.0000

#
==

Pressure

COVARIANCE MATRIX

# Layer	1	2	3	4	5	6
1	0.25062	0.27802	0.18424	0.24531	-0.05610	0.31368
2	0.27802	0.37781	0.19893	0.27392	-0.03703	0.37021
3	0.18424	0.19893	0.14729	0.18257	-0.05956	0.21132
4	0.24531	0.27392	0.18257	0.27636	-0.03660	0.31365
5	-0.05610	-0.03703	-0.05956	-0.03660	0.40088	0.03091
6	0.31368	0.37021	0.21132	0.31365	0.03091	0.46630

#
===

CORRELATION MATRIX

# Layer	1	2	3	4	5	6
1	1.00000	0.90349	0.95896	0.93211	-0.17700	0.91759
2	0.90349	1.00000	0.84329	0.84770	-0.09515	0.88201
3	0.95896	0.84329	1.00000	0.90490	-0.24512	0.80635
4	0.93211	0.84770	0.90490	1.00000	-0.10997	0.87372
5	-0.17700	-0.09515	-0.24512	-0.10997	1.00000	0.07149
6	0.91759	0.88201	0.80635	0.87372	0.07149	1.00000

#
===

EIGENVALUES AND EIGENVECTORS

Number of Input Layers Number of Principal Component Layers
6 6

# PC Layer	1	2	3	4	5	6
# Eigenvalues						
	1.38541	0.41690	0.05449	0.04033	0.01961	0.00251
# Eigenvectors						
# Input Layer						
1	0.41777	-0.05822	-0.18907	-0.02348	-0.44991	-0.76378
2	0.49543	0.00755	0.74029	-0.42947	0.14778	0.01331
3	0.30150	-0.09789	-0.30223	-0.35360	-0.57304	0.59562
4	0.42298	-0.01264	-0.56187	-0.25259	0.66429	-0.01213
5	-0.05902	0.97227	-0.06878	-0.19751	-0.07788	-0.03742

```
     6          0.55382      0.20372      0.06671      0.76627      0.00351      0.24527
#
======================================================================

#              PERCENT AND ACCUMULATIVE EIGENVALUES

# PC Layer   EigenValue   Percent of EigenValues   Accumulative of EigenValues
     1       1.38541         72.1844              72.1844
     2       0.41690         21.7219              93.9063
     3       0.05449          2.8393              96.7456
     4       0.04033          2.1015              98.8471
     5       0.01961          1.0219              99.8690
     6       0.00251          0.1310             100.0000
#
======================================================================
```

State

COVARIANCE MATRIX

# Layer	1	2	3	4	5	6	7	8
1	0.44353	0.23118	0.16512	0.20724	0.39125	0.43368	0.36119	0.20212
2	0.23118	0.22779	0.17656	0.20490	0.30949	0.33362	0.28342	0.19157
3	0.16512	0.17656	0.23531	0.20735	0.27754	0.28387	0.25423	0.14420
4	0.20724	0.20490	0.20735	0.25654	0.35057	0.37742	0.31313	0.21479
5	0.39125	0.30949	0.27754	0.35057	0.68485	0.74817	0.62235	0.30766
6	0.43368	0.33362	0.28387	0.37742	0.74817	0.84044	0.68325	0.34761
7	0.36119	0.28342	0.25423	0.31313	0.62235	0.68325	0.75172	0.28166
8	0.20212	0.19157	0.14420	0.21479	0.30766	0.34761	0.28166	0.36406

===

CORRELATION MATRIX

# Layer	1	2	3	4	5	6	7	8
1	1.00000	0.72730	0.51112	0.61438	0.70990	0.71032	0.62552	0.50300
2	0.72730	1.00000	0.76262	0.84762	0.78358	0.76249	0.68491	0.66524
3	0.51112	0.76262	1.00000	0.84393	0.69136	0.63834	0.60448	0.49267
4	0.61438	0.84762	0.84393	1.00000	0.83638	0.81281	0.71305	0.70282
5	0.70990	0.78358	0.69136	0.83638	1.00000	0.98616	0.86738	0.61616
6	0.71032	0.76249	0.63834	0.81281	0.98616	1.00000	0.85961	0.62843
7	0.62552	0.68491	0.60448	0.71305	0.86738	0.85961	1.00000	0.53840
8	0.50300	0.66524	0.49267	0.70282	0.61616	0.62843	0.53840	1.00000

