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A RAPID RESPONSE ASSESSMENT

SPREADING LIKE WILDFIRE

THE RISING THREAT OF EXTRAORDINARY LANDSCAPE FIRES

UNITED NATIONS DECADE ON
ECOSYSTEM
RESTORATION
2021-2030

Ministry for Foreign
Affairs of Finland

NICFI
Norway's
International Climate
and Forest Initiative

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A RAPID RESPONSE ASSESSMENT

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Foreword

Across Earth's ecosystems, wildfires are growing in intensity and spreading in range. From Australia to Canada, the United States to China, across Europe and the Amazon, wildfires are wreaking havoc on the environment, wildlife, human health, and infrastructure. *Spreading like Wildfire: The Rising Threat of Extraordinary Fires* is the first report by UNEP and GRID-Arendal to take stock of the scale and extent of the global wildfire crisis and has been commissioned in support of the UN Decade of Ecosystem Restoration. Over 50 experts from research institutions, government agencies, and international organisations from around the globe have contributed to this report. Their findings are that while the situation is certainly extreme, it is not yet hopeless.

Fire is changing because we are changing the conditions in which it occurs. Not all fires are harmful, and not all fires need to be extinguished as they serve important ecological purpose. However, wildfires that burn for weeks and that may affect millions of people over thousands of square kilometres present a challenge that, right now, we are not prepared for. Lightning strikes and human carelessness have always – and will always – spark uncontrolled blazes, but anthropogenic climate change, land-use change, and poor land and forest management mean wildfires are more often encountering the fuel and weather conditions conducive to becoming destructive. Wildfires are burning longer and hotter in places they have always occurred, and are flaring up in unexpected places too, in drying peatlands and on thawing permafrost.

The costs in human lives and livelihoods can be counted in the number who perish in the flames, or contract respiratory diseases from the toxic smoke, or in the number of towns, homes, businesses, and communities affected by fire. A recent study published in *The Lancet* indicates that annual exposure to wildfire smoke results in more than 30,000 deaths across the 43 countries included in the study. Other species also pay the price: besides a devastating loss of habitat, the smouldering swathes of land left behind in a wildfire's wake are scattered with the charred remains of animals and plants possibly fast-tracking extinctions. Last year, fires that got out of control in the Pantanal, the world's largest tropical wetland in Latin America, destroyed almost a third of one of the world's greatest biodiversity hotspots and there are now genuine concerns that this wetland will never fully recover. Not only can wildfires reduce biodiversity, but they contribute to a climate change feedback loop by emitting huge quantities of greenhouse gases into the atmosphere, spurring more warming, more drying, and more burning.

Over 50 experts from research institutions, government agencies, and international organisations from around the globe have contributed to this report. Their findings are that while the situation is certainly extreme, it is not yet hopeless.

The heating of the planet is turning landscapes into tinderboxes, while more extreme weather means stronger, hotter, drier winds to fan the flames. Too often, our response is tardy, costly, and after the fact, with many countries suffering from a chronic lack of investment in planning and prevention. This report makes it clear that the true cost of wildfires – financial, social, and environmental – extends for days, weeks, and even years after the flames subside. To better prepare ourselves and limit the widespread damage done by wildfires, we need to take heed of the clear warnings and recommendations for future action outlined in this report. We must work with nature, communities, harness local knowledge, and invest money and political capital in reducing the likelihood of wildfires starting in the first place and the risk of damage and loss that comes when they do. For policymakers, these are the crucial next steps:

1 Audit your full wildfire costs and invest in planning, prevention, and recovery, not just response: One assessment estimated the annualized economic burden from wildfire for the United States to be between $71.1 billion to $347.8 billion ($2016 US). Most nations do not have any assessment. Commonly, more than half the expenditures related to wildfires are for response, while planning typically receives just 0.2 per cent of the total budget for wildfires. However, to reduce the outsized costs from damage and loss – which greatly exceeds all spending on wildfire management – we need to rebalance our efforts. While more work is required to determine the optimal allocation for each country, as a starting point, countries may consider rebalancing investments by up to 1 per cent for planning, 32 per cent for prevention, 13 per cent for preparedness, 34 per cent for response, and up to 20 per cent for recovery.

2 Learn from others, best practice is out there: Governments and communities need to proactively learn from each other's experiences, seeking out best practices and inspiring examples from around the world with the sharing of data, information, and analysis to improve forecasting and learning. Specific efforts to include both indigenous leaders and women in disaster risk management is crucial as research has shown that their input on risk reduction is fundamental towards effective solutions. No single country has yet formulated the perfect response, but many are making progress in different aspects of managing the risks of wildfires. Together we can learn from each other.

3 A stronger multilateral response is needed: The UN system lacks robust wildfire expertise dedicated to this challenge as the management of wildfires is seen as mainly a national responsibility. This must change. Wildfires need to be placed in the same category of global humanitarian response as major earthquakes and floods. New capability and financial support should be made available to affected countries, with engagement from Civil Defense. Fires do not respect national borders, so a coordinated, agile, and anticipatory wildfire management mechanism is needed.

Regardless of mitigation, the authors say, "we must learn to live with fire," and indeed we must. We must learn to better manage and mitigate the risk of wildfires to human health and livelihoods, biodiversity, and the global climate. This report provides a roadmap for doing just that.

Susan Gardner
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We must learn to better manage and mitigate the risk of wildfires to human health and livelihoods, biodiversity, and the global climate. This report provides a roadmap for doing just that.

Summary

Experiencing a wildfire at close proximity is a dangerous and terrifying experience. Uncontrollable and devastating wildfires are becoming an expected part of our seasonal calendars. Wildfires occur on every continent except Antarctica and most regions experience weather conditions conducive to the outbreak of a wildfire at some point in the year.

For the purposes of this report, "wildfire" is defined as "an unusual or extraordinary free-burning vegetation fire which may be started maliciously, accidently, or through natural means, that negatively influences social, economic, or environmental values". In contrast, the landscape fires that we are more accustomed to are an integral part of our world, critical to the healthy functioning of many ecosystems and an important cultural and land management tool. Whether caused by humans or nature, when fires burn out of control, they can become wildfires. {Section 1.1 of full report}

	Landscape fires	**Wildfires**
Frequency	Often seasonal; occur under moderate fire conditions; quite often intentionally lit	Linked to extreme fire weather
Intensity	Low to moderate intensity with short episodes of high intensity	Mostly high intensity with some periods of moderate intensity
Suppressibility	Easily controlled with regular firefighting resources	Control measures may exceed regular firefighting resources
Impact	Low impact, with a positive impact on some species	High impact on one or more values (social, economic, environmental)

The risk of wildfires is changing

The risk that wildfires pose to people and the environment is increasing due to numerous factors, including, but not limited to, climate change. A wildfire results from a complex interaction of biological, meteorological, physical, and social factors that influence the likelihood of a wildfire breaking out, its propagation and intensity, duration and extent, and its potential to cause damage to economies, the environment, and society (see Figure S1). Around the world many of these factors – climate, land use and land management practices, and demographics – are changing. As a consequence, the risk of wildfires in many regions is also changing. Where wildfires previously occurred, risk may increase or decrease; in regions that previously did not experience wildfires, risk is increasing. {Recommendation 1}

Eliminating the risk of wildfires is not possible, but much can be done to manage and reduce risks. There are management actions that can be taken to mitigate the risk of wildfires and their impact, including restriction of activities that might lead to accidental fire ignitions, management of vegetation and vegetation debris (wildfire fuel) to decrease fire hazard prior to wildfires occurring, management of

Figure S1. Factors influencing wildfire outcomes and management options. A wildfire is the result of a complex interaction of biological, meteorological, physical, and social factors that influence its likelihood, behaviour, duration, extent, and outcome (i.e., severity or impact). Changes in many of these factors are increasing the risk of wildfire globally (e.g., climate change is increasing the frequency and severity of weather conducive to wildfire outbreaks, changed demographics in high-risk regions are increasing the potential impacts of wildfires). Management options at junctures, such as fuel management (managing fuels prior to a wildfire occurrence), fire management (undertaking fighting of the fire once it has started), or relocating those threatened during a wildfire event (e.g., evacuation) can mitigate some of the economic, environmental, or societal impacts of wildfire but it is impossible to mitigate all risks for all fires. As a result, communities often have to learn to live with the residual risk of wildfire.

Factors influencing wildfire outcomes and management actions

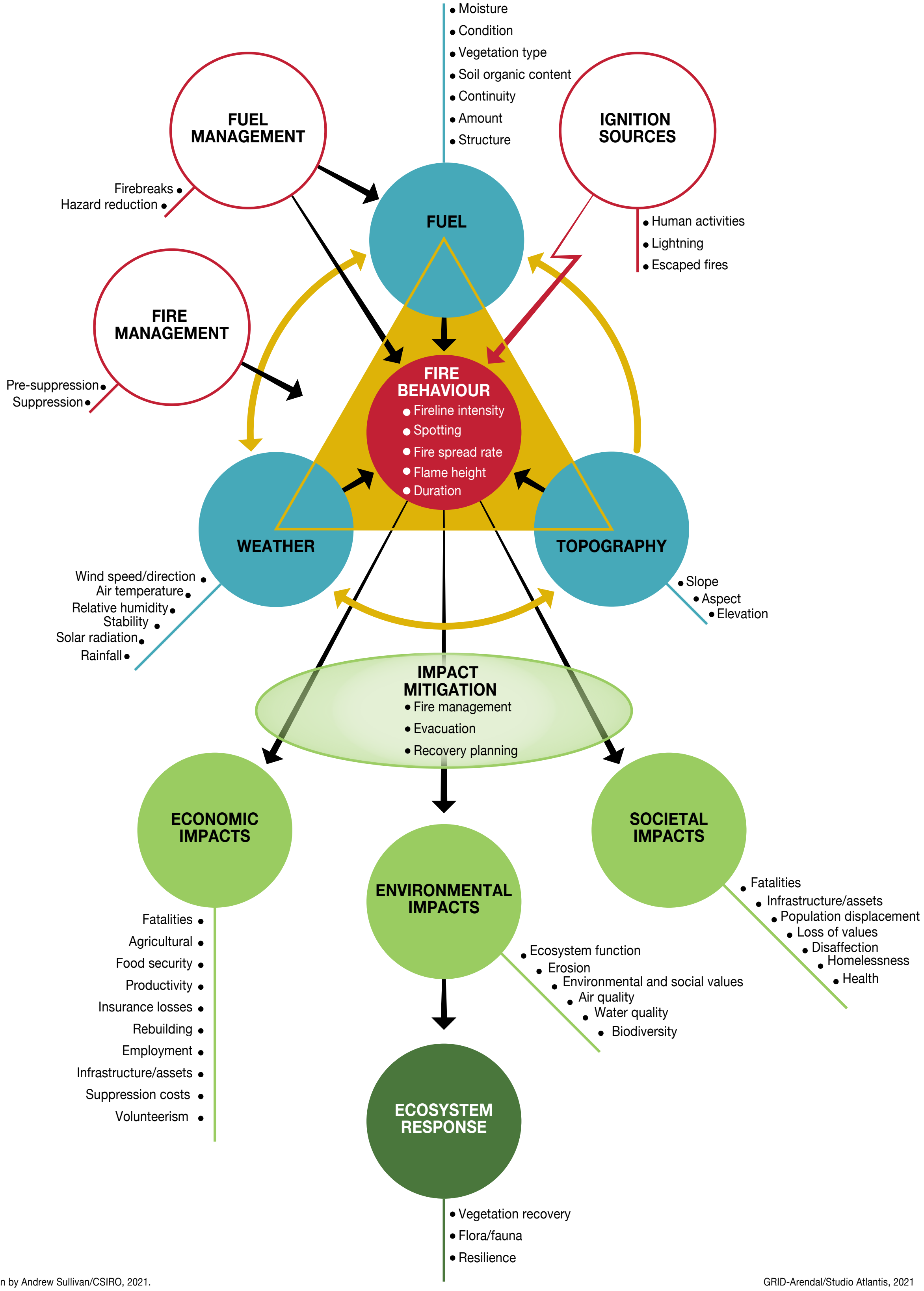
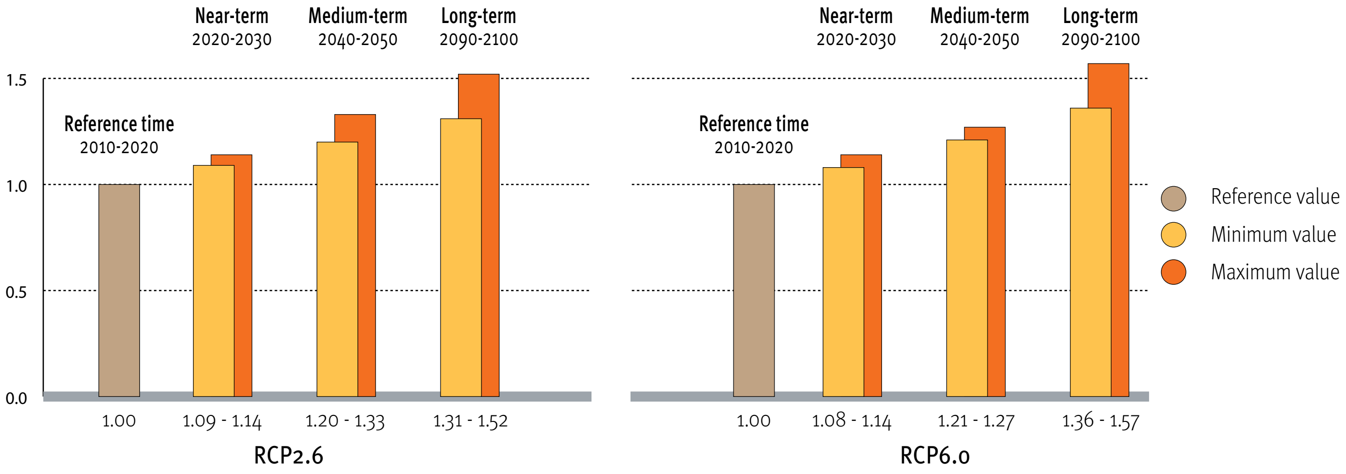


Illustration by Andrew Sullivan/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

Global change in wildfire events



Source: Douglas I. Kelley, UK Centre for Ecology and Hydrology

Kristina Thygesen, GRID-Arendal, 2022

Figure s2. By the end of the century, the likelihood of catastrophic wildfires events will increase by a factor of 1.31 to 1.57. Even under the lowest emissions scenario, we will likely see a significant increase in wildfire events. See appendix for construction.

wildfires (i.e., firefighting and control efforts) when they do occur, relocation of threatened populations and assets during a wildfire, and more long-term sensitive land-use planning that accounts for multiple risks. However, it is practically impossible to entirely remove the risk posed by wildfires. Consequently, more communities around the world must learn to live with the residual risk of wildfire and plan appropriately to minimise the disruption it may cause. {Recommendations 3, 4, 6, 7}

Scientific evidence shows that around the world, fire regimes (the characteristic pattern of fire established over time and space) are changing due to climate change, and land-use and population change. Land-use and population change can both increase and decrease the risk of wildfire. Climate change has led to numerous environmental changes that can increase the frequency and magnitude of dangerous fire weather – increased drought, high air temperatures, low relative humidity, dry lightning, and strong winds, resulting in hotter, drier, and longer fire seasons. The increase in the frequency and magnitude of dangerous weather conditions is causing vegetation that would not usually burn (e.g., rainforests, permafrost, and peatland swamps) to dry out and combust. {Sections 2.1, 2.2, 2.3; Recommendations 1, 2}

Evidence suggests that wildfires will become more frequent in some areas. The impact of climate change on fire behaviour in the future is complex, but current models suggest that some areas, such as the Arctic, are very likely to experience a significant increase in burning by the end of the century. Areas of tropical forest in Indonesia and the southern Amazon are likely to see increased burning if greenhouse gas emissions continue at their current rate. There will also be significant changes in burnt area in landscapes that currently experience burning. This includes tropical savannas and tropical and temperate grasslands, which are predicted to be altered by increased burning in some areas and decreased burning in others. {Section 2.5; Recommendation 1}

Significance of wildfires to the environment and societies

Wildfires can significantly affect the global carbon cycle. Wildfires in ecosystems like peatlands and rainforests, which store large amounts of irrecoverable terrestrial carbon, release vast quantities of CO_2 into the atmosphere, exacerbating global warming. In this way, wildfires may accelerate the positive feedback loop in the carbon cycle, making it more difficult to halt rising temperatures. {Section 2.4; Recommendation 1}

Wildfires can have significant economic impacts on individuals, communities, and nations. The cost of these disasters is often difficult to ascertain but in terms of long-term impact, the world's poorer communities are disproportionately affected. The disadvantages that wildfires bring to these communities could slow progress towards achieving the United Nations Sustainable Development Goals. {Section 3.1; Recommendations 3, 4, 5, 7}

Wildfire smoke contains particulates and toxic combustion products that have been shown to cause respiratory harm and evidence is mounting for detrimental cardiovascular impacts and increased risk of neurological disorders. Sustained exposure to the particulate matter in smoke can be fatal, especially to those with impaired lung function or other pre-existing health problems. Modelling also suggests that exposure to smoke particulates above safe levels can cause chronic impacts that reduce life expectancy and increase pressure on public health systems. {Section 3.2; Recommendations 7, 8}

Wildfires can be devastating to wildlife due to mortality during the fire and, for some animals, post-fire habitat changes. Wildfires can impact vegetation on multiple scales, from landscapes to individual plants. There is evidence that wildfires are pushing some animal and plant species closer to extinction. {Section 4.1; Recommendations 2, 3}

Wildfires can negatively impact water catchments. Contaminants, increased soil erosion, changed soil composition, and slope stability affect both yield and quality for extensive periods. Sensitive ecosystems within water catchments require careful fire management to maintain ecosystem function without negatively impacting catchment performance. {Section 4.2; Recommendations 2, 3}.

Reducing the risks posed by wildfires

When it comes to fighting wildfires, technology has very clear limitations. This is because controlling wildfire behaviour is highly dependent on the prevailing weather and fuel conditions, and accessibility. It is often only a change in weather that can help bring a wildfire under control. Therefore, the limits and appropriateness of suppression strategies and tactics and the associated suppression resource types must be well understood. This can ensure that resources are employed efficiently and effectively when conditions are most suitable and that the risk to firefighters is minimised. {Sections 5.2, 5.3; Recommendations 6, 8, 9}

Managing the available fuel before a wildfire breaks out through planned (prescribed or hazard reduction) burning or other hazard mitigation actions (e.g., physical removal or chemical treatment) can reduce the intensity and thus likely impact of a wildfire. The use of prescribed fire for the management of fuels can effectively reduce wildfire size, fireline intensity, and fire severity. Managed fuels improve the effectiveness of fire suppression efforts, increase firefighter safety, and decrease detrimental impacts on ecosystem services. Traditional knowledge of land management in many regions – particularly the use of fire for fuel management – can also be an effective way of implementing hazard reduction efforts while maintaining ecological values and biodiversity. However, prescribed burning is not without its risks, including decreased air quality and the potential for unintended consequences. It is essential that the effect of prevailing weather and fuel conditions on wildfire behaviour and the resultant efficacy of hazard reduction measures be understood. {Section 5.5; Recommendations 2, 4}

Integrated wildfire management is key to adapting to current and future changes in global wildfire risk. It consists of five interlinked and often overlapping phases: review and analysis, risk reduction, readiness, response, and recovery (the 5Rs) (also known as PPPRR: planning and prevention, preparedness, response, and recovery). Irrespective of the wildfire risk mitigation strategies in any integrated fire management system (including hazard reduction and suppression), residual risks will remain and planning for impacts and recovery continue to be key elements of disaster management. Communities residing in wildfire-prone areas and the local governments that support them must be aware of the risk of wildfire and the potential impact it may have on property and infrastructure. Identifying critical assets that require protection (e.g., hospitals or major transport routes), understanding their level of exposure, determining possible alternatives if these assets are affected during a wildfire event, and identifying evacuation routes and safer places for sheltering, are part of learning to live with fire. Additionally, the collection of human data disaggregated by sex can help understand gender-related differences in risk perception to take into account in community engagement efforts. When communities and governments (at all levels) are informed of the risks of wildfire, including specific threats that may arise during a wildfire event, they are better able to prepare for, respond to, and recover from wildfire. {Chapter 5, Recommendations 3, 7, 9}

Recommendations

1 Recognise and respond to the impact of climate change on the prevalence and behaviour of wildfires

Climate change is increasing the likelihood of fire occurrence in many regions. The most recent Intergovernmental Panel on Climate Change (IPCC) report indicates that weather conducive to wildfires ("fire weather" – hot, dry, and windy) is becoming more frequent in some regions and will continue to increase with higher levels of global warming. Countries must meet and exceed their commitments under the Paris Agreement to reduce global warming and the prevalence and behaviour of wildfires globally. This will, in turn, reduce the social, economic, and ecological impact of wildfires.