===

EIGENVALUES AND EIGENVECTORS

# Number of Input Layers	Number of Principal Component Layers
8	8

# PC Layer	1	2	3	4	5	6	7	8
# Eigenvalues								
	2.99471	0.25767	0.20404	0.15745	0.12280	0.03714	0.02240	0.00803
# Eigenvectors								
# Input Layer								
1	0.29803	-0.22093	0.88243	-0.07680	-0.16465	0.21598	-0.06306	-0.00710

2	0.23466	-0.30130	0.06669	0.22671	-0.12496	-0.86609	0.17961	0.01825
3	0.20654	-0.24422	-0.17050	0.67857	-0.15023	0.42054	0.43715	0.13281
4	0.25735	-0.27623	-0.19759	0.29972	0.02761	0.04822	-0.85169	-0.01125
5	0.46692	0.16314	-0.04334	0.03594	0.39803	0.03053	0.13552	-0.75794
6	0.51396	0.19885	-0.02524	-0.12948	0.51895	-0.01504	0.05865	0.63713
7	0.45366	0.51297	-0.16660	-0.11348	-0.69795	-0.00818	-0.05390	0.01754
8	0.24341	-0.62862	-0.34421	-0.60123	-0.13810	0.15113	0.15024	-0.03376

==

PERCENT AND ACCUMULATIVE EIGENVALUES

# PC Layer	EigenValue	Percent of EigenValues	Accumulative of EigenValues
1	2.99471	78.7205	78.7205
2	0.25767	6.7732	85.4937
3	0.20404	5.3636	90.8573
4	0.15745	4.1388	94.9961
5	0.12280	3.2279	98.2240
6	0.03714	0.9762	99.2002
7	0.02240	0.5888	99.7890
8	0.00803	0.2110	100.0000

#
==

Impact

```
#                COVARIANCE MATRIX

#   Layer          1          2          3          4          5          6
#  ---------------------------------------------------------------------------
     1     1.963511e+000 1.475014e-002 -1.056570e-003 9.274204e-003 5.836966e-003 2.095251e-002
     2     1.475014e-002 2.415855e-003 -6.948309e-006 4.136448e-005 3.408815e-005 1.183219e-004
     3     -1.056570e-003 -6.948309e-006 7.353174e-005 -6.125268e-006 -1.068579e-005 -2.253218e-005
     4     9.274204e-003 4.136448e-005 -6.125268e-006 1.200272e-004 3.688471e-005 1.234141e-004
     5     5.836966e-003 3.408815e-005 -1.068579e-005 3.688471e-005 1.790794e-004 8.871951e-005
     6     2.095251e-002 1.183219e-004 -2.253218e-005 1.234141e-004 8.871951e-005 3.652748e-004
#  ===========================================================================


#                CORRELATION MATRIX

#   Layer          1          2          3          4          5          6
#  ---------------------------------------------------------------------------
     1        1.00000    0.21416    -0.08793    0.60412    0.31128    0.78237
     2        0.21416    1.00000    -0.01649    0.07682    0.05183    0.12596
     3       -0.08793   -0.01649     1.00000   -0.06520   -0.09312   -0.13749
     4        0.60412    0.07682    -0.06520    1.00000    0.25158    0.58941
     5        0.31128    0.05183    -0.09312    0.25158    1.00000    0.34689
     6        0.78237    0.12596    -0.13749    0.58941    0.34689    1.00000
#  ===========================================================================


#              EIGENVALUES AND EIGENVECTORS

# Number of Input Layers     Number of Principal Component Layers
        6                        6
# PC Layer          1          2          3          4          5          6
#  ---------------------------------------------------------------------------
# Eigenvalues
                1.96391    0.00231    0.00019    0.00013    0.00007    0.00007
# Eigenvectors
# Input Layer
     1        0.99990   -0.00725     0.00974    0.00720   -0.00154     0.00061
     2        0.00752    0.99971    -0.01715   -0.01340    0.00522    -0.00477
     3       -0.00054    0.00056     0.11318    0.06944    0.87306     0.46918
     4        0.00472   -0.01295    -0.19560   -0.23196    0.48464    -0.82029
     5        0.00297   -0.00486    -0.77294    0.63290    0.03564     0.02648
     6        0.01067   -0.01834    -0.59254   -0.73524   -0.03988     0.32600
```

```
#  ========================================================================

#             PERCENT AND ACCUMULATIVE EIGENVALUES

# PC Layer   EigenValue   Percent of EigenValues   Accumulative of EigenValues
      1       1.96391        99.8598              99.8598
      2       0.00231         0.1173              99.9770
      3       0.00019         0.0094              99.9865
      4       0.00013         0.0065              99.9929
      5       0.00007         0.0037              99.9966
      6       0.00007         0.0034             100.0000
#=========================================================================
```

Response

```
#                COVARIANCE MATRIX

#   Layer          1          2          3
4
#  ------------------------------------------------------------------------
     1        1.31271     1.26222     0.78034     0.29319
     2        1.26222     2.19051     1.48460     0.36853
     3        0.78034     1.48460     1.05508     0.25673
     4        0.29319     0.36853     0.25673     0.13315
#
========================================================================


#                CORRELATION MATRIX

#   Layer          1          2          3
4
#  ------------------------------------------------------------------------
     1        1.00000     0.74435     0.66307     0.70128
     2        0.74435     1.00000     0.97655     0.68240
     3        0.66307     0.97655     1.00000     0.68497
     4        0.70128     0.68240     0.68497     1.00000
#
========================================================================


#             EIGENVALUES AND EIGENVECTORS

# Number of Input Layers     Number of Principal Component Layers
        4                        4
# PC Layer           1          2          3          4
#  ------------------------------------------------------------------------
# Eigenvalues
               4.09532     0.51416     0.06169     0.02029
# Eigenvectors
# Input Layer
     1        0.47699     0.85955     0.09515    -0.15686
     2        0.72034    -0.32739     0.25933     0.55378
     3        0.48548    -0.38268    -0.22055    -0.75447
     4        0.13375     0.08689    -0.93544     0.31545
#
========================================================================


#             PERCENT AND ACCUMULATIVE EIGENVALUES
```

# PC Layer	EigenValue	Percent of EigenValues	Accumulative of EigenValues
1	4.09532	87.2931	87.2931
2	0.51416	10.9596	98.2527
3	0.06169	1.3149	99.5676
4	0.02029	0.4324	100.0000

#
===

APPENDIX 6

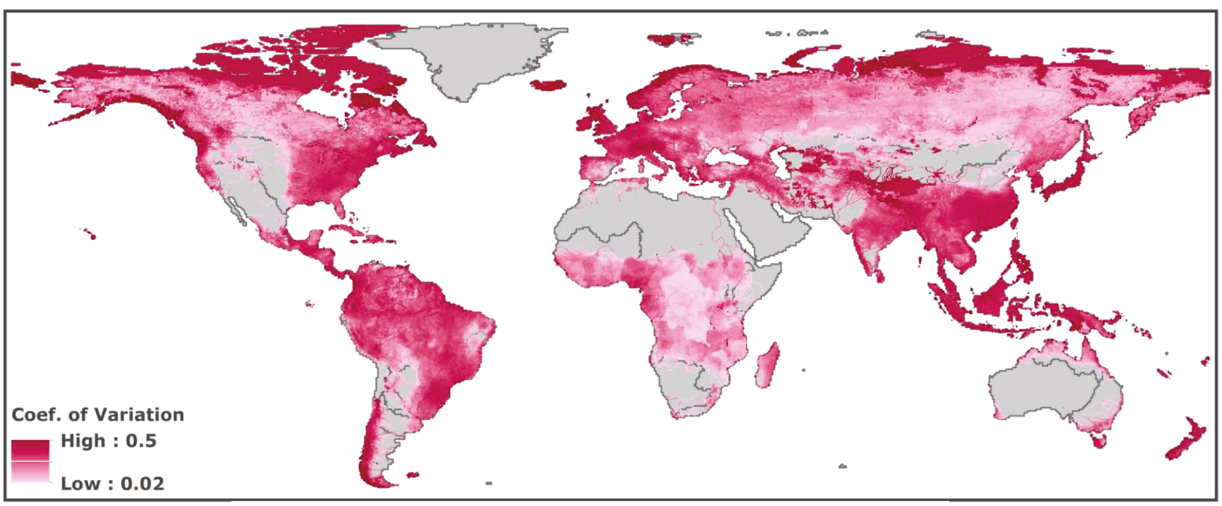
CORRELATION BETWEEN GLOBAL INDICATORS

The table hereafter presents Spearman correlation coefficients between the five global indicators used to compute the global WWR index.