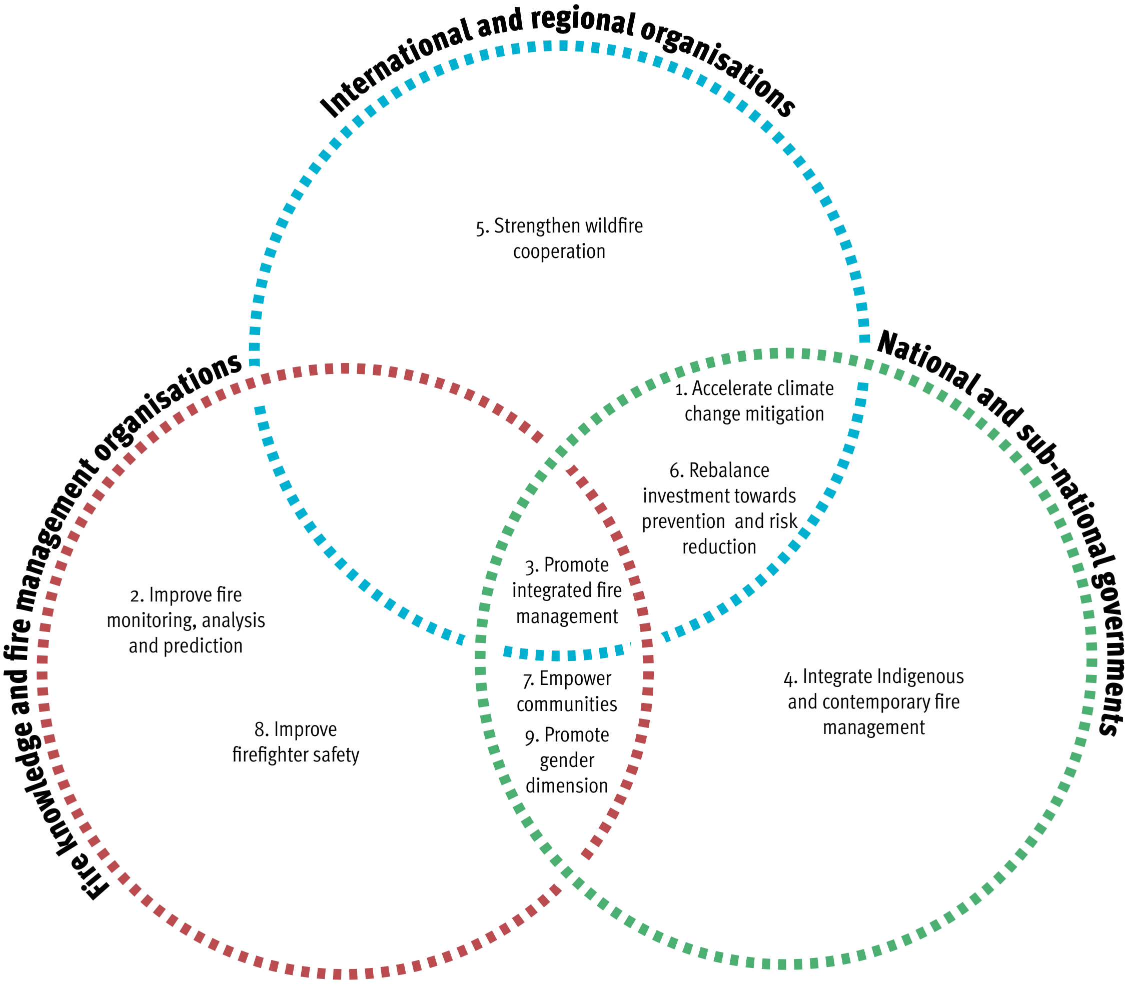


Figure R1. Recommendations arising from this Rapid Response Assessment for international and regional organisations, national and sub-national governments, and fire knowledge and fire management organisations. These recommendations are applicable to all wildfire management contexts across the globe. Jurisdictions with decades of experience in managing wildfires and those rapidly learning to do so are all being challenged. No single jurisdiction has solved the problem of wildfires and lessons can be shared by all as the risk of wildfires changes.

2 Understand wildfire behaviour and improve fuel management and wildfire monitoring

There is a critical need to better understand the behaviour of wildfires in different ecosystems and under a changing climate. This knowledge will support consistent fire data collection and analysis across organisations and countries, thereby improving the management of wildfire fuels, facilitating ignition prevention, and reducing gaps in fire management preparedness and response. Identifying how existing wildfire management practices encourage or discourage harmful wildfires can help improve decision-making and management systems. Improved data collection and analysis will also help monitor changes in fire activity, assess ecosystem response to changing fire regimes, and enhance climate models.

3 Promote an integrated fire management approach

While fire is a natural ecological process, changes to our climate and land-use are contributing to more wildfires. Dealing effectively with the increase in wildfires requires policies and incentives that promote integrated fire management approaches. Achieving and sustaining adaptive land and fire management requires a well-designed and balanced combination of policies, a clear legal framework, and incentives that encourage appropriate land and fire use. These approaches maintain and restore healthy ecosystems while meeting the social, economic, and health needs of human populations.

Review and Analysis in Timor-Leste

Since its independence in 2002, Timor-Leste has made tremendous progress in establishing government institutions essential for running the country's economy. With a population of 1.3 million people, Timor-Leste is a peaceful, democratic nation. However, Timor-Leste remains one of the poorest countries in the Asia-Pacific Region. The 2020 World Risk Report ranked Timor-Leste as the 20th most at-risk country in the world to natural disasters, as result of its location, geography, and very limited capacity to prepare for and recover from climate impacts.

Due to its dependence on low productivity, rainfed agriculture, Timor-Leste is particularly vulnerable to climate-induced hazards. A key driver to vulnerability in Timor-Leste is the predominance of shifting slash-and-burn agriculture practices. In some situations, these fires may escape and, uncontrolled, they may impact negatively upon other farms, settlements, and forested areas.

Responding to a Government request, the Ministry of Agriculture and Fisheries, with technical assistance from FAO, conducted a thematic review on 'fire' in order to support Government efforts in addressing the important issues of slash-and-burn and uncontrolled fire or wildfire. The review aimed to reinforce the national capacity to reduce fire incidence and address the negative impacts of fire in Timor-Leste, and included the following:

- Review of the fire-based land management system practiced in Timor-Leste and analysis of the customary norms related to land management, and identification of the key drivers of fire at community level;
- Analysis of the relevant legal and policy framework;
- Analysis of how the incidence of fire may have changed over time and what factors contributed to these changes; and
- Identification of information gaps for additional research and studies.

Key findings of the review and analysis:

- The rural communities in Timor-Leste are structured and function around their heredity-based social groups (uma lisan) and land is claimed and managed by these uma lisan.
- Slash-and-burn agriculture (also called swidden or shifting cultivation), which has been practiced for at least a millennium, is the dominant land-use system in Timor-Leste.
- With an estimated 123,000 farming households slashing and burning forests, bush, crop residues and grasslands in Timor-Leste, uncontrolled fires from slash-and-burn agriculture are the main contributor to uncontrolled or wildfire events.
- Remote sensing analysis indicates that the annual burned area from 2001 until 2020 is on average 3.3 percent of the national territory (approximately 50,000 hectare), but with significant annual variations (0.6 to 4.7 percent) and a strong indication these estimates underrepresent the reality of fires.
- While the use of fire is currently an important part of the food production system, it is not considered by most community respondents as a sustainable land management practice that will maintain soil fertility for future generations.
- Timor-Leste is equipped with a comprehensive set of adequate laws to conserve and protect the environment, including those related to fire.
- Two categories of drivers of fire have been identified: a) those related to the use of fire with a purpose; and b) those related to uncontrolled fire that are damaging assets.

Governance and Risk Reduction in Portugal

Arising from the tragic fires of 2017 which resulted in 117 fatalities (48 female and 69 male) and over 540,000 ha burnt, the Government of Portugal undertook an ambitious process to develop a new integrated wildland fire management plan with the goal of protecting Portugal from severe wildland fire.

The vision in the resulting 2020-2030 National Plan for Integrated Wildland Fire Management is "a Portugal protected from severe rural fires" with the mission to "protect people and property from rural fires and develop rural land, ensuring ecosystems are properly tended to" (Agency for Integrated Rural Fire Management [AGIF] 2020). The four Strategic Objectives of the National Plan are:

1. Valuing the rural areas – Recognizing rural areas as enablers of wealth and sustainability.
2. Active management of rural areas – Preserving rural areas through the use of fire management practices in line with citizens' well-being and safety.
3. Behaviour change – Promoting the adoption of responsible behaviours for citizen safety and the preservation of a productive and safe territory, reducing ignitions, and improving decision-making processes for individual and collective protection.
4. Efficient risk management – Implementing risk management throughout the whole value chain in order to reduce losses, with clear priorities and effective use of public resources.

The National Plan has the following targets:

1. Design and implement a national strategic programme for large-scale fuel reduction.
2. Burnt areas of more than 500 hectares are less than 0.3% of fires.
3. Reduction of ignitions on high fire danger days.
4. Adding value to biomass by connecting harvesting and processing in rural areas to markets.
5. Build the skills in agencies for effective risk management.

4 Support and integrate Indigenous, traditional, and contemporary fire management practices into policy

Globally, there is growing recognition of the important role that Indigenous and traditional knowledge and experience can play in informing land management practices that assist in the prevention and mitigation of wildfires. Indigenous and traditional knowledge of land management in many regions – particularly the use of fire to manage fuel, including for wildfire mitigation – can be an effective way of reducing hazard. It can also ensure that biodiversity, and cultural (including understanding traditional gender roles that can govern burning activities) and ecological values are respected, as well as create livelihood opportunities. Recognizing and supporting the inclusion of Indigenous and traditional fire knowledge within government policy, practice, and programmes can have multiple benefits (e.g., vegetation management, cultural, spiritual, social, economic, health and well-being benefits, and political-self-determination).

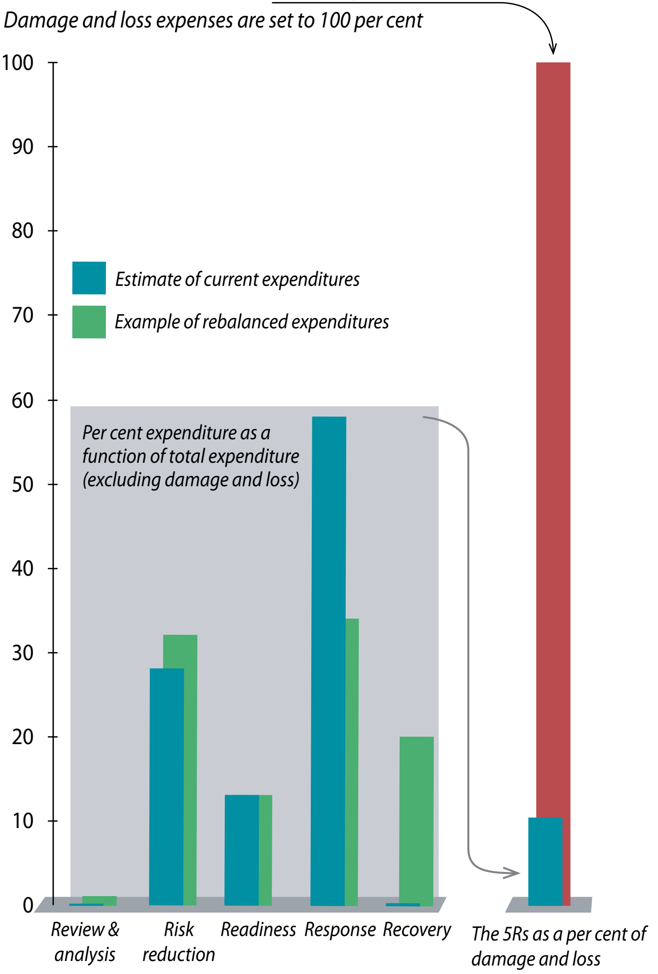
5 Strengthen international and regional cooperation on wildfires

The greatest potential for coherent and consistent improvement in fire management is through continued international interaction and exchange, joint problem solving, and sharing experiences in wildfire management and research. Existing networks and working groups tend to be focused on fire response. These efforts should be encouraged and supported, while expanding their focus to include cooperative work around mitigating fire risk before wildfires occur and building back better following a wildfire. Development of an international standard for wildfire management will facilitate international cooperation and help all wildfire-prone countries build capacity for both domestic application and international assistance.

6 Rebalance investments spent on reactive suppression to proactive wildfire mitigation and management

Wildfires become uncontainable when they exceed the limits of suppression. Given the current limitations of fire suppression and a future predicted to have longer fire seasons and more severe fires due to increasingly worse fire weather conditions, making targeted investments in preparedness measures now will yield significant benefits (Figure R2). Wildfire risk reduction activities represent a sound return on investment as they reduce the potential impacts of wildfires. In the long term, they will be more cost effective than relying on reactive firefighting and post-disaster recovery efforts. Auxiliary risk management strategies should also be in place to reduce the likelihood of adverse fire impacts arising from the predicted increase in extreme fire weather.

The 5Rs compared to damage and loss



Illustrated by Kristina Thygesen GRID-Arendal

Source: Thomas et al. 2015

Figure R2. Illustration of costs associated with wildfire management – the 5Rs – Review and analysis, Risk reduction, Readiness, Response, and Recovery. The estimate of current expenditures across the 5Rs (blue) comes from U.S. data (Thomas et al. 2015) but is considered likely to closely represent spending in developed states. Currently, there is very little spending on review and analysis (critical in determining the effectiveness of investment), a disproportionate amount on response, and very little on recovery. The suggested expenditures (green) represent a preliminary attempt to rebalance investments in a way that could reduce damage and loss (red), which currently greatly exceeds all spending on the 5Rs.

7 Empower communities and local authorities

Enabling communities and local authorities in wildfire-prone areas to understand and accept the residual risk of wildfires will strengthen coordination of key stakeholders and build capacity to prepare for, respond to, and recover from wildfires. Activities include risk reduction (at the dwelling, locally, and regionally), infrastructure hardening, evacuation planning, air quality alerts, and social and infrastructure recovery and rebuilding. Key stakeholders need to be involved throughout the fire management process. This includes involving women and men from local communities so that local needs, concerns, and potential barriers to action can be addressed, and a common understanding and long-term vision for how to live with fire is developed, shared, understood, and acted upon.

8 Improve firefighter safety

While firefighting is an essential component of fire management at all scales, the safety and long-term health of firefighters is paramount. The risk of harm to both female and male firefighters, before, during, and after operations must be minimised. Fire management bodies must take measures to ensure safe work practices in all aspects of firefighting, ensuring that they understand and reduce the risks of smoke inhalation, minimise the potential for life-threatening entrapments (i.e., burn-overs), and provide firefighters with access to adequate hydration, nutrition, rest, and recovery between shifts. In many instances, internationally agreed standards for assuring effectiveness of firefighting efforts may also act to minimise the exposure of firefighters to life-threatening situations.

9 Promote the collection of data and information on the gender dimension of wildfires

Available research indicates that women and men have different approaches to wildfires, including risk perception and decision making. The collection of sex-disaggregated data will help to identify patterns for further analysis, including national, regional, or global trends. Understanding gendered risk perceptions can help policymakers develop more effective and robust approaches to wildfire management and improve safety for all members of society. Improving gendered knowledge extends to helping firefighting become a more inclusive activity. Women firefighters face various challenges ranging from gender discrimination and sexual harassment to ill-designed equipment and protective clothing that puts them at greater risk of injury.

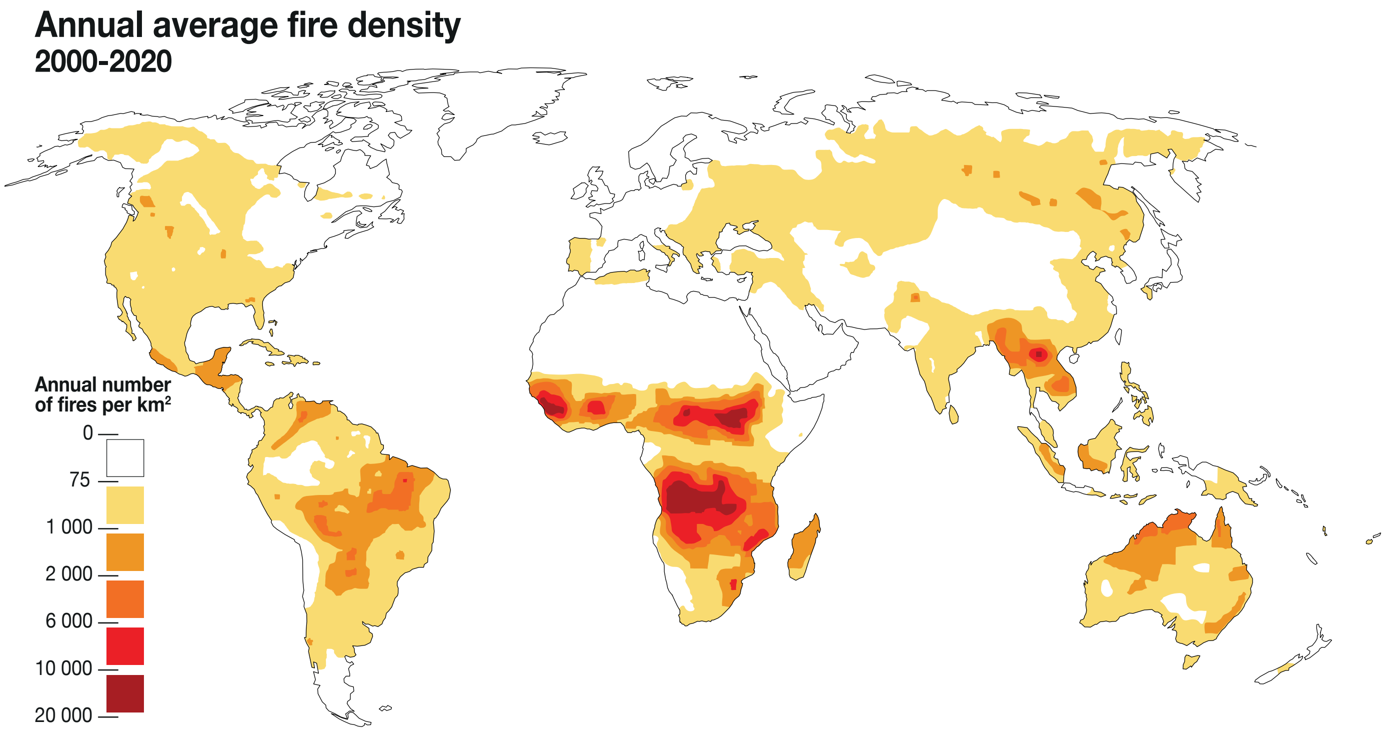
Chapter 1 – Our planet on fire

> ***Wildfire:*** *A wildfire is an unusual or extraordinary free-burning vegetation fire that poses significant risk to social, economic, or environmental values. It may be started maliciously, accidently, or through natural means.*
>
> *A wildfire can be short in duration and small in area but more commonly burns for an extended period and over a wide area. The behaviour of a wildfire can be largely benign around its perimeter but will sometimes be characterised by periods of rapid spread and intense behaviour at its front against which suppression and other risk mitigation efforts may be ineffective. The impacts of a wildfire may be immediately and directly apparent or may materialise sometime after the fire is extinguished.*

1.1 Introduction

Free-burning landscape fire is an important natural phenomenon critical to the healthy functioning of many ecosystems. It is an important land management tool, culturally, economically, and ecologically. Therefore, not all vegetation fires are unwanted. This Rapid Response Assessment (RRA) focuses on the apparent increase in the occurrence, extent, duration, and consequences of *wildfires* – unusual or extraordinary free-burning vegetation fires. These are the fires that can destroy habitats, threaten species, impair ecosystem services, endanger human health, lives and livelihoods, damage national economies, and release significant amounts of particulate matter and greenhouse gases into the atmosphere.

The RRA considers the changing patterns of wildfires around the world, their impact on people and the environment, and options for mitigating risk and improving our response. The most recent Intergovernmental Panel on Climate Change



Map by Miguel Castillo Soto/University of Chile, 2021.
Source: NASA 2020

GRID-Arendal/Studio Atlantis, 2021

Figure 1.1. Almost every vegetation region of the world has a free-burning fire at some time of the year. This figure shows the annual concentration of all vegetation fires (landscape fires and wildfires) observed per square kilometre (km²) for the period 2000–2020.

(IPCC) report indicates that weather conducive to wildfires (hot, dry, and windy) has become more frequent in some regions and will continue to increase with higher levels of global warming (IPCC 2021). Improved understanding of wildfire disaster risks can help governments develop effective policies and emergency measures. Reducing global wildfire risk is a necessary component to achieving the United Nations' 2030 Agenda for Sustainable Development (the commitment to eradicate poverty and achieve sustainable development worldwide by 2030), the objectives of the Sendai Framework for Disaster Risk Reduction 2015–2030 (to substantially reduce disaster risk and losses in lives, livelihoods, health, and productive assets by 2030), and the aims of the United Nations Decade on Ecosystem Restoration 2021–2030 (to prevent, halt and reverse the degradation of ecosystems worldwide). Regardless of mitigation efforts, humans still need to learn to live with and manage the threats from wildfire.