	Drivers	**Pressure**	**State**	**Impact**	**Response**
Drivers	1				
Pressure	0.954	1			
State	0.92	0.92	1		
Impact	0.542	0.349	0.509	1	
Response	0.154	0.032	0.054	0.277	1

SPATIAL SENSITIVITY OF THE INDEX

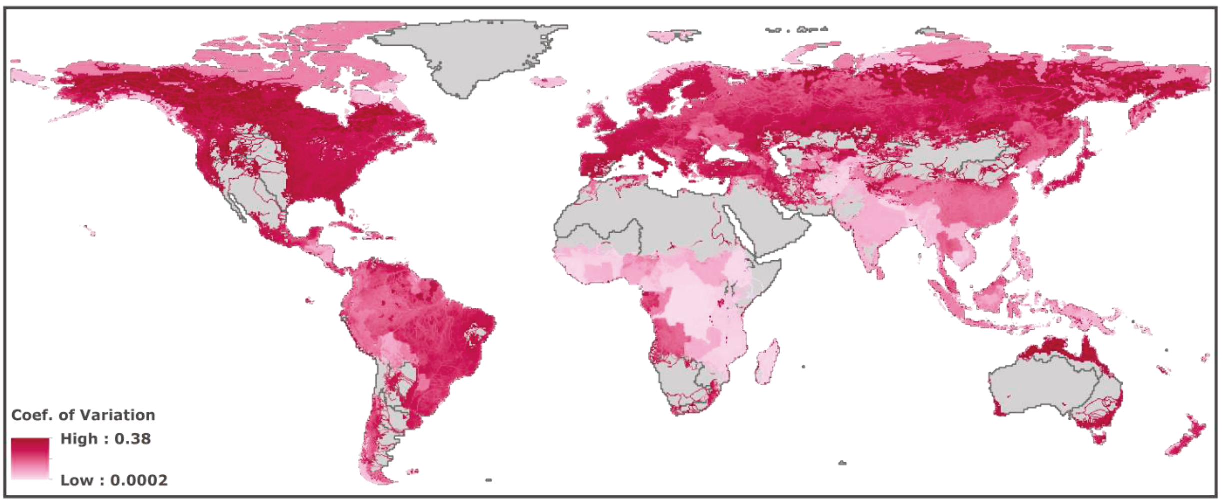
- Jackknifing



Coefficient of variation for Jackknife sensitivity analysis

Jackknifing sensitivity analysis is based on the recalculation of the index minus one of the global indicators. It is done iteratively so all the combinations, based on the remaining four global indicators, are tested (i.e. 20). We present here the coefficient of variation resulting of the permutations. It appears that two main indicators seem to control the geography of the WWR. First, the sensitivity to fire activity controls the risk in highest and lowest latitudes, as well as in mountain ranges. It seems logical as those areas are naturally less prone to fire activity because of fuel and ignition limitations. An exception is the Amazonian forest, which also presents some variability due to limitations in fire activity. Second, the potential impact, highly driven by population, seems to be the main control in densely populated areas, such as in Eastern North-America, Western Europe, South-East Asia, and Indonesia.

- High-Low case scenarios



Coefficient of variation for High-Low case scenario sensitivity analysis

High-low case sensitivity analysis changes the weights of the indicators according to their distribution. In a low case scenario, those indicators whose distribution is skewed to the right are given the highest weight (in a predefined range) and the lowest weight is given the left-skewed indicators, and vice-versa for the high case scenario. We present here the coefficient of variation resulting from those two scenarios. The overarching control seems to be the Response category, which defines resilience capacities and is highly influenced by GDP. Therefore, developed countries and the BRICS, with higher GDPs, are clearly advantaged, especially in a high-case scenario. It also seems that the Drivers, Pressure, and

State categories play a higher role in defining the spatial pattern of the index. Northernmost latitudes, Africa, South East Asia, and Indonesia appear less sensitive to those changes in indicators weighting, as they show the lowest values in the Response category, a fact that is hardly diminished by a higher weighting of the lower end of the distribution or a lower weighting of the upper end. High-case scenarios helps thinking in terms of best and worst case scenario, for which our sensitivity analysis shows that even if more fire occur in the Northern hemisphere, the capacity to respond to potential disaster make those areas more resilient.