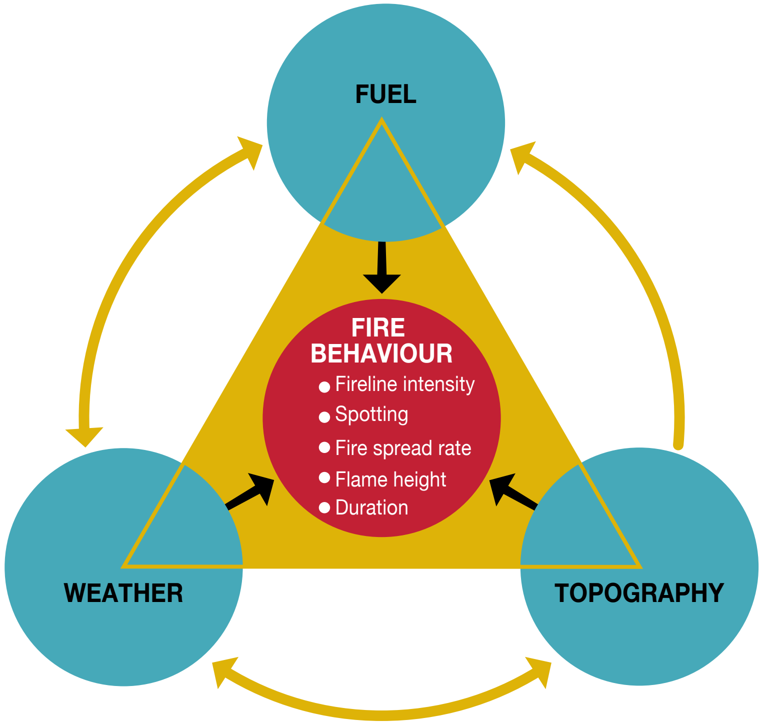
Vegetation fires can occur at any time, in nearly every type of landscape, around the world (Figure 1.1). While such fires can be destructive, over hundreds of millions of years they have also played a crucial role in evolution, shaping the global distribution of vegetation, and sustaining biodiversity (Bond et al. 2005). Humans learned to control fire between 300,000 and 400,000 years ago (James et al. 1989; Roebroeks and Villa 2011). Before this, fires were ignited almost exclusively by lightning and sometimes by volcanic activity. Control of fire was a major step in the technological evolution of human society (Oakley 1956; Levi-Strauss 1969).

In addition to naturally occurring fires, people light vegetation fires for a variety of reasons, including clearing land for agriculture, cleaning up crop and forestry residue, hunting, conducting warfare, and stimulating forage for grazing. Free-burning fire has long been a cultural and land management tool, especially for Indigenous peoples. However, sometimes fires escape control due to human errors or unexpected changes in the weather. Fires are also often lit maliciously and intentionally left to spread. Whether caused by humans or nature, when fires burn out of control, they can become wildfires.

1.2 Describing fires

There is no universally agreed-upon terminology to describe fires, but they are often categorised in terms of the vegetation or "fuel" type in which they burn (e.g., forest, shrub, grass, peat, etc.), their behaviour (see Figure 1.2) and the severity of their impacts. Very often the more generic "wildland fire" is used. In some regions, "forest fire" is used for any landscape vegetation fire regardless of whether it is burning in a forest or not. In Australia, "bushfire" is common, where "bush" is a colloquial term for a landscape outside of urban locations.

Fire behaviour triangle



Source: Countryman, 1966.

GRID-Arendal/Studio Atlantis, 2021

Figure 1.2. The fire behaviour triangle illustrates the key variables that affect how a wildfire behaves (Countryman 1966). Anything that burns is fuel, mainly live and dead vegetative material. Weather influences fire through the effects of wind, air temperature, precipitation, and relative humidity. Topography can directly influence the speed of fires and the type and condition of the fuel. Together, these variables determine the behaviour (speed, direction, and flame characteristics), and intensity of a fire.

The fire behaviour triangle (Figure 1.2) illustrates the major factors that influence fire behaviour. Anything that burns is “fuel”. In a wildfire, this is the live and dead vegetative material available to combust. However, the fuel that directly influences the behaviour and spread of a wildfire is predominantly fine fuel largely comprised of matter such as leaves, bark, twigs, shrubs, and grasses, with a diameter of less than 6 mm for dead fuels and less than 3 mm for live fuels (McArthur 1967; Sullivan et al. 2012). Flammability is influenced by factors such as

Factors and conditions influencing wildfire occurrence

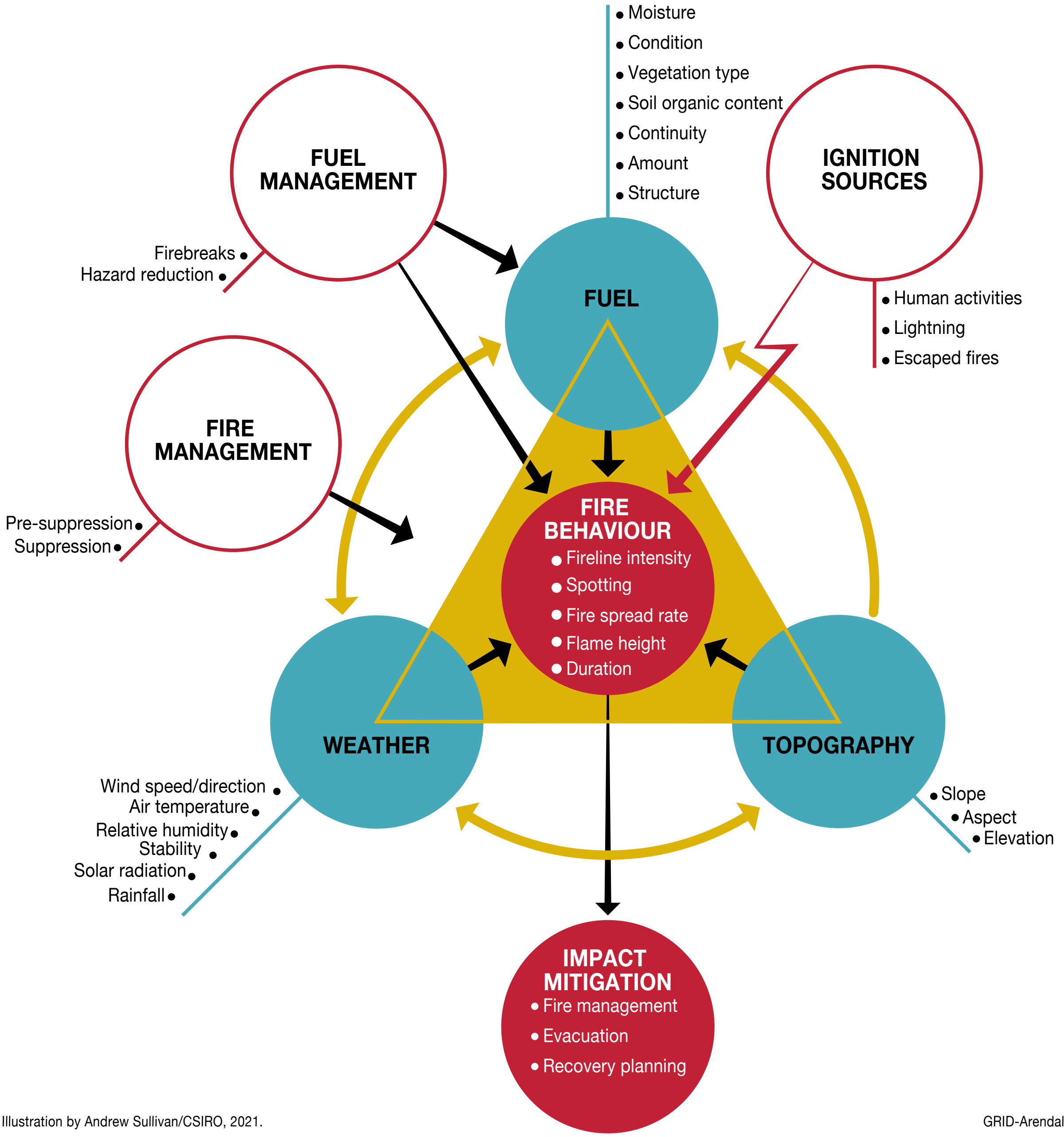


Illustration by Andrew Sullivan/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

Figure 1.3. A wildfire is the result of a complex interaction of biological, meteorological, physical, and social factors that influence its likelihood, behaviour, duration, extent, and outcome (i.e., its severity or impact). Changes in many of these factors are increasing the risk of wildfire globally (e.g., climate change is increasing the frequency and severity of weather conducive to wildfire outbreaks, changed demographics in high-risk regions are increasing the potential impacts of wildfires).

chemical composition, structure and arrangement, spatial continuity, and the density of fuel. Fuels other than fine live and dead vegetation may combust in a wildfire but these are generally consumed well behind the fire front (Hollis et al. 2011). While they may contribute significantly to the fire's gaseous and particulate emissions (Surawski et al. 2016), they do not influence its behaviour. These fuels include larger vegetative material such as downed woody debris (i.e., branches and logs), standing trees and woody shrubs, and organic soils, as well as fabricated materials that may have been discarded or abandoned in the landscape which might include synthetics and other volatile materials. ***Weather*** influences fire through changes in atmospheric stability, wind (speed and direction), air

Fire intensity and fire severity

Fire intensity describes the rate of energy released from the combustion of biomass consumed in a fire (Keeley 2009; Sullivan 2017). This energy release appears almost exclusively as radiation and convection from the combustion zone of a fire. The most common measure of fire intensity in vegetation fires is "fireline intensity" (Byram 1959), the total energy released in the flaming region per lineal metre of fire front (kW m^{-1}). It is calculated as the product of the total amount of fuel consumed (kg m^{-2}), the heat yield of the fuel (kJ kg^{-1}), and the speed of propagation of the fire (m s^{-1}).

While fireline intensity cannot be directly measured, calculated values indicate a correlation between fuel types and fire behaviour, such as flame length, flame height, spotting distance (the distance to which burning debris is transported ahead of a fire front), and fire radiative power (e.g., Byram 1959; Burrows 1994; Wooster et al. 2005). An estimate of fireline intensity can be used to gauge suppression potential. However, relying on fireline intensity values can be problematic, as the fire behaviour and suppression potential at a given fireline intensity varies between fuel types due to differences in heat release rate. For example, the behaviour and spread of a 10 MW m^{-1} grassland fire is very different in terms of fire behaviour from a 10 MW m^{-1} forest fire. This is because the fireline intensity does not consider the dynamic rate at which the energy is released. A kilogram of dry biomass has sufficient energy to power a 100 W light bulb for 50 hours. This energy can be released very quickly (in minutes or even

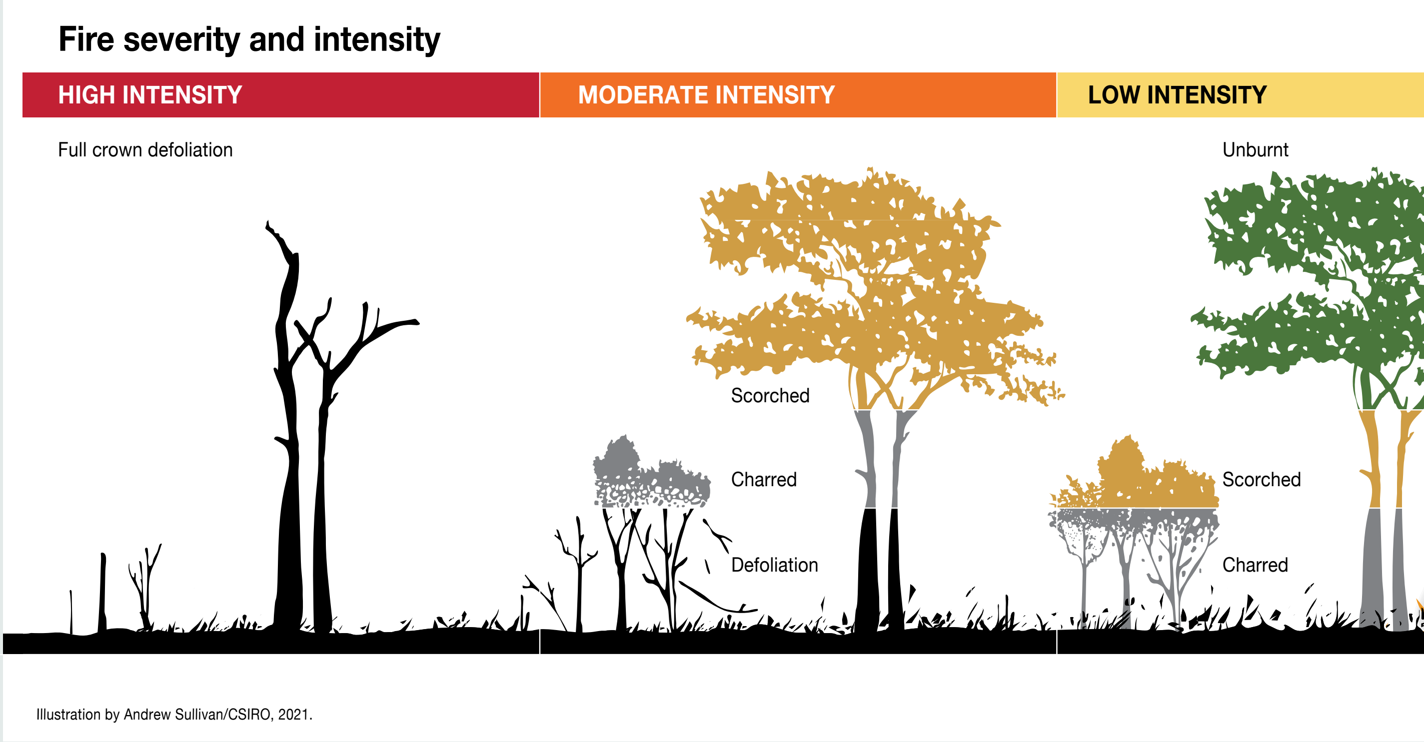


Illustration by Andrew Sullivan/CSIRO, 2021.

Figure 1.4. The severity of a fire depends upon the intensity of the fire and the nature of the vegetation through which it is burning. High-intensity fire will generally result in high severity (full defoliation) of even moderately tall trees. As a fire's intensity decreases it may result in high severity for shorter trees but lesser severity in taller trees

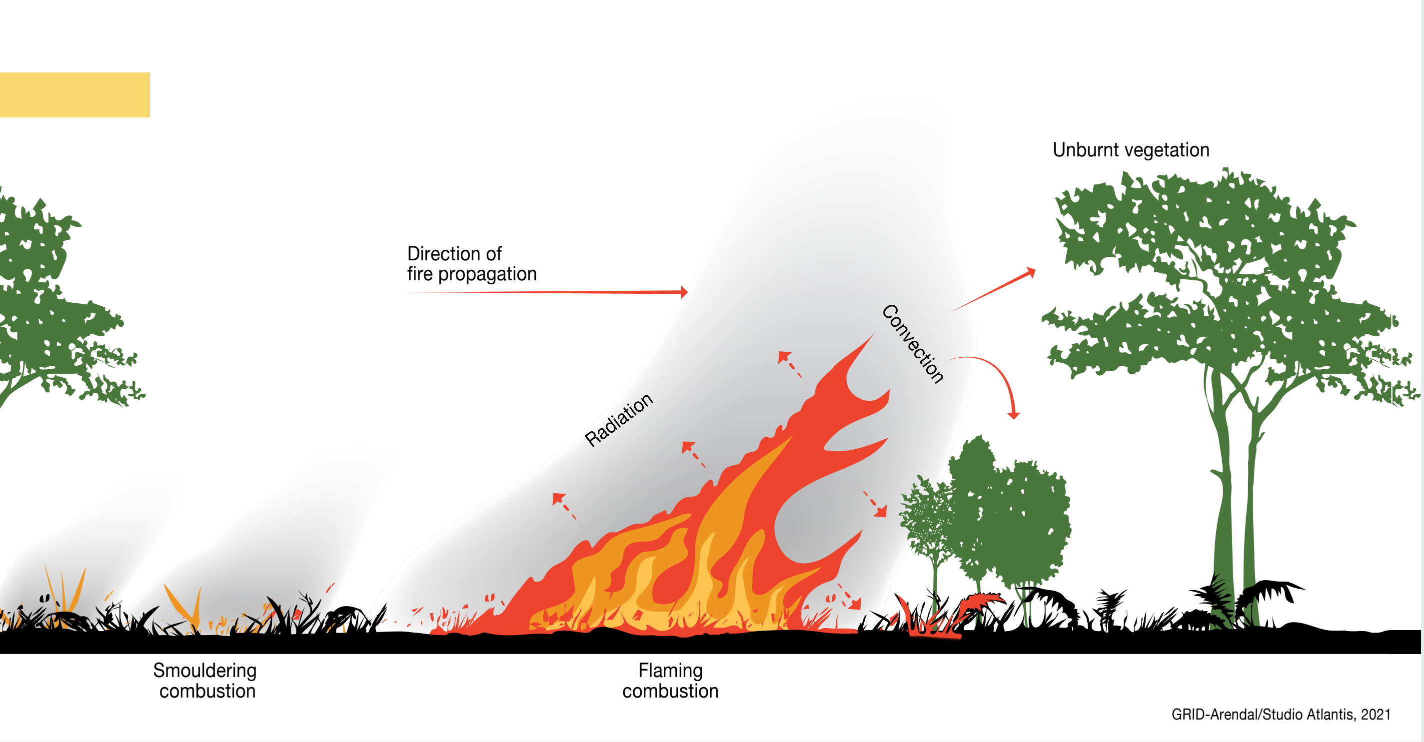
temperature, precipitation, and relative humidity which can modify the combustibility of the fuel (e.g., moisture content) and the rate of transfer of heat from flames to adjacent fuel and therefore the rate of spread, intensity, and size of a fire. ***Topography*** (including slope steepness, elevation, and aspect) can directly influence the speed of fires (fires generally spread faster uphill than down) and the type and condition of the fuel by creating microclimates with localised moisture and growth conditions. Together, these variables control the behaviour (speed, direction, and flame characteristics) and intensity of a fire. Factors such as the type of vegetation through which a fire burns, its duration, and any attempts at control, determine the severity and extent of the fire.

seconds, as is common in grass fires) or slowly (over hours or days). The rate varies between fuel types, fire spread modes and combustion conditions.

Fire severity (and the related term burn severity) describes the immediate and direct impact of a fire on an ecosystem, both above and below ground. It is driven to a great extent by fire intensity (Figure 1.4). However, it depends on the conjunction of several factors, including season of burn, species present, their adaptations to fire and their health at the time of burning, and weather conditions. Descriptions of fire severity in an ecosystem can include the degree of scorch (i.e., leaf death) or consumption of vegetation strata, the mortality of trees, and biodiversity loss (Keeley et al. 2008).

Understanding fire severity and the factors that influence it is a critical requirement in post-fire assessment and ecosystem management. However, the methods used to categorise fire severity are highly variable and are often developed for specific purposes in specific ecosystems. The lack of specific criteria in many methods limits their application and the ability to infer the behaviour of the fire after the fact.

Earth observation remote sensing can be used to estimate levels of fire severity (e.g., Moderate Resolution Imaging Spectroradiometer [MODIS], Landsat, and Meteosat products) determining the difference in vegetation cover and soil condition before and after the fire (Roy et al. 2006).



GRID-Arendal/Studio Atlantis, 2021

While fuel, weather, and topography influence the behaviour of a wildfire, several other factors influence the likelihood of a wildfire, its severity, duration, and extent (Figure 1.3). The likelihood of a wildfire occurring is determined by the number and density of successful ignition sources, such as dry lightning strikes, human activities likely to cause an ignition (e.g., grinding steel or harvesting), or existing fires that escape control. Management of fuels, including fuel hazard reduction and creation of firebreaks within continuous fuels aid fire management actions, restrict fire spread, and reduce the potential for fires to become large uncontained conflagrations. Fire management more generally, including pre-suppression prior to the outbreak of a fire (such as raking around high-risk ignition locations and application of suppressants to fuels) and active suppression once a fire does break out, can also reduce potential for widespread propagation of a fire. Lastly, once a fire does begin to threaten assets, steps can be taken to mitigate the potential impact of the wildfire. These include targeted fire management (i.e., asset protection), evacuation of affected populations, and commencement of post-fire recovery planning to minimise the period during which populations are impacted and infrastructure is compromised.

Where fire is an important ecological component of an ecosystem, and where some broad level of stability exists in the severity, spatial and temporal occurrence, and impact of fire, fire ecologists describe a characteristic “fire regime”. The components of a fire regime include the types of ignition sources, frequency, intensity (energy output), severity (the effect of fire on the ecosystem), extent, seasonality, and heterogeneity (patchiness). While the fire behaviour triangle is useful in understanding drivers of behaviour of a single fire, fire regimes are characterised over broader scales of space (ecosystem or landscape) and time (decades to centuries). Factors that influence the frequency, duration, extent, and severity of fire in a fire regime include vegetation type, structure, and continuity (i.e., productivity, flammability, and distribution), climate and seasonal flammability (i.e., moisture content, length and severity of the dry season, extreme wind patterns), and ignition sources (i.e., seasonality, density, location, and timing). Understanding the factors that can alter the fire regime of an ecosystem can help determine potential changes in fire behaviour and thus the likely impact of fire upon an ecosystem over a given period.

1.3 Fires as ecological disturbance

An ecological disturbance was originally defined as a rare ecologically destructive event, where “destructive” meant that it killed or removed biomass (Grime 1979) but has been expanded to include discrete normal events that are part of natural ecosystem dynamics (e.g., Rykiel 1985). Therefore, ecological disturbances include storms, floods, landslides, volcanic eruptions, herbivory, and fire, to name a few. These events can alter the structure of ecological systems, the availability of resources, and the physical environment (Pickett and White eds. 1985).

The way a disturbance functions in a given ecosystem is described by the ecosystem's "disturbance regime" (Pickett and White eds. 1985). In the case of fire, it can influence – both negatively and positively – nutrient cycling and energy flow, decomposition rates, ground and surface water hydrology, carbon sequestration and storage, soil moisture and temperature, ecosystem composition and structure, biodiversity, plant regeneration, plant and animal habitat, pollination, seed dispersal, and ecosystem succession (Booysen and Tainton eds. 1984; Wright and Heinselman 2014; Van Wagtendonk et al. eds. 2018).

One of the most important impacts that humans have on natural ecosystems is their tendency to alter disturbance regimes (Bowman et al. 2011). Humans dam streams and change flow regimes; they introduce livestock and change grazing regimes; they build infrastructure in steep terrain and change soil erosion regimes; and they increase or decrease ignitions and/or manage vegetation and thereby change fire regimes. Humans also alter disturbance regimes at a global scale by modifying the world's climate (Dale et al. 2001). Climate warming changes the frequency of extreme weather conditions that drive the occurrence and spread of wildfires as well as the production and drying of fuels that influence the availability of fuel for combustion (IPCC 2021). The disturbance caused by wildfires can result in a large range of destructive ecological and social impacts (Figure 1.5).

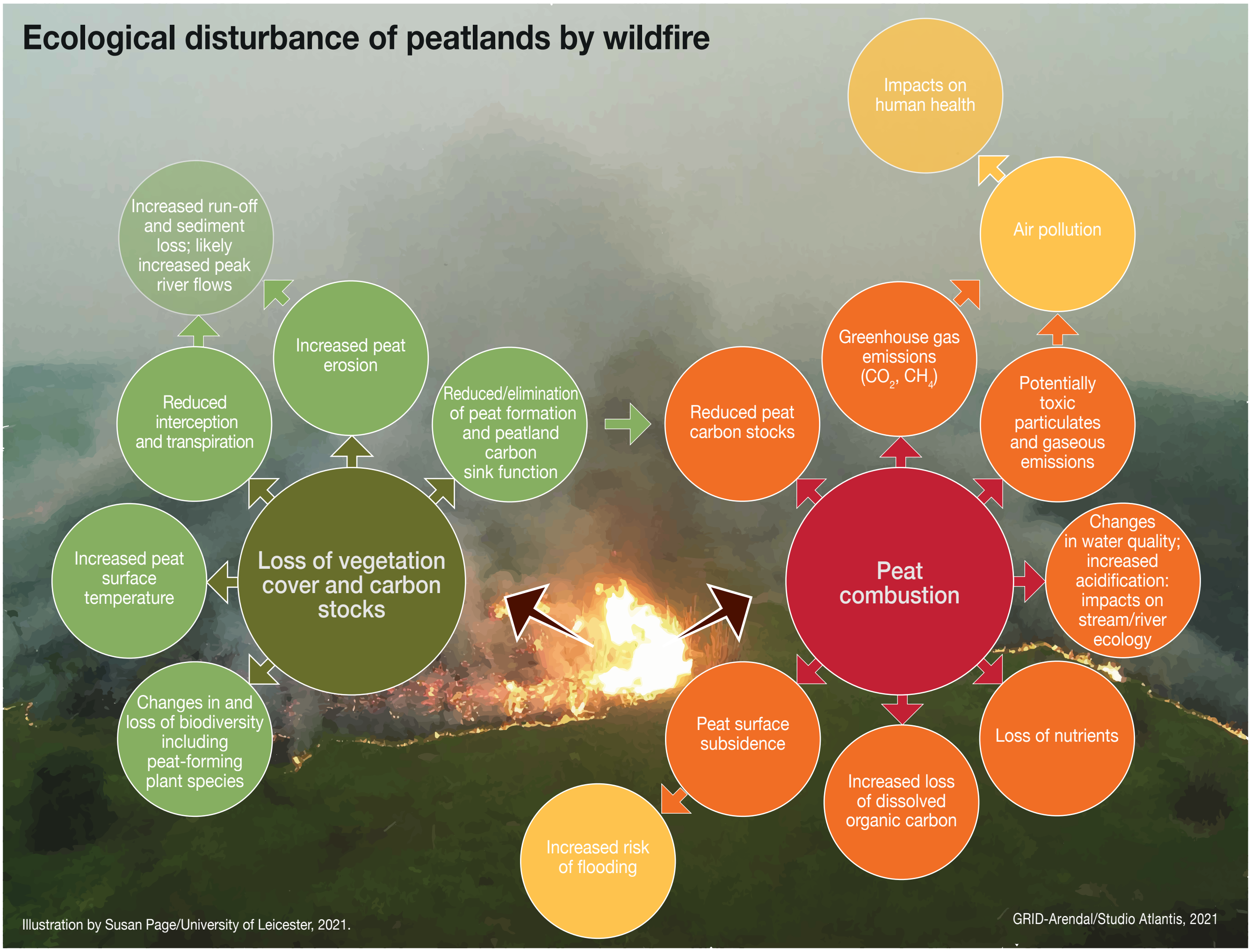


Illustration by Susan Page/University of Leicester, 2021.

GRID-Arendal/Studio Atlantis, 2021

Figure 1.5. An example of the ecological disturbance that can result from wildfires on peatlands. Peatlands contribute significantly to carbon sequestration and storage, biodiversity conservation, water regime and quality regulation, and the provision of other ecosystem services. Climate and land-use change increases the vulnerability of peatland ecosystems to fire, which are particularly difficult to extinguish, and have a range of ecological, hydrological, and social impacts.

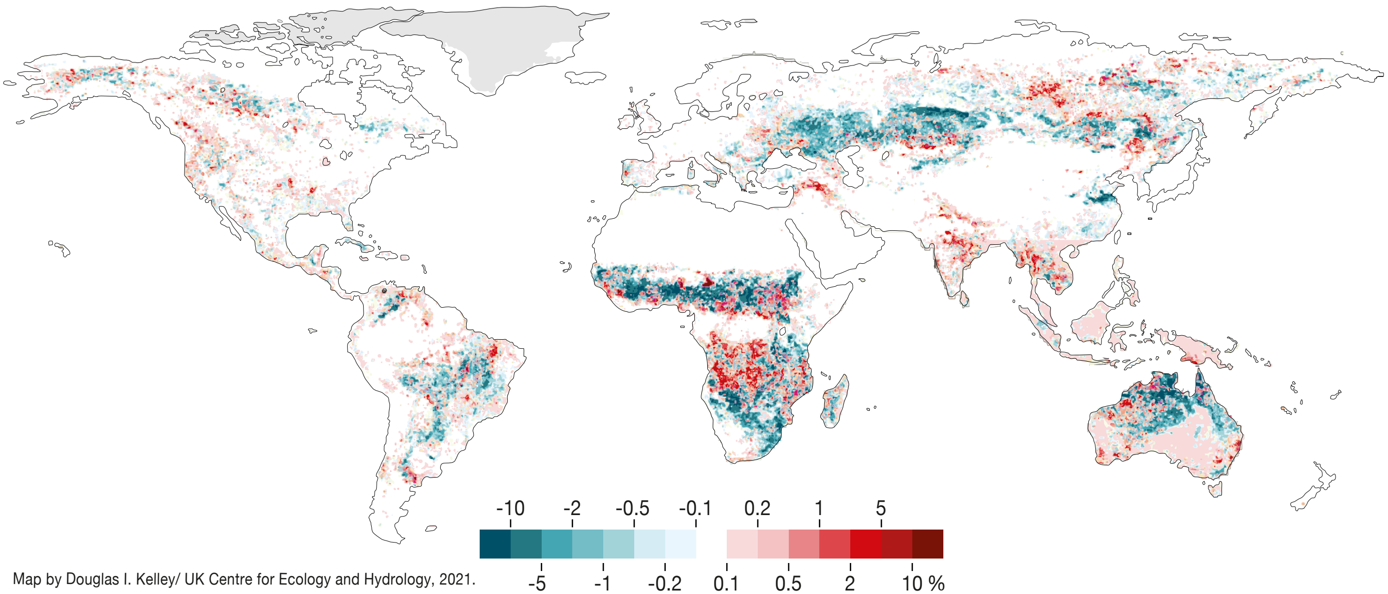
1.4 Global wildfire occurrence and distribution

As vegetation fire is endemic to many parts of the world, wildfires are those beyond the common fire occurrence and impact, and thus likely to be a cause for concern. The increased prevalence of wildfires globally, quantified here as the difference between the long-term average annual area burnt by fire and the average annual area burnt over the last five years, is shown in Figure 1.6. In this figure, as compared with Figure 1.1, many of the regions traditionally associated with frequent fire show a decrease in burnt area over the last five years (e.g., areas in sub-Saharan Africa and northern Australia), while fires in some regions previously not considered fire-prone have increased (for example, northern India, Russia, and Tibet).

Over the last decade, it appears that more wildfires are occurring, not only in regions where seasonal fires are common, but also in areas where fires do not normally occur. For example, eastern Australia and the west coast of the United States of America (USA) generally experience frequent summer fires, but the 2019–2020 fire season saw record-breaking numbers and extent of wildfires in these regions. The Arctic and the Amazon, however – areas not generally prone to extensive wildfires – experienced record-breaking blazes in recent years. A common factor in these fire events is the persistent hot, dry, and windy conditions such as those that occurred around the world in 2019–2020 – the year 2020 tied with 2016 (which was helped by El Niño driving global temperatures) as the hottest year in recorded history (Voosen 2021). As human-induced global warming increases so does the frequency and intensity of the weather conditions conducive to wildfires (Jones et al. 2020). When combined with increases in other factors such as number of ignition sources and high levels of available fuel, the threat of wildfires becomes extreme.

From the combination of factors that govern the behaviour of a wildfire (Figure 1.2) and the conditions necessary for a wildfire to occur (such as ignitions; Figure 1.3), it is possible to determine the dominant factors for wildfire activity as evidenced by burnt area (Figure 1.7). In many regions of the world, the presence of sufficient available fuel to carry fire is the dominant factor (the green areas in Figure 1.7), whereas in others it is sufficient sources of successful ignition (the red areas in Figure 1.7). In some tropical and subtropical zones, the dominant factor influencing burnt area is fuel moisture, meaning that while fuel and ignitions may be sufficient, fuels are often not dry enough to combust. Curtailment of fires through fire prevention, active suppression, or land-use fragmentation can be the controlling factor in developed regions.

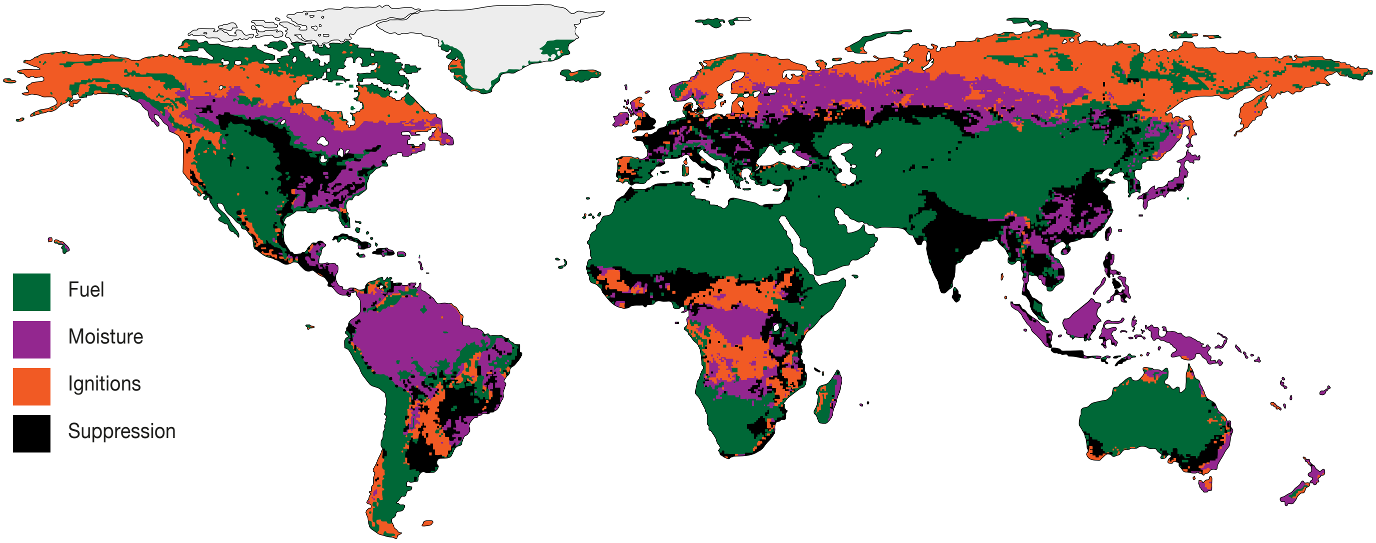
Prevalence of recent global wildfire activity, 2014-2019



Map by Douglas I. Kelley/ UK Centre for Ecology and Hydrology, 2021.
Source: MODIS Fire_cci Burnt Area product; Chuvieco et al. 2018.

GRID-Arendal/Studio Atlantis, 2021

Figure 1.6. An indication of the prevalence of recent global wildfire activity, calculated as the normalised difference between the annual average burnt area from January 2014 to December 2019 (the last five years of the MODIS record at the time of writing) and the long-term annual average burnt area (January 2001 to December 2019; based on the MODIS Fire_cci Burnt Area product; Chuvieco et al. 2018). In many regions, the return interval of wildfires may be longer than this illustrative period and many wildfire events may not be detectable at this scale. The red areas represent regions that have essentially increased in fire activity in the last five years compared to the long-term average, while the blue areas represent those where fire activity has decreased.



Dominant factors contributing to burnt area

Source: Kelley et al. 2019.

GRID-Arendal/Studio Atlantis, 2021

Figure 1.7. The dominant factors contributing to burnt area during the height of the fire season. The strength of each control is measured as the sensitivity of, or marginal change in, the burnt area to either sufficient fuel, dry-enough fuel moisture, sufficient successful ignitions, or effective suppression by the local population and land-use fragmentation. The height of the fire season is defined as the month of the year which, on average, experiences the highest burnt area.

Chapter 2 – The changing pattern of wildfires

Current evidence points to a dramatic shift in fire regimes worldwide. This is driven by a combination of land-use change and climate change (Andela et al. 2017; Forkel et al. 2019; Kelley et al. 2019; Bowman et al. 2020a), with potentially widespread Earth system impacts on humans, vegetation dynamics, atmospheric composition and radiative forcing (Archibald et al. 2018), and even ocean biogeochemistry and ice melt (Bowman et al. 2009).

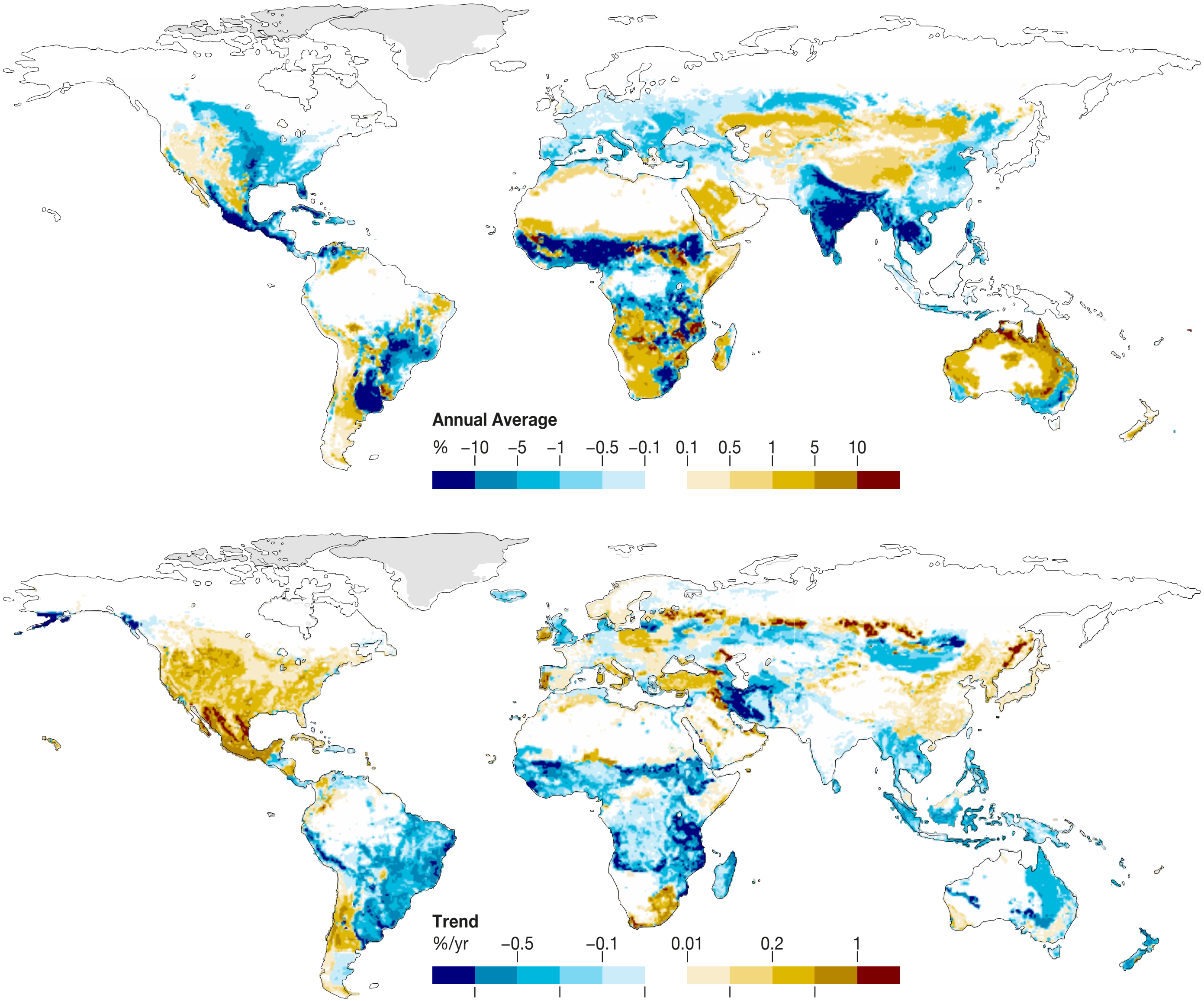
2.1 Impact of human-induced landscape change on global fires

How humans adapt and manage the land is one of the biggest influences on global fire regimes. Anthropogenic land-use change generally refers to converting land, often forest, for agricultural use – i.e., crops, pasture, or rangeland. Land-use change can also refer to afforestation (e.g., establishing plantations on former agricultural land, replanting with different species, or rewilding with nonextant species). Locally, the impact of land-use change alters the dominant vegetation and fire dynamics.

Land-use change can act as a source of wildfire ignition (Aragão et al. 2008) where, for example, people use fire to clear forests or manage agriculturally productive areas (see section 2.4.2 on the Amazon) or temporarily increase fuel loads (for example, the build-up of forest debris after logging). Land-use change can also increase landscape fragmentation, with different impacts in different biomes (Andela et al. 2017). In dense forest biomes, fragmentation can introduce higher flammability and increase the number of ignition points (Armenteras et al. 2017; Silva et al. 2018). Conversely, in savanna areas with sparse vegetation, fragmented connectivity between fuels inhibits fire spread which can limit fire size (Figure 2.1). Increased fragmentation in the savanna, particularly in the Sahel in North Africa, which experiences widespread annual burning, is the primary driver of the substantial and sustained reduction in global burnt area over the twentieth and twenty-first century (Marlon et al. 2008; Andela et al. 2017; Forkel et al. 2019; Kelley et al. 2019). This reduced burnt area can alter vegetation assemblages and carbon uptake.

Agricultural intensification can also result in fewer wildfires due to a reduction in available fuel (e.g., from animal grazing) or increased active suppression (Bistinas et al. 2014; Knorr et al. 2014; Andela et al. 2017). In many locations, including large parts of India, south-east China, temperate Europe, the American Midwest and South America, extensive agriculture

has almost wholly inhibited wildfire (as opposed to fires lit for land management purposes) (Figure 2.1). However, land-use change such as abandonment and reforestation can drive an increase in burning, as seen throughout North America and Europe, and in tropical forests (Armenteras et al. 2013; Kauppi et al. 2018; Moreira et al. 2020; Nikonovas et al. 2020). Land-use change can also have a substantial impact on the regional climate. Deforestation has been shown to reduce evapotranspiration (Cochrane and Laurance 2008) and cloud cover (Jimenez et al. 2018) and decrease precipitation (Bagley et al. 2014). The resultant drying can increase fire conditions (Brando et al. 2014; Castello and Macedo 2016).



Direct impacts of human ignitions, active suppression, and land fragmentation from agriculture on burnt areas, 2001–2014

Source: Kelley et al. 2019.

GRID-Arendal/Studio Atlantis, 2021

Figure 2.1. The direct impacts that the combined effects of human ignitions, active suppression, and land fragmentation from agriculture had on burnt areas between 2001–2014 (from Kelley et al. 2019). Top: The annual average change in the land area burnt compared with a reconstructed burnt area with the same fuel, moisture, and natural ignitions amounts but without active human ignition sources, fire suppression, and land fragmentation. Bottom: the yearly change in burnt area due to these human impacts.

Case study: Land-use change and fire feedback loops in Cambodia

Tonlé Sap in Cambodia is the largest lake in Southeast Asia. Every year during the wet season, it floods and is then partially drained by seasonal "pulses" which occur when the flow of water between the lake and the Mekong River changes direction. During the wet season, many of the lake's 200 fish species move into the adjacent, newly flooded forests and grasslands to breed and spawn. As a result, this ecosystem supports one of the world's most productive inland fisheries and is Cambodia's primary protein source. Fishing is the only source of income for more than 100,000 people living in floating houses on the lake. The lake itself is also home to dozens of globally threatened species (Davidson ed. 2006) and is Southeast Asia's largest waterbird colony. However, like many of the world's largest lakes, it is rapidly shrinking due to climate change, upstream damming, and diversion of tributaries for crop irrigation. A significant fire feedback loop is compounding this situation.

As the lake shrinks, people are burning formerly flooded forest to clear the land for rice farming (Mahood et al. 2020). This is the main driver of fires in the Tonlé Sap Biosphere Reserve (TSBR). Meanwhile, Cambodia's climate is warming, especially during the hottest, driest months (March–May). In a feedback loop, the clearing of seasonally flooded forests results in an even warmer and drier climate, leading to more intense and frequent fires and further tree cover loss, as observed in other tropical systems (Nepstad et al. 2001; Hoffman et al. 2003).

Between 2008 and 2018, approximately 2,800 km^2 of seasonally flooded habitat in the TSBR was lost to expanded dry-season rice cultivation (Mahood et al. 2020; Figure 2.2). The water that rice farmers use for irrigation no longer flows into the peatland swamp forests where it would have supported fish, fishers, and wildlife, and soaked the peaty soil to protect against forest fires (Turetsky et al. 2015). In 2019 and 2020, traditional fishing grounds remained dry, where in previous years the floodplain forests were inundated. Sedge beds extended up to 3 km from the lakeshore into what had previously been open water. As a result, fires increasingly burn out of control, destroying large areas of flooded forest that are then converted to agricultural land, continuing the cycle. These trends could lead to the complete loss of the Tonlé Sap Lake, with catastrophic economic, political, and biological impacts.

Without adequate and appropriate control, forest fires in the TSBR, Southeast Asia and the Asia-Pacific region will continue to adversely impact health and livelihoods, destroy biodiversity, and contribute to climate change. In this part of the world people cause most ignitions, so a technology-centred approach to addressing forest fires will fail. To control fires, governments need to be proactive. Instead playing catchup, they should define an acceptable level of burning, where and when burning can occur, and the level of acceptable risk. An approach that involves community-based fire management which operates locally, drawing on assessments of social, economic, cultural, and ecological conditions can be employed to minimise damage and maximise fire benefits (Ganz 2020). This more proactive approach can complement the efforts of government and build an effective partnership in forest management and protection (see more on community engagement in chapter 5).

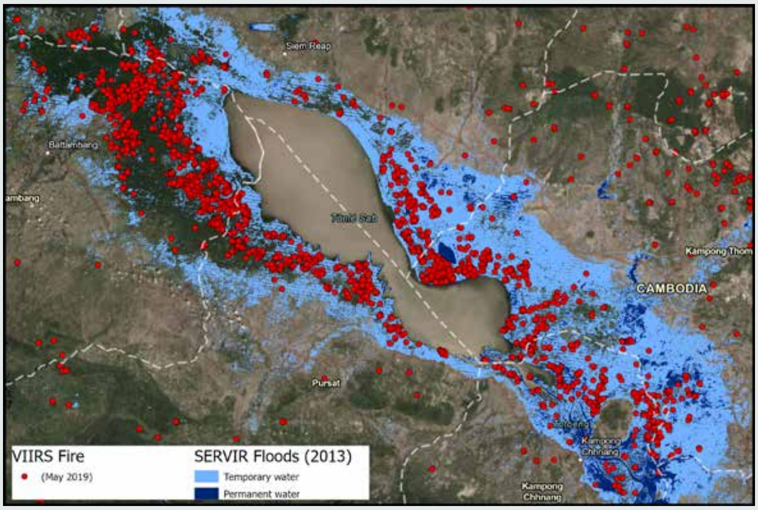


Figure 2.2. Map of Tonlé Sap, Cambodia showing the extent of fires in 2019 on lands previously inundated on a seasonal basis.

Case study: Wildfires in the degraded peatlands of Kalimantan, Borneo

In the past, major wildfires in tropical peatlands have been relatively rare, but recent events in Southeast Asia suggest that peatlands have become increasingly vulnerable to fire due to the expansion of clearcutting and large-scale draining. Although fires have historically occurred in the peat swamp forests of Kalimantan, since the late twentieth century these fires have increasingly become an annual occurrence, largely due to changes in land cover and shifts in rainfall. Local communities have sustainably used the peat swamp forest for centuries but providing food for Indonesia's growing population has driven land-use change in the region. Between 1996 and 1998, large areas of Central Kalimantan were cleared of forests and drained through an extensive system of canals for the Mega Rice Project (Notohadirprawiro 1998). Assisted by the severe drought conditions of the 1997–1998 El Niño, fires lit to clear the land escaped. Owing to a lack of resources to fight the fires and the remoteness of the area, the fires continued to burn for months (see Box in chapter 3 for an estimate of the economic cost of peat fires in Indonesia). The smoke plume that formed stretched into the Indian Ocean affecting numerous countries. It is estimated that more than 20 per cent of the peat swamp forest of Central Kalimantan was destroyed in 1997 (Boehm and Siegert 2001). The area contained globally important biodiversity hotspots (Wich et al. 2012) and the peat swamp forest held a significant store of carbon (Page et al. 2011).

Although the Mega Rice Project quickly failed, it left a legacy of increased surface run-off, reduced water retention across peatlands and a lowered groundwater table in the neighbouring areas. As a result, the cleared areas of the project dry out during the dry season and significantly increase the risk of wildfires in the region. Currently, 62 per cent of Central Kalimantan is subject to high fire vulnerability, with roughly another 10 per cent of the area having a very high vulnerability. Indonesia has committed to restoring peatlands and ending practices that contribute to wildfire vulnerability.

Training for members of the twenty-five community-based fire brigades established by the Central Kalimantan Peatlands Restoration Project.

Case study: Chilean production forests

Chile has around 18 million hectares of forest, of which just over 3 million hectares are plantation forests composed primarily of pine and eucalyptus species. In 2017, forestry employed over 100,000 people and earned more than USD 5 billion in exports (Richardson and Lehman 2018).

The establishment of plantations began in the 1970s, encouraged by government subsidies. They had the dual purpose of expanding the forest industry and helping to recover extensive areas of soil degraded by erosion. The proliferation of plantations opened the debate on the threat of fires, due to the increased availability of fuel and the establishment of extensive continuous areas where fire can spread rapidly and unconstrained. Accumulation of flammable fuels in monoculture plantations and their low moisture content in dry periods, plus the increase in extended droughts due to climate change, generate increasingly frequent conditions conducive to high-intensity forest fires. The silvicultural practice of extensive tree thinning and pruning reduces the likelihood of crown fires, hence fire spread rate and intensity. However, many plantations – especially those owned by small landowners – are not managed, increasing the risk of extreme fire events. Approximately 20,000 hectares of plantations and another 45,000 in native forest, grassland, and scrub, are burnt every year in Chile on average (however, in 2017 more than 500,000 hectares of plantations and 67,000 hectares of native forest were burnt) (Bowman et al. 2019).

In recent years, private and public approaches to integrating fire prevention and protection have been applied in both plantation and native forest areas. Agricultural and forestry owners have become more knowledgeable regarding the care necessary for the management of flammable vegetation. The extreme fires of 2017 sparked a change and led to a review of fire management. Forestry companies in Chile are committed to meeting ecological and environmental goals, as well as advancing community and local engagement (see chapter 5 for more information on community engagement). This is an essential requirement for facing the challenge of increasing wildfire activity in the region.

Aerial image after the Valparaíso fire in 2014.

Rapidly spreading fire in an interface zone in Valparaíso (Chile) in 2014.

2.2 Impact of climate change on global fires

Since industrialisation (1850–1900), the Earth has experienced a long-term warming trend, with an estimated increase in the global mean surface temperature of 1.09°C (IPCC 2021). Some areas of the planet have experienced accelerated warming with an increase of 1.59°C over land and, for example, temperatures in the Arctic rising more than twice as fast as the global average (IPCC 2018; IPCC 2021). One of the most important effects of anthropogenic climate warming has been its contribution to observed changes in fire regimes (Bowman et al. 2011). Warming has increased the frequency and magnitude of extreme weather conditions that drive the occurrence and spread of wildfires and has caused vegetation that would not usually burn to dry out and combust (e.g., rainforests, permafrost, and peat swamps). A review of 116 articles written since 2013 on climate change and fire concluded that there is a strong consensus that climate change is increasing the likelihood of fire occurrence in many regions (Smith et al. 2020).

Climate change influences on fire can be categorised as (1) direct effects on fire weather through drought, higher temperatures, and changes in the strength and seasonality of winds; (2) indirect effects resulting from changes in the nature and availability of biomass/fuel; and; (3) direct and indirect changes in the frequency and location of natural and human-caused ignitions via changes in dry lightning profiles, and changes in demographics and human behaviour resulting from revised climate and land management policies (Figure 2.3) (Dale et al. 2001; Krawchuk and Moritz 2011; McKenzie and Littell 2017; Restaino and Safford 2018).

El Niño-Southern Oscillation (ENSO)

Changing weather patterns also result from large-scale modes of climate variability, which influence atmospheric circulation on inter-annual to decadal timescales. The El Niño phase of the El Niño-Southern Oscillation (ENSO), for example, commonly results in higher temperatures and reduced precipitation across the tropics, leading to increases in fire (Prentice et al. 2011; Chen et al. 2017). Burnt area and fire emissions spike in El Niño years. For example, fires in the Brazilian Amazon increased between 2015 and 2016 (Aragão et al. 2018; Libonati et al. 2021); 4.5 million hectares in Indonesia burnt in 2015 (Lohberger et al. 2018), with emissions from peat fires alone reaching between 1.5 to 1.75 $GtCO_2$ – more than the entire total annual emissions of Japan for that year (World Bank 2015; Field et al. 2016; Crump ed. 2017; United Nations Framework Convention on Climate Change [UNFCCC] 2017). On average, El Niño events are responsible for an increase of 133 per cent in fire emissions in pan-tropical forests compared with La Niña years (Chen et al. 2017). As climate change continues to bring higher temperatures and more precipitation extremes, El Niño events could become more frequent and more intense in the future, increasing the risks from wildfires across the tropics (Fasullo et al. 2018).

The direct impacts of climate change on fire behaviour are already apparent in some places (Figure 2.4) and are the dominant drivers of fire regime change in much of the world's tropical and boreal forests (Touma et al. 2021). These impacts include changes in prevailing weather patterns during the fire season, resulting in periods of reduced rainfall and relative humidity, extreme air temperatures, and an increase in strong winds. These conditions increase the potential for successful fire ignitions and fire outbreaks. They also make it more difficult to suppress fire and increase the potential for outbreaks to become wildfires that burn unchecked for extended periods (Adams et al. 2020; Filkov et al. 2020).

Changes to weather and climate also have indirect effects on the type, nature, and condition of fuel, influencing the behaviour of the fire. How the landscape responds to climate change depends on how growing conditions change (e.g., rainfall, temperature, and evapotranspiration), what happens to fire regimes (i.e., the pattern of fires over time; Bradstock et al. eds. 2002), land management practices, and the ecophysiology[1] of biomass fuel species. Elevated carbon dioxide levels or increased plant-available water from changes in rainfall patterns may enhance vegetation productivity and vegetation fuel production, thereby increasing fuel loads (Booth et al. 2008). Conversely, decreased rainfall and increased drought may decrease long-term vegetation growth and thus above-ground fuels (but may lead to increased drying of peatlands making them more prone to fires; Turetsky et al. 2015). Worldwide, changing fuel loads are emerging as the dominant cause of fire regime change in tropical and Mediterranean savannas and grasslands, temperate woodland, and arid systems such as shrub and desert (Figure 2.4).

The impact of these complex interacting processes on fire regimes are not well understood and depend on whether factors act synergistically or antagonistically (Balch et al. 2009; Williams et al. 2011). While decreased rainfall and increased drought reduce fuel loads, they may also lead to decreased atmospheric humidity resulting in decreased fuel moisture, increasing fuel flammability and the potential for successful fire ignitions and increased fire spread rates (Littell et al. 2016). This process is driving the fire regime shift in the Kazakhstan-Russia fire zone (Kelley et al. 2019). Conversely, in North American and European Boreal

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[1] Here, "ecophysiology" refers to the way in which the physiology of a plant responds to the environment.

Potential reinforcing feedback loop of climate change on wildfires

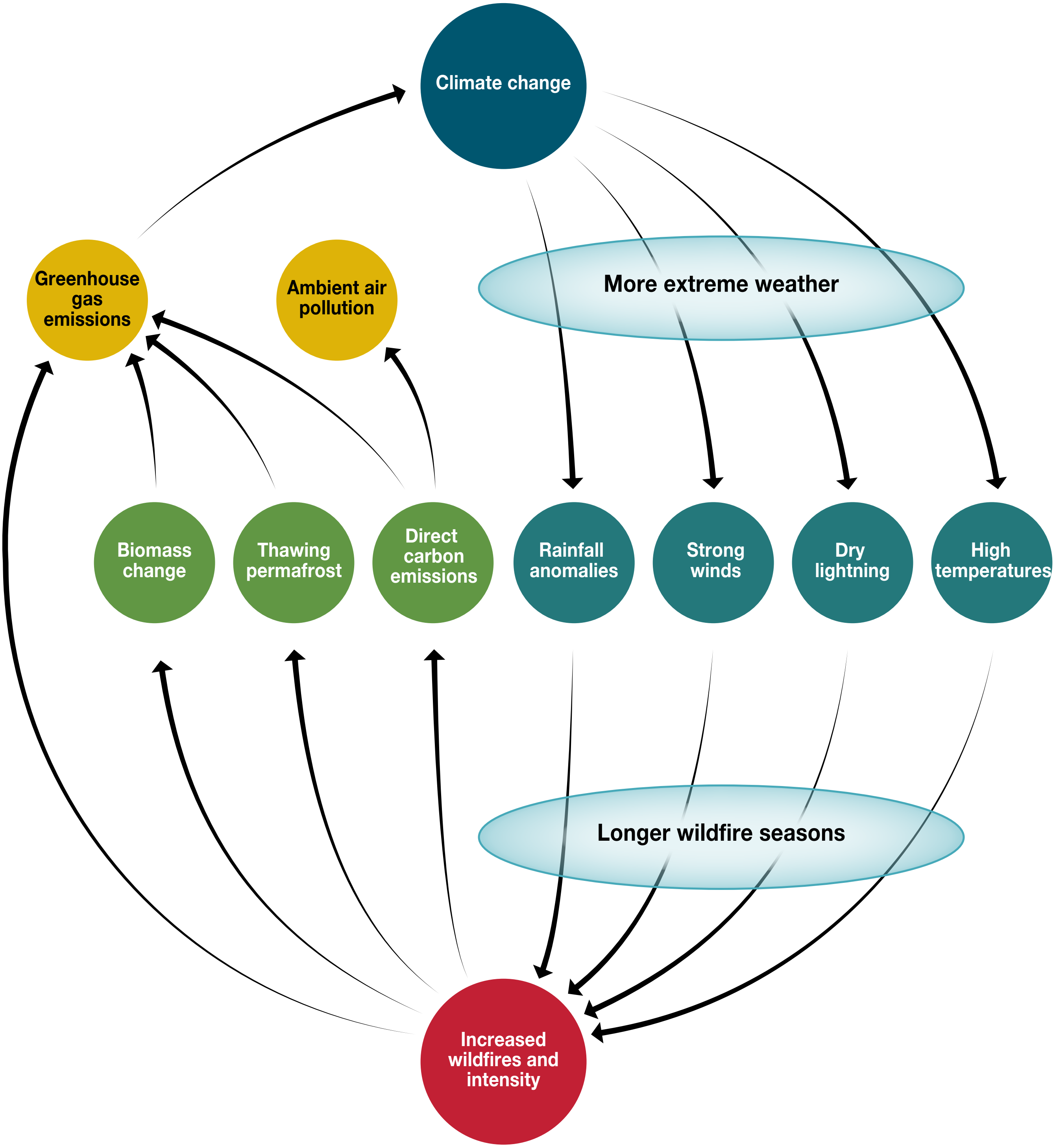
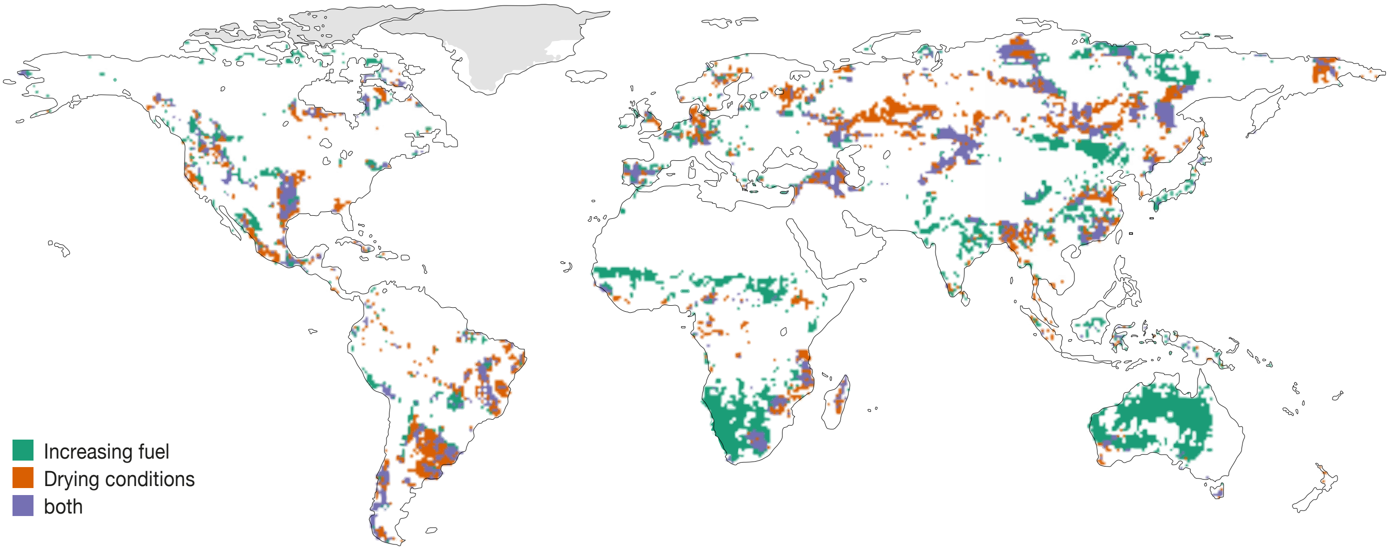


Illustration by Andrew Sullivan/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

Figure 2.3. Potential reinforcing feedback loop of climate change on wildfires. Climate change will directly affect the frequency and magnitude of extreme weather conducive to the outbreak and spread of wildfires. It will also lead to longer wildfire seasons where the fire season may begin earlier and end later. Increased wildfire activity can positively impact greenhouse gas emissions that reinforce climate change drivers.

Increases in burnt area due to changing fuel and/or moisture, 2001-2014



Source: Kelley et al. 2019.

GRID-Arendal/Studio Atlantis, 2021

Figure 2.4. Areas where increases in burnt area between 2001 and 2014 were driven by changes in either fuel loads, moisture content, and/or fuel and moisture, using the same fuel and moisture controls as in Figure 1.7.

forests, the increased moisture conditions that enhance fuel production also partly offset the amount of fuel available for combustion (Kelley et al. 2019).

Accelerated warming in the Arctic is extending growing season length and vegetation productivity, as well as thawing permafrost (Osterkamp 2005; Swanson et al. 2021), thereby increasing fuel flammability (Figure 2.4). Some of the largest percentage increases in fire occurrence are found in the Arctic (Holloway et al. 2020; McCarty et al. 2020). Meanwhile, drying conditions in China's tropical and warm temperate forests are compounded by increased fuel continuity because of cropland cover retreat (Kelley et al. 2019).

2.3 Influence of climate change on extreme fire events

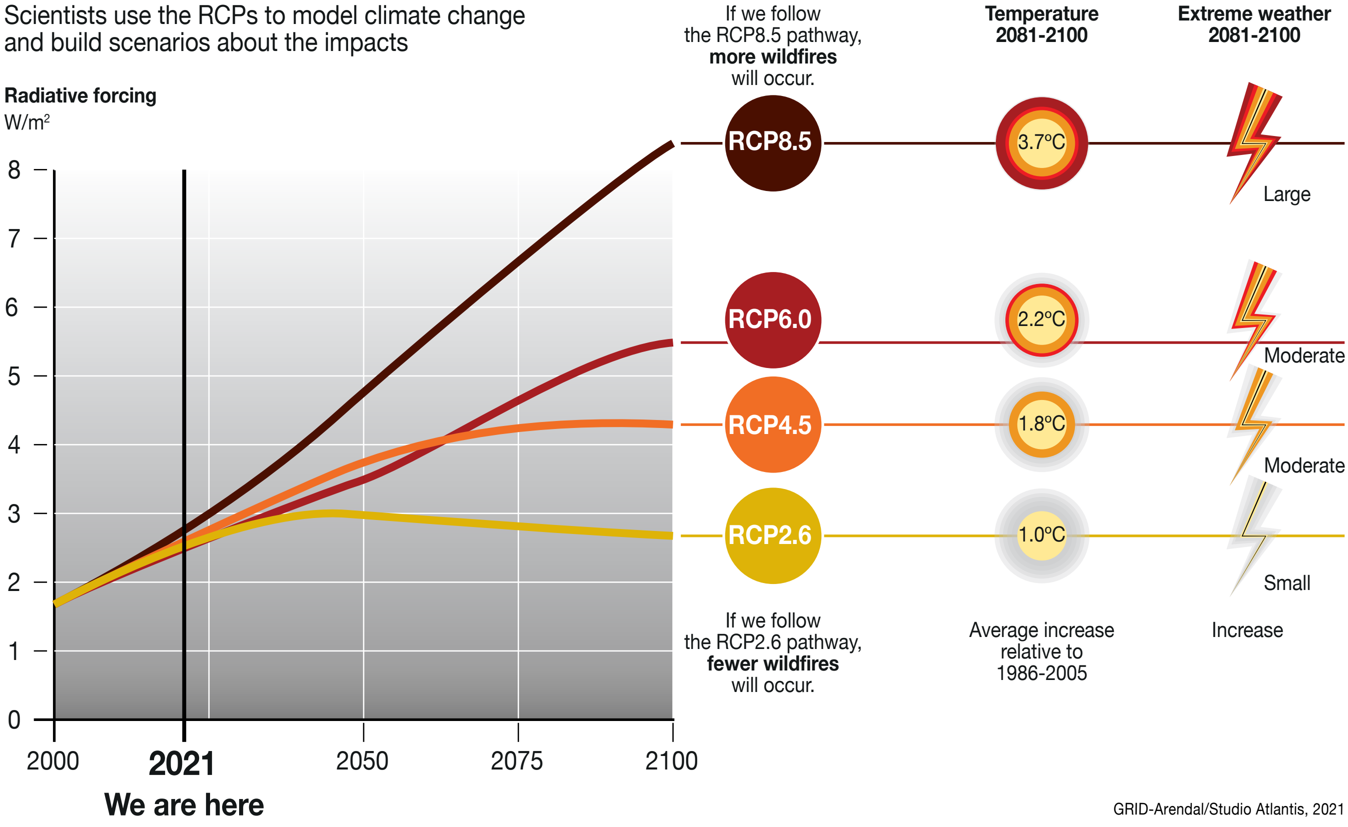
Some of the large-scale wildfire events that have made the headlines in recent years occurred in areas that typically experience much less burning. There have also been wildfire events in normally fire-prone ecosystems at a much higher intensity or over larger areas than expected (for example, Australia in 2019–2020 and California in 2020 – see subsequent case study on Australian wildfires or over a much longer season (see Arctic fire case study).

In some cases, there is an obvious and apparent meteorological cause of these extreme fire events – a scorching hot period of weather, strong winds, or lightning storms, among others. When we experience extreme or unusual weather events, the question frequently asked is, "was climate change the leading cause?" There is often no definitive answer to this question. However, with attribution studies, we can investigate how an event's likelihood has changed due to climate change by using models to compare the real world to a hypothetical world without anthropogenic emissions and associated warming.

Several studies have used this method to attribute unusual fire events to natural or anthropogenic causes. Kirchmeier-Young et al. (2017), for example, assessed the record fire season in Canada in 2017, where 1.2 million hectares of land in British Columbia burnt, and concluded that climate change increased the area burnt by a factor of 7–11. The extreme weather conditions that were potentially a leading cause of the fire season in 2019–2020 in Australia have been shown to be 30 per cent more likely to have occurred because of climate change (van Oldenborgh et al. 2020). Several attribution studies have focused on fire in the western USA. They have concluded that as a result of climate change, the number of autumn days with weather suitable for wildfires has doubled since the 1980s (Goss et al. 2020) and that fire extent has increased fivefold since the 1970s, also very likely due to human-induced warming and the resultant drying of fuels (Williams et al. 2019). The 2020 Siberian heatwave that was associated with extensive burning in the Arctic Circle was the first event shown to be almost impossible without climate change, with the likelihood of this happening being only once in 80,000 years without anthropogenic emissions, and climate change increasing the chances of prolonged heat by a factor of at least 600 (Ciavarella et al. 2020). Conversely, by looking at the fuel and moisture drivers at the land surface, Kelley et al. (2021) found very little meteorological influence for the 2019 Amazonian deforestation fires, suggesting that landscape modification or human fire ignitions were the main drivers. Libonati et al. (2021) also showed that even though 2015 had the most severe drought ever recorded in the Amazonia, the total number of remotely-sensed active fire counts were 21 and 50 per cent lower than those reported for the extreme droughts of 2010 and 2005, respectively. While not yet possible, direct tests of land management, cover change, and ignitions will be useful in determining the influence humans have on exacerbating or mitigating wildfire events, helping us learn how to respond to future potentially more extreme events.

Representative Concentration Pathway (RCP)

Scientists use the RCPs to model climate change and build scenarios about the impacts



GRID-Arendal/Studio Atlantis, 2021

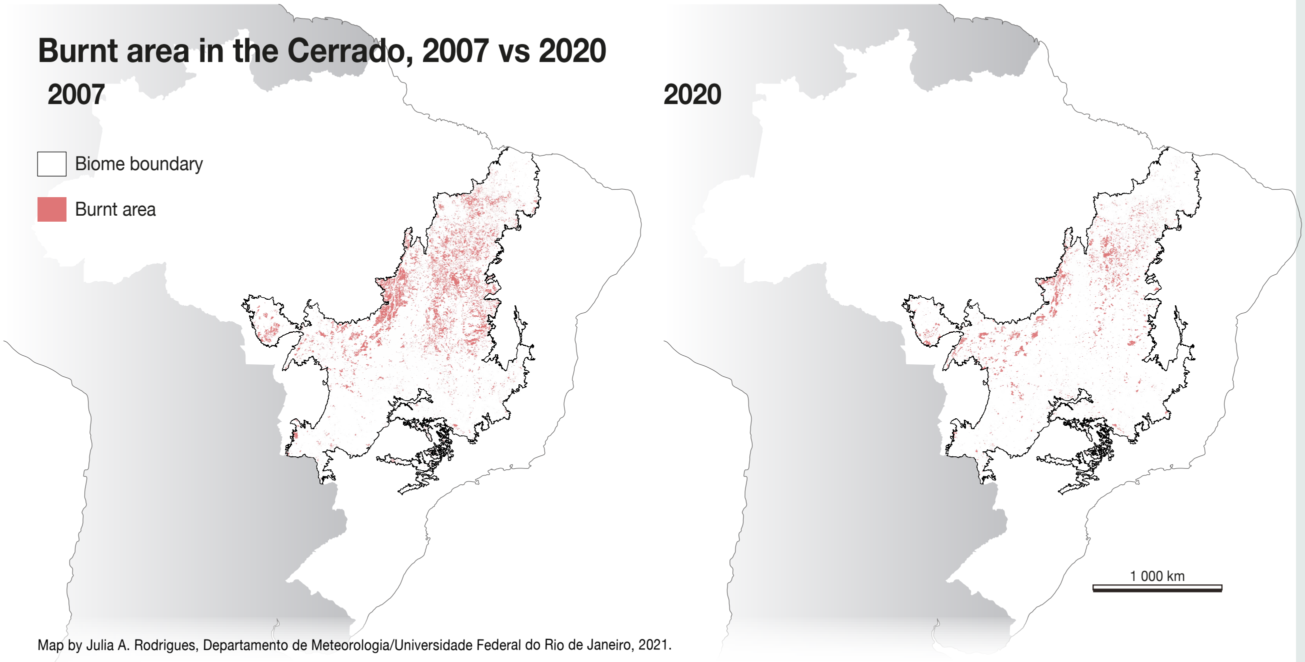
Figure 2.6. Representative Concentration Pathway(s) (RCPs) are trajectories of greenhouse gas concentrations used for climate modelling in the IPCC Fifth Assessment Report (IPCC 2013). The numerical values of the RCPs (i.e., 2.6, 4.5, 6.0 and 8.5) refer to the possible range of radiative forcing values in the year 2100. RCPs are used to build future climate scenarios based on greenhouse gas emissions from human activities, depending on the efforts taken to limit greenhouse gas emissions (high efforts taken under RCP2.6, low efforts under RCP8.5). RCP2.6 is the scenario that will likely keep global warming below 2°C by 2100 – this alone will have a significant impact on reducing wildfire occurrence (see also Figure 2.8).

Case study: The changing fire regime in the Brazilian Cerrado

The Cerrado is a Brazilian tropical savanna – a fire-prone biome that covers almost 2 million km^2 of which less than 60 per cent remains as natural vegetation (Strassburg et al. 2017). Only 3 per cent of the original area is currently protected (Ferreira et al. 2020), although the Cerrado is considered the most botanically diverse savanna and is recognised as a biodiversity hotspot. The region experiences increased fire activity from August to October and has historically accounted for more than half of Brazil's annual burnt area (Figure 2.5). The use of fire for land conversion is common, and the highest fire activity is observed in regions where most of the biome's natural vegetation cover remains, along the new agricultural frontier ("MATOPIBA", the region comprising the Brazilian state of Tocantins and some parts of the states of Maranhão, Piauí and Bahia) and in the transitional area between the Cerrado and Amazonia biomes (also known as the Arc of Deforestation) (Silva et al. 2021).

In recent years, increased deforestation for agriculture, fire suppression policies, and regional climate changes have led to an increasingly altered fire regime (Pivello 2011). Late dry season fires have become more frequent in many regions of the Cerrado, with extreme wildfires occurring every two to three years, burning both fire-resistant and fire-sensitive vegetation (Schmidt and Eloy 2020).

The Cerrado is projected to experience increasing temperatures, lower relative humidity, and altered precipitation regimes for the remainder of the century (Silva et al. 2016). A recent study suggests that weather factors are responsible for more than two-thirds of inter-annual variability in the Cerrado burnt area (Silva et al. 2019). Using IPCC's climate change scenarios (RCP2.6, RCP4.5, and RCP8.5), the burnt area is expected to increase in the Cerrado, associated with a higher probability of extreme events (see Figure 2.6 for an explanation of the RCPs). The medium CO_2 stabilization scenario, RCP4.5, indicated a 39 per cent increase in the burnt area by 2100, while the most ambitious CO_2 mitigation scenario, RCP2.6, resulted in a 22 per cent increase by 2050 compared with the historical period, followed by a decrease to 11 per cent by 2100. The conditions predicted under RCP2.6 show the importance of limiting global warming to 1.5°C by the end of the century to minimise the environmental and social costs associated with wildfires in the Cerrado.



Burnt area in the Cerrado, 2007 vs 2020

Map by Julia A. Rodrigues, Departamento de Meteorologia/Universidade Federal do Rio de Janeiro, 2021.

Contribution of the Cerrado to Brazil's annual burnt area, 2001–2019

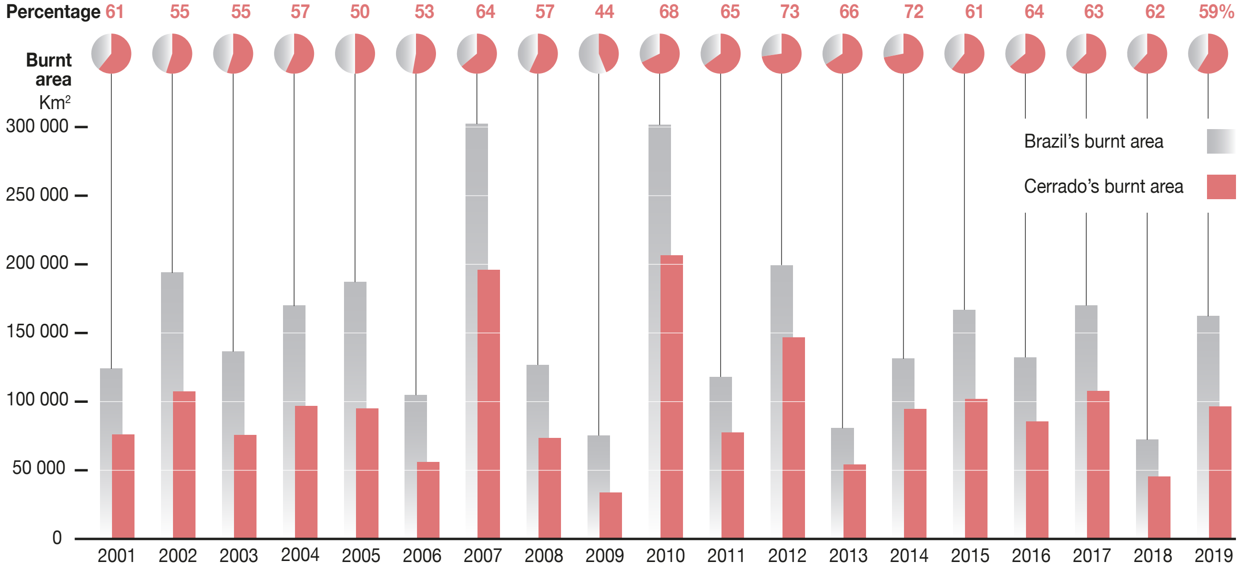


Illustration by Patrícia S. Silva, Instituto Dom Luiz/Universidade de Lisboa, 2021.

Source: MODIS Collection 6 MCD64A1 Burned Area product (Giglio et al. 2018).

GRID-Arendal/Studio Atlantis, 2021

Figure 2.5. Top: Difference in fire activity in an ordinary (2020) and extraordinary (2007) year in the Cerrado. Changes in the burnt area reflect the occurrence of wildfires in 2007. Bottom: The inter-annual variability of burnt area in Brazil from 2001–2019 is shown in grey and the corresponding percentage from the Cerrado is shown in red. All panels use the MCD64A1 500m product.

2.4 The impact of wildfires on the global carbon cycle

Human and climate modifications of global fire regimes can significantly affect the terrestrial carbon budget, atmospheric carbon dioxide concentrations, and global temperatures (Bowman et al. 2009; Pan et al. 2011; Lasslop et al. 2020).

2.4.1 Peatlands

Peatlands are one of the world's largest sources of terrestrial carbon. Despite covering only 3 per cent of the world's surface, they store 30 per cent of the global total soil carbon (Dargie et al. 2017), and contain more carbon than all other vegetation types in the world combined (Turetsky et al. 2015). These carbon-rich peatlands are at risk of degradation from multiple sources, including wildfires. During the 1997–1998 El Niño, hot, dry conditions across Indonesia combined with large areas of degraded, drained swamp forest peatlands and extensive uncontrolled fire, resulting in fires unprecedented in extent, severity, and duration. In addition to large-scale smoke generation, the carbon emissions were equivalent to 13–40 per cent of the mean annual global total carbon emissions from all fossil fuels (Page et al. 2002). Widespread forest and peatland fires also occurred in more recent years, notably in 2015 and 2019.

Human-caused or naturally occurring peatland fires double the carbon emissions of degraded peatlands, currently representing 5 per cent of the total anthropogenic CO_2 emissions worldwide (Joosten 2015; Urák et al. 2017). Peatland fires usually consume both the surface vegetation and a portion of the underlying peat layer. The fires cause regional air pollution, resulting in widespread chronic health impacts (Crippa et al. 2016; Koplitz et al. 2016) and economic losses (Glauber et al. 2016). They contribute significant greenhouse gases to the atmosphere, affecting climate on a global scale, and emit pollutants, which can affect both ecosystem and human health.

The Arctic contains nearly half of the world's peatlands and approximately 80 per cent of the global peatland carbon and nitrogen stocks (Hugelius et al. 2020). Positive feedback, known as Arctic amplification, causes the Arctic to warm more rapidly than the rest of the world. This leads to accelerated thawing of permafrost and melting glaciers. The thawing of permafrost peatlands makes them susceptible to fire, which can turn these historical carbon sinks into a net source of carbon in the atmosphere, further accelerating climate change (Hugelius et al. 2020). In 2020, Arctic fires were responsible for releasing 0.244 Gt CO_2 into the atmosphere – 35 per cent more than the previous year, which also set records (Witze 2020). The additional warming and thawing of permafrost peatlands due to wildfires can speed up the development of hazardous landscape features such as sinkholes and thermokarst bogs,[2] adversely impacting Arctic communities (Crump ed. 2017; Gibson et al. 2018).

2.4.2 Tropical forests – Amazon

Amazon forests contain nearly half of the world's tropical forest carbon stocks (Pan et al. 2011). Although natural fires are extremely rare in the Amazon, a combination of deforestation, changing land-use practices and droughts have made this ecosystem susceptible to wildfire (Aragão et al. 2018; Libonati et al. 2021). Over the past two decades, droughts and heatwaves have become more frequent and severe (Geirinhas et al. 2018; Panisset et al. 2018). These hot and dry periods increase tree mortality and vulnerability to fire (Machado-Silva et al. 2021). Deforestation fragments the forest, which also has an impact on the increase in fire incidence at the forest edges and patches. Ninety-five per cent of active fires and the most intense fires are found within 1 km of the forest edge (Silva Junior et al. 2018).

Burnt forests have been found to contain almost 60 per cent less above-ground carbon than undisturbed forests (Berenguer et al. 2014). Recovery is slow, with significantly lower biomass levels than unburnt forests still observable decades after burning (e.g., 25 per cent less biomass has been recorded 31 years after burning; Silva et al. 2018). This lower biomass results from canopy destruction and high tree mortality, which is not compensated for by the growth of the surviving trees or new trees (Silva et al. 2020). With more frequent droughts predicted, an increase in wildfires can be expected (Figure 2.8), along with a corresponding increase in burnt fallen and standing trees. Reducing carbon storage in the Amazon may create significant positive climate feedback that will further increase warming trends. In the Brazilian Amazon, fire emissions during extreme drought years can be higher than deforestation emissions. Forest fires alone currently contribute to mean annual emissions of 0.454±0.496 Gt CO_2 $year^{-1}$ (2003–2015) or 31±21 per cent of the estimated emissions from deforestation (Aragão et al. 2018).

2.5 The future of wildfires

The impacts of climate change on fire behaviour will be complex. The IPCC Sixth Assessment Report states that weather conducive to wildfires (hot, dry, and windy) has become more frequent in some regions and will continue to increase with higher levels of global warming (IPCC 2021). Fire regimes are sensitive to small changes in precipitation and rainfall distribution, and, compared with temperature, climate model projections for these are less certain, especially in the mid-latitudes (Langenbrunner et al. 2015). Forecasting fire outbreaks is similarly uncertain. As a result, projecting specific future changes in burning can be extremely imprecise even in regions where modelled climate-fire relationships are strong. Scholze et al. (2006) instead identified areas of the world where climate model projections show large changes in future burnt areas, even if models disagreed on the magnitude or even direction of that change.

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[2] A bog formed by the thawing of ice-rich permafrost.

Future wildfire risk

Human-induced climate change is increasing the incidence of wildfires around the globe (see section 2.2). But how bad is it likely to get? It is hard to say exactly because even though the latest IPCC report confidently predicts an increase in extreme fire weather (IPCC 2021), other complex interacting processes influence wildfire risk (see Figure 1.3). However, despite the limitations of current global wildfire models to capture all these interactions (Hantson et al. 2016; Kloster and Lasslop 2017; Hantson et al. 2020), the available information illustrates an increasing risk in many locations. For example, an upward trend in burnt area (despite large interannual variability) is evident in the forests of eastern Australia (Canadell et al. 2021; Abram et al. 2021), Siberia (e.g., Sizov et al. 2021; Kharuk et al. 2021), Canada (e.g., Coops et al. 2018), and the western United States (e.g., Abatzoglou et al. 2021). Similarly, Figure 2.7 illustrates a predicted increase in extreme fire events between now and the end of the century under different future climate scenarios (Kelley et al. 2021).

It is well established that exposure to wildfire smoke causes adverse human health impacts (see section 3.2). A recent study examined the relationship between mortality and wildfire-related $PM_{2.5}$ in 749 cities across 43 countries from 2000 to 2016 (Chen et al. 2021). The authors found that 0.62 per cent of deaths were attributable to acute exposure to wildfire smoke (10 μg/m3 increase in the 3-day moving average of $PM_{2.5}$). This equates to an estimated annual death toll of 33,510 people across the 749 cities. Wildfire smoke can also make people more susceptible to other respiratory illnesses, including COVID-19. A recent study from the western United States found that over 19,700 cases of COVID-19 and 748 deaths were attributable to increased $PM_{2.5}$ from wildfire smoke (Zhou et al. 2021).

In recent years, wildfires have been responsible for up to 50 per cent of the $PM_{2.5}$ air pollution in the western United States – a substantial increase over the last decade (Burke et al. 2021). A related study estimated that 82 million people in this region would be affected by "smoke waves" (two or more days of unsafe PM2.5 levels related to wildfires) by the middle of the 21st century (Liu et al. 2017). Without increased fuel management, exposure to wildfire smoke will continue to increase (Burke et al. 2021), especially as more and more people move into areas adjacent to wildland (estimated at one million new homes every three years across the USA). Livestock are also increasingly affected by smoke. A survey of farmers carried out after the 2020 wildfire season in western USA indicated that on top of direct losses to fire, sheep, cows, and goats experienced a range of impacts. These included poor weight gain, reduced conception, decreased milk production, and cases of pneumonia (O'Hara et al. 2021).

Even with the most ambitious efforts to curb greenhouse gas emissions, the planet will still experience a dramatic increase in the frequency of extreme fire conditions (Figure 2.7). This means that by the end of the century, the probability of wildfire events similar to Australia's 2019–2020 Black Summer or the huge Arctic fires in 2020 occurring in a given year is likely to increase by 31–57 per cent. Wildfires are already affecting the health of millions of people and animals, straining national and local economies, and increasing economic inequality. As wildfires become more frequent, the impacts will increase (Hsiang et al. 2017).

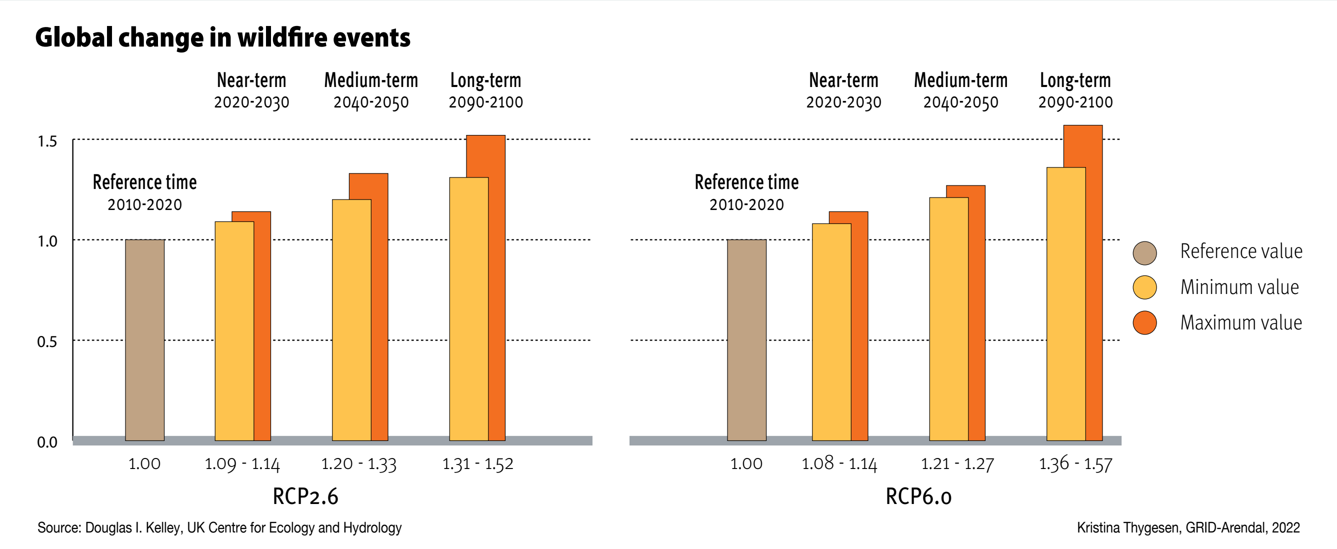
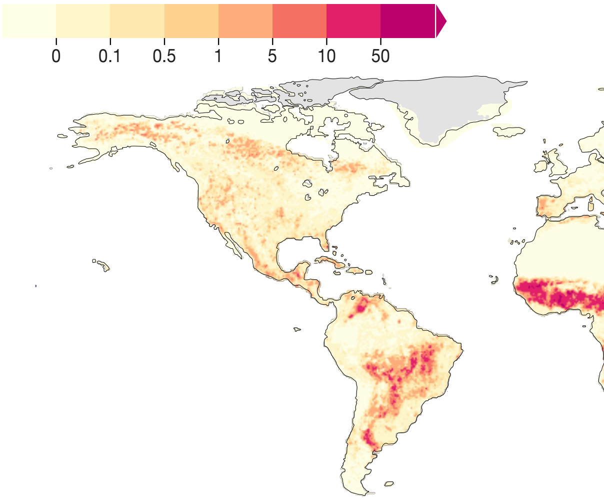


Figure 2.7. By the end of the century, the likelihood of catastrophic wildfire events will increase by a factor of 1.31 to 1.57. Even under the lowest emissions scenario, we will likely see a significant increase in wildfire events. See appendix for construction.

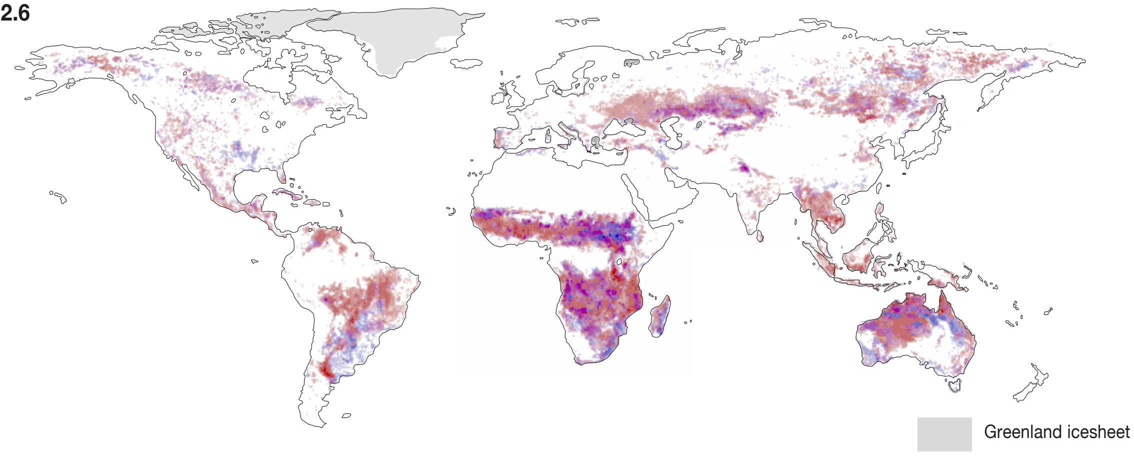
Future changes in burnt area

Historic

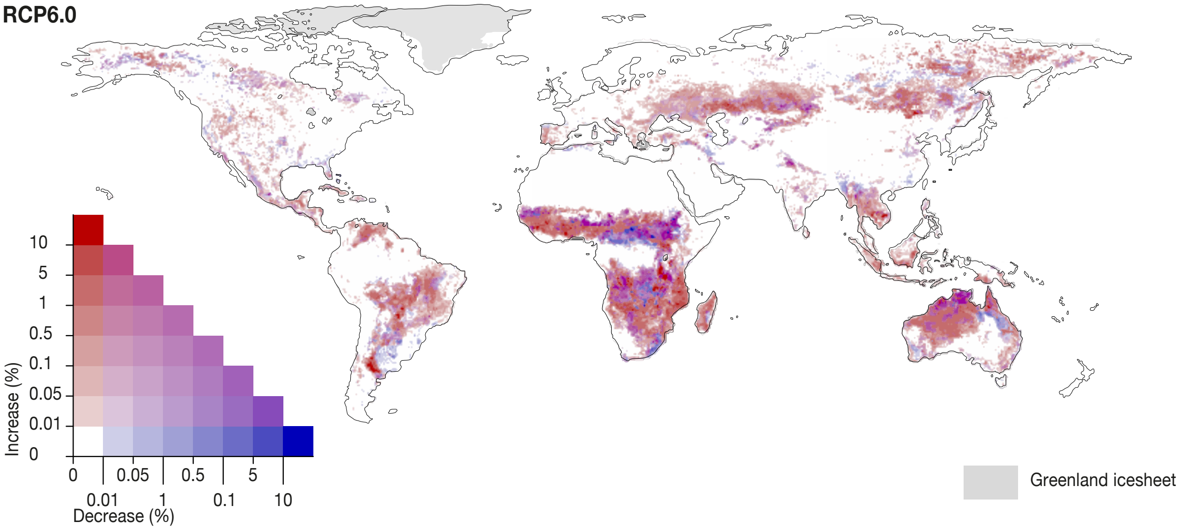


Change in burnt area as a % of land area

RCP2.6



RCP6.0



Map by Douglas I. Kelley/ UK Centre for Ecology and Hydrology, 2021.

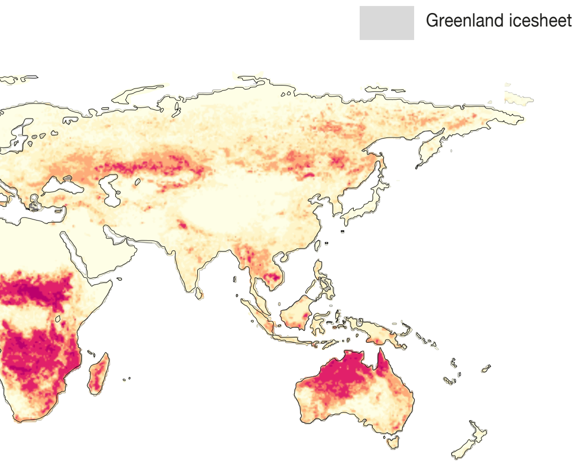
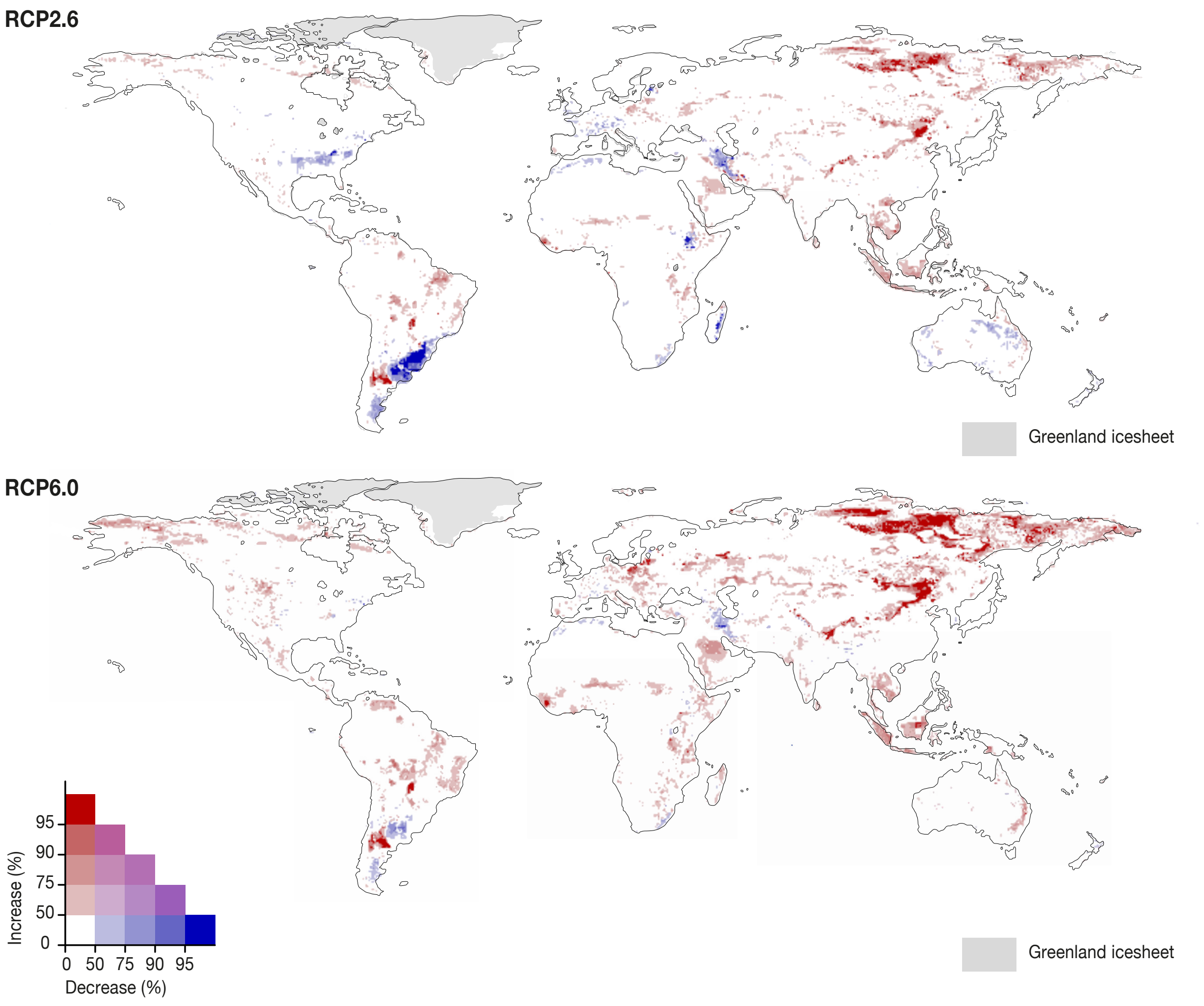


Figure 2.8. Changes in burnt area. Left: future changes in burnt area compared with historic burnt area (top). Right: areas with significant changes in burnt area under RCP2.6 (best case scenario – middle row) and RCP6.0 (bottom row). Inter-Sectoral Impact Model Intercomparison Project 2b (ISMIP2b) climate model ensemble by the last decade of the century compared with 1996–2005. Areas in red are where the burnt area is projected to increase, while the blue shows areas where it will decrease. Purple areas are those where some climate models project an increase while others a decrease. See appendix for construction.

Figure 2.9. *(overleaf)* Change in fuel continuity (left) and dryness (right) by the last decade of the twenty-first century compared with 1996–2005 for RCP2.6 and RCP6.0. The black dots indicate areas of a significant shift in either fuel or dryness. See appendix for construction.

Areas of significant change in burnt area



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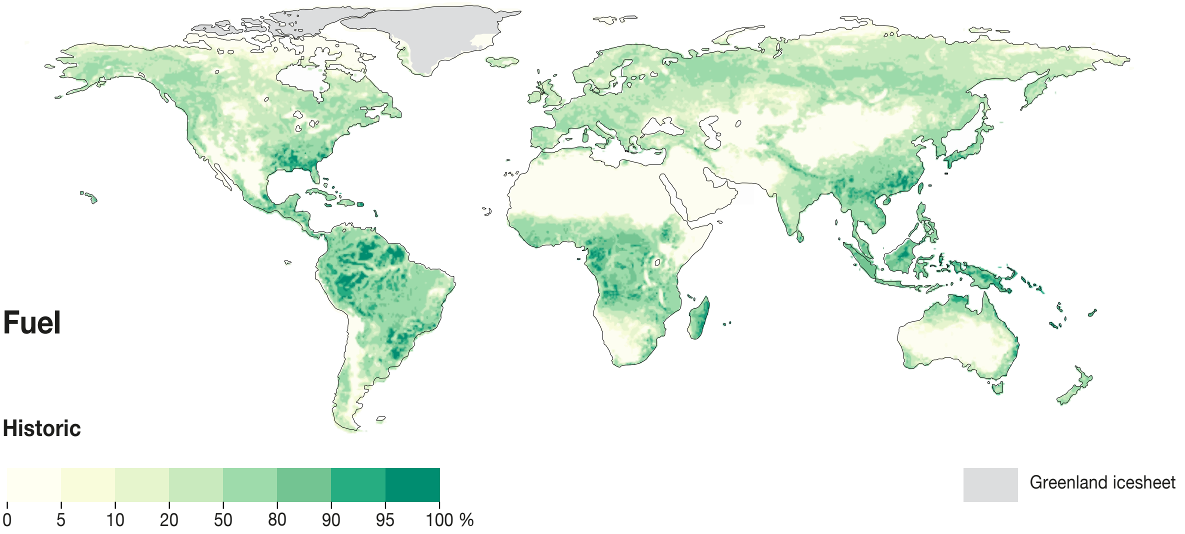
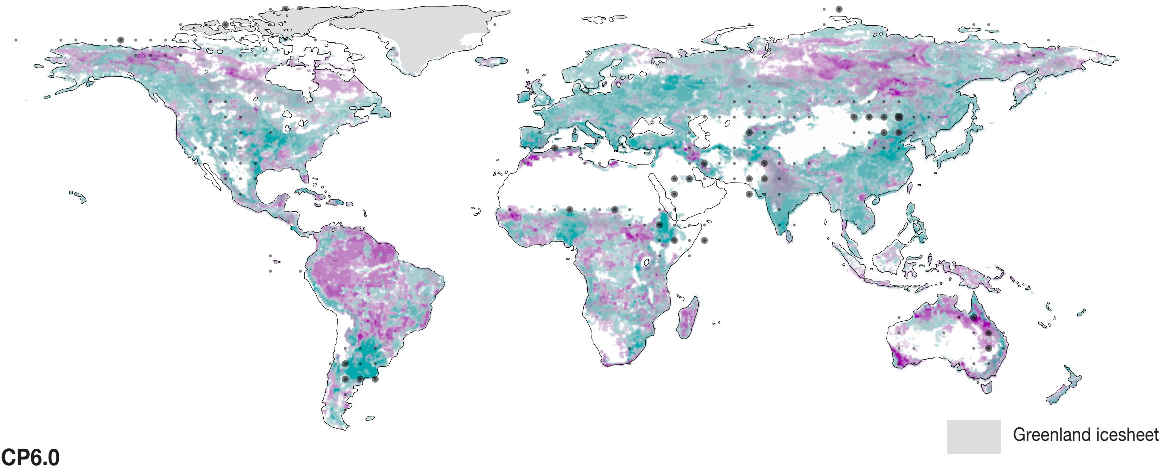
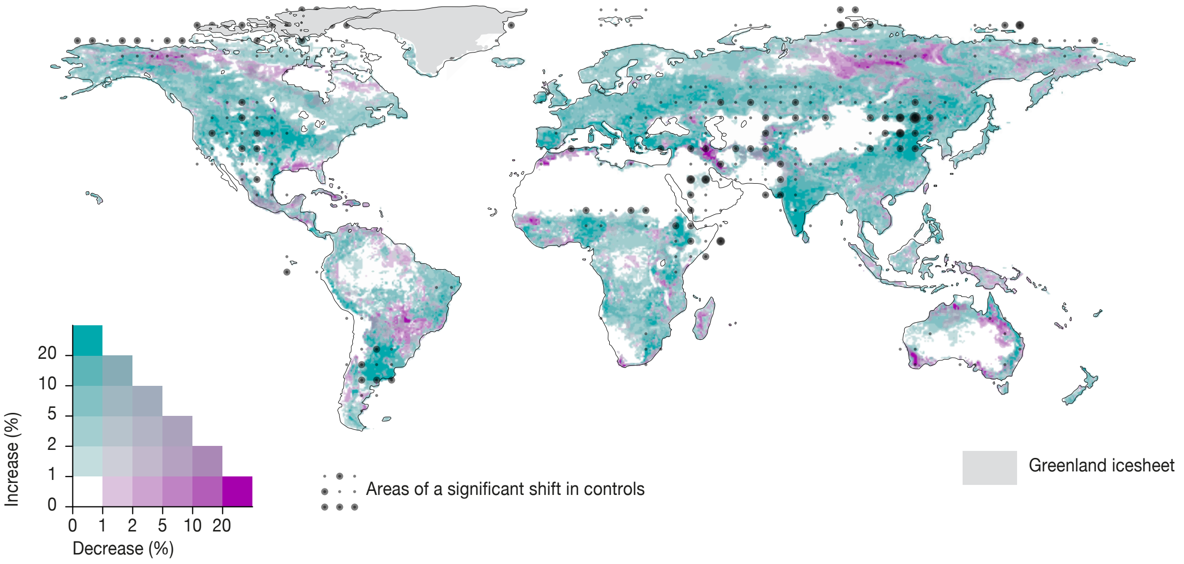
Areas of significant change in fuel continuity and dryness



RCP2.6



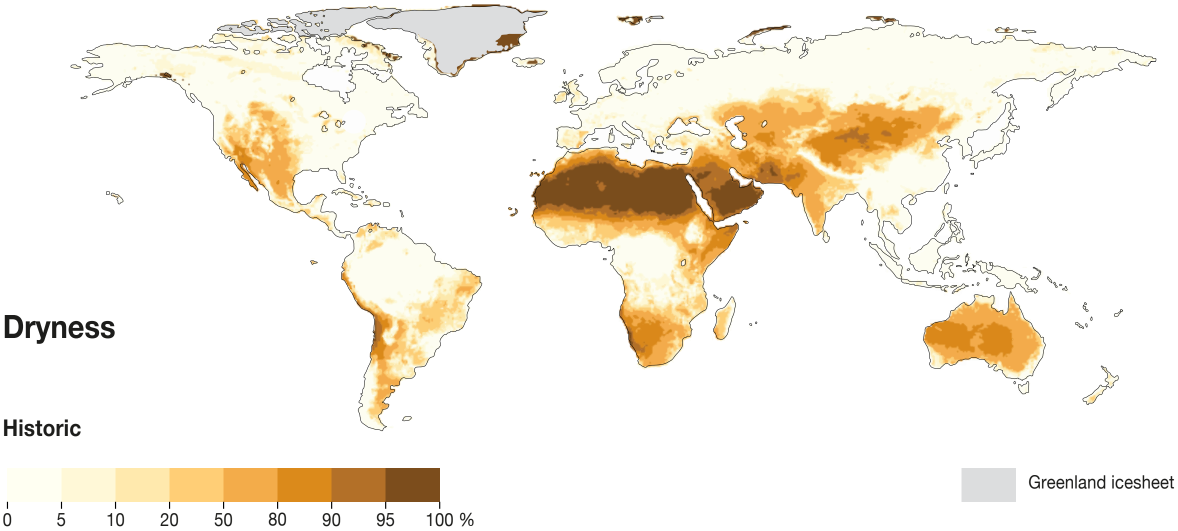
RCP6.0



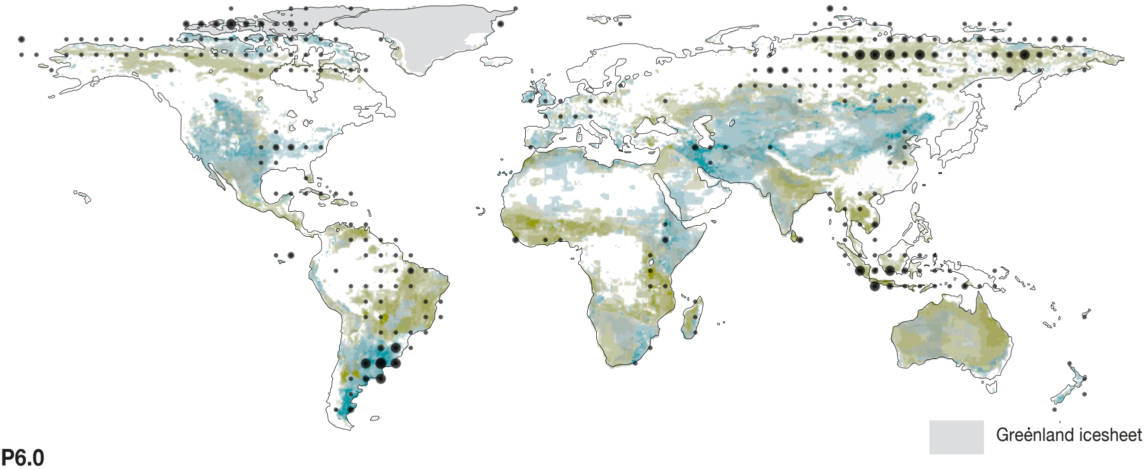
Map by Douglas I. Kelley/ UK Centre for Ecology and Hydrology, 2021.

Dryness

Historic



RCP2.6



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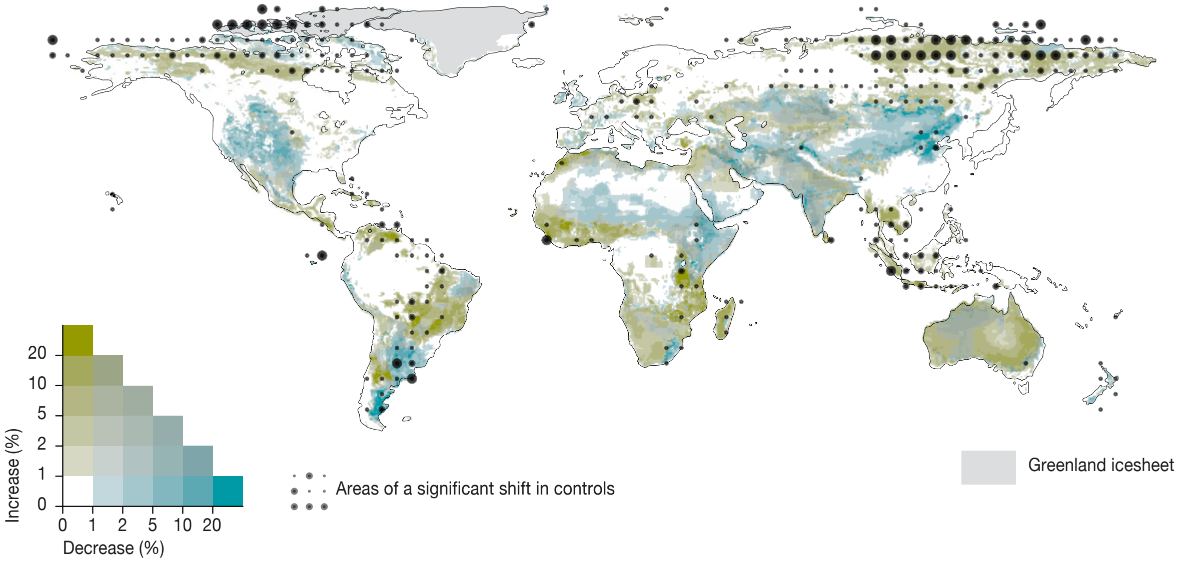

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Updating their analysis with more recent climate models and advances in future fire modelling techniques suggest large changes by the end of the century in areas of the world associated with frequent burning today, irrespective of future emission scenarios (Figure 2.8). These areas include tropical savannas and tropical and temperate grasslands. However, changes here are not consistent across models – some models suggest a large increase in burnt area, while others suggest a large decrease. In many ecosystems, the most important question is what effect climate change is going to have on the structure and distribution of the fuel that drives fire behaviour, and which is not necessarily the dominant vegetation in the ecosystem (Matthews et al. 2012).

There are some areas with little burning today where most models suggest a small but significant and consistent change in future burnt area. In Arctic areas, especially in northern Siberia, most climate models imply a significant drying of conditions as likely to increase burning by the end of the century. Drying conditions over Indonesia could also lead to more burning. These areas are of particular concern given the potential to release the high carbon content of peatlands and their irrecoverable carbon (Goldstein et al. 2020), potentially exacerbating global warming.

Meanwhile, an increase in fuel due to shifts in rainfall and CO_2 fertilisation may see the encroachment of fires in eastern Asia's arid areas, central USA, and desert areas of South America. These patterns seem to be mostly consistent with different warming levels, though the more extreme future climate scenarios exacerbate these changes. There is also a strong agreement between models of increasing moisture and decreasing fire in northern Argentina, southern Brazil, and Uruguay, and along the east coast of the USA under less extreme emission scenarios (but not under more extreme scenarios – Figure 2.9).

Whether wildfires increase or decrease in a region, the resultant changes in fire regime may result in distinct changes in fuel type (e.g., frequent burning may convert closed forest to open forest with a grassy understorey or change vegetation entirely to shrubland – Bowman et al. 2020a). How catastrophic these changes are will depend on the extent and behaviour of fire and its interaction with at risk populations and assets.

Wildfire risk to REDD+ project accounting

REDD+ is a framework created by the United Nations Framework Convention on Climate Change (UNFCCC) Conference of the Parties (COP) to support activities that reduce emissions from deforestation and forest degradation. In developing countries, it also supports the growth of carbon stocks through the sustainable management and conservation of forests. This framework includes results-based payments (RBPs) to reward countries for achieving reduced emissions from forests. The payments are made based on tonnes of CO_2e emissions avoided.

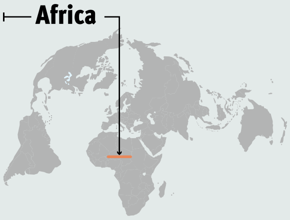
RBPs are picking up pace. As of today, the Green Climate Fund has approved US$500 million in RBPs to forest countries. The World Bank's Forest Carbon Partnership Facility (FCPF) reported agreements for RBPs totalling US$181 million (FCPF 2021). The LEAF Coalition (Lowering Emissions by Accelerating Forest Finance) aims to provide US$1 billion to pay for emission reductions between 2022 and 2025 (LEAF Coalition 2021). The voluntary carbon market has mobilized about US$ 400 million in forest carbon finance between 2017 and 2019 (Forest Trends' Ecosystem Marketplace 2021).

Confidence in emission reductions from REDD+ relies on ensuring that these are accurately calculated and permanent (e.g., Chagas et al. 2020). There is a risk of sequestered forest carbon being re-emitted into the atmosphere (termed reversals) due to disturbances such as fire, logging, land clearing, pests, and landslides. To compensate for this risk, REDD+ project and jurisdictional programmes employ a "buffer" system, whereby a percentage of carbon credits are placed into a buffer account or pool. Buffer credits are managed by an independent body and in the event of emission reversals, the carbon losses are accounted for by withdrawing an equivalent number of credits from the buffer account.

The number of buffer credits required to be set aside for potential reversals is based on an evaluation of the risk of reversal for a particular area. There are a several methodologies used by different carbon standards. All methodologies explicitly or implicitly include the risk of fire (calculated from long-term averages and sometimes with discounts for active fire management). However, due to a lack of data and analysis, accounting for the risk of wildfires is difficult and there is currently no generally accepted methodology. Individual projects and jurisdictional programmes rely on having enough credits in a buffer account to cope with what can potentially be large-scale reversals.

The threat of wildfires has the potential to make maintaining an appropriate buffer level more challenging and expensive (due to increased uncertainty). The United Nations Environment Programme (UNEP) is examining ways to improve the understanding of wildfire risk to more accurately calculate the number of buffer credits required to insure against wildfire-induced carbon loss.

Case study: Fire in African savanna and grassland ecosystems



Human activity is considered the primary source of ignitions in the tropical forests, savanna, and agricultural regions of Africa (Archibald et al. 2009). Fire activity in Africa's grasslands and savannas has, however, declined by 25 per cent over the last two decades (Andela et al. 2017). Typically, small to intermediate and less intense land-use fires dominate the African landscape (Archibald et al. 2013). Humans have been managing fire for millennia, and fire remains integral to savanna and grassland ecology (Laris et al. 2015).

In sub-Saharan Africa, the grassy-savanna biomes dry out annually and provide the fuel for fires. Wildfires will occur almost anywhere there is enough biomass (i.e., fuel) to sustain connectivity (Archibald et al. 2013) and will vary in relation to productivity (i.e., rainfall) and the season (Pausas and Ribeiro 2013). African savannas and the variability and interactions between rainfall, humans and fire enables the turnover in grass biomass (Bistinas et al. 2014) and the coexistence of trees and grass cover (Staver et al. 2011; Aleman and Fayolle 2020).

In large parts of Africa where there is less than 800 mm of rainfall annually, increased temperatures and drought results in less burnt area due to a decline in grass biomass and productivity (Alvarado et al. 2019). Therefore, not all ecosystems burn more when exposed to drought and high temperatures. In these arid regions of Africa, it is actually periods of increased rainfall that lead to higher fire activity due to greater accumulated biomass (van Wilgen et al. 2004). Furthermore, attempts to suppress fire can lead to biomass accrual and more severe fires. The pattern observed today of increased fire during wetter periods is also found in charcoal records (Daniau et al. 2013). Future changes in rainfall for Africa are uncertain, and global projections for rainfall and fire frequency will change due to increasing CO_2 and climate change (Hoffman et al. 2019). The effects of climate change on wildfires will therefore depend on the interaction between ignition conditions, fuel biomass accumulation, rainfall seasonality, and human management of fires.

Although the total area burnt and fire emissions decrease with drought in Africa's grassy ecosystems, extreme wildfire events do occur in African ecosystems and are associated with anomalous climate extremes, such as heatwave days and high-fire danger days that are projected to increase in frequency with global warming (Engelbrecht et al. 2015). Moreover, there is evidence that during these extreme conditions, forests – which are usually resistant to fire – can burn (Beckett 2018). This is exacerbated by the presence of alien vegetation that burns more intensely than indigenous vegetation. For example, in South Africa, one of the most damaging wildfire events on record occurred in the Knysna region of the Western Cape in 2017. It claimed 800 buildings, 5,000 hectares of forest plantations, and the lives of seven people (Kraaij et al. 2018). The severity of the fires was intensified by an unprecedented drought, the conversion of natural fynbos shrublands to timber plantations, and the invasion of alien trees, in combination with a history of fire suppression which resulted in fuel build-up. The management of invasive alien vegetation is crucial for the prevention of extreme wildfire events.

African savanna fires are fuelled by grass. However, the savanna biome is experiencing a rapid increase in woody plants (Hoffman et al. 2019). Woody encroachment is now widespread throughout sub-Saharan African savannas, as well as other parts of the world (Stevens et al. 2017). With an increase in global CO_2, tropical and subtropical savannas and grasslands are predicted to shift towards woody vegetation and associated changes in fire regimes (e.g., reduced area burnt) (Higgins and Scheiter 2012; Moncrieff et al. 2015). Therefore, both forest expansion and savanna thickening present significant challenges for the long-term management of protected areas in Africa (Jeffery et al. 2014). Feedback between vegetation structure (dense woody stands of trees versus grass-laden savannas) and rainfall and burnt area are important interactions (Alvarado et al. 2019) with management implications. Frequent early-season management burns are important in maintaining open canopy and preventing intense dry season burns, whereas infrequent, intense fires are important for managing woody encroachment or bush thickening in savannas (Smit et al. 2016).

Although communities have a limited ability to control global factors, they can manipulate fire and use this process to manage ecosystem services, including biodiversity, grazing, veld foods, and nature tourism potential. For example, the deliberate use of early dry season fires (April to July) by local communities has been shown to reduce and prevent the spread of the late hot dry season fires in West African savannas (Laris 2002) and in north-eastern Namibia's savanna-woodlands (Humphrey et al. 2020).

Case study: Australian 2019–2020 "Black Summer" fire season

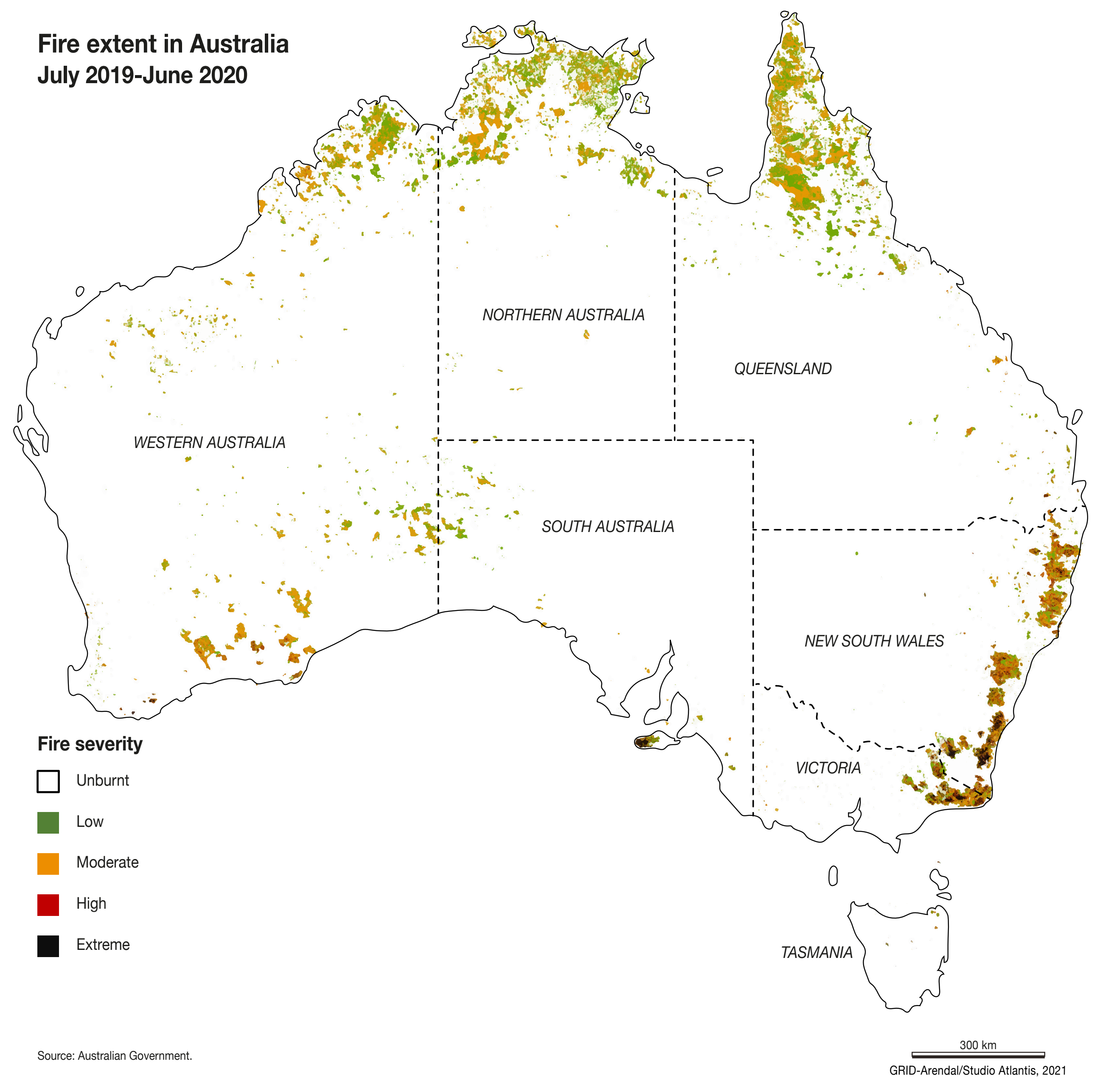
In Australia, the 2019–2020 fire season is widely considered one of the worst in recent memory. While there have been previous seasons where more fires occurred or more area was burnt (such as in 1974–1975; Luke and McArthur 1978), the 2019–2020 fire season is noted for the number of fires and area burnt and the direct impact these fires had on some of the most populated regions on the continent. This included extensive areas on the eastern seaboard (Figure 2.10), as well as some areas in southern and central eastern Australia.

While the fire season appeared to start earlier than expected, the timing of the commencement of many of the fires was not unusual when compared with long-term observations (Luke and McArthur 1978; Bowman et al. 2020b). Indeed, the only unusual aspect of the fire season was its rapid curtailment by broad steady rains across the south-east of the continent from mid-February that effectively extinguished all fire in the landscape.

What was different that season was the extensive reduction in rainfall across much of the continent over most of the preceding 12–18 months, followed by significantly warmer-than-average air temperatures during the fire season (with the annual national mean temperature 1.52°C above average). Annual national mean maximum temperature was the warmest on record (2.09°C above average) (King et al. 2020). The combination of extensive and extended rainfall deficit and higher than average temperatures preconditioned much of the landscape to easy ignition and rapid development of fire outbreaks.

During the fire season the almost continuous dry and hot weather conditions resulted in increased difficulty of suppression. Most fires were caused by natural ignition sources such as dry lightning. The frequent arrival of days of strong winds during this period resulted in an unusually high number of days of total fire ban. However, fire behaviour generally was not unexpected for the conditions during this period, with timely and effective warnings of potential fire spread issued by fire authorities. As a result, no members of the general public were killed due to a lack of warning of fire potential, although 33 people (including firefighters) were killed while attempting to directly control the fires (Filkov et al. 2020). Four hundred and twenty-nine premature deaths were attributed to indirect health impacts of the fires, many quite some distance from the fires themselves, including fatalities from cardiovascular and respiratory disorders (Johnston et al. 2021).

Fire behaviour was, for the most part, characterised by uncontrollable spread in mostly forested regions on the east coast. The lack of spread in grasslands and agricultural land was most likely a combination of reduced grass fuels due to reduced growth and overgrazing during the drought (Owens and O'Kane 2020) and the consequent greater ease of access for fire suppression by firefighters.



Source: Australian Government.

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Figure 2.10. Map of fire extent in Australia during the period 1 July 2019–30 June 2020. Data processed via GEOGLAM (http://earthobservations.org/geoglam.php) using the fire severity method of the New South Wales Department of Planning, Industry and Environment.

Case study: Arctic fire

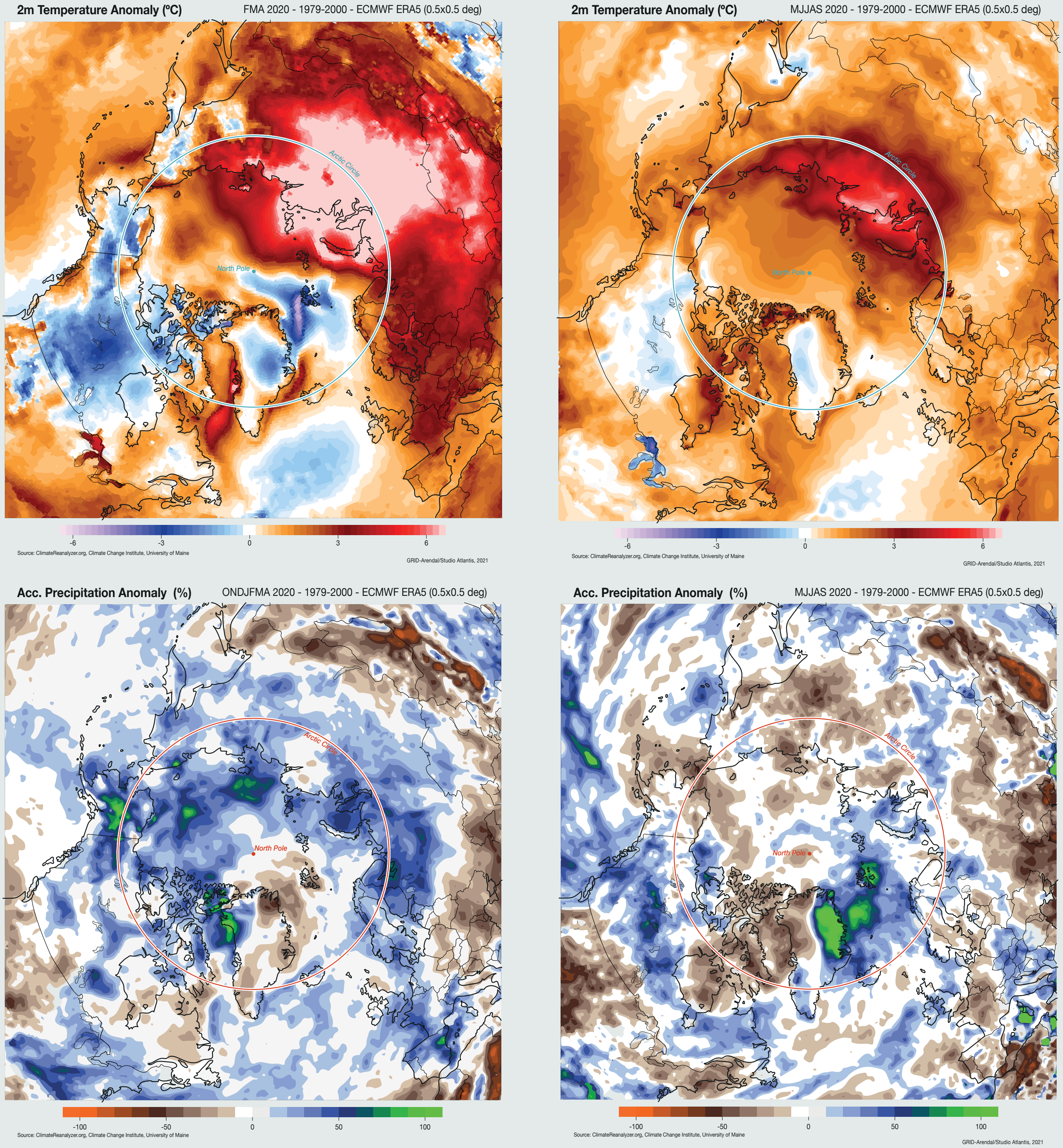
Wildfires in the Arctic have captured media attention in recent years as fire has become increasingly common in ecosystems that many do not think of as “fire prone”. However, evidence from the modern and palaeontological records indicates that northern boreal, tundra, and peatland systems burn under the right conditions. Tundra fires have occurred in the paleo record (Chipman and Hu 2017; York et al. 2018) and historical past (French et al. 2015; Hu et al. 2015) and tundra systems may burn frequently under the right conditions. Boreal forests are known from the paleo record to be flammable, but recent trends exceed rates of burning during the Little Ice Age (Kharuk et al. 2016) and even warmer times of the Holocene (Kelly et al. 2013). Decreased summer precipitation and increased summer temperatures increase the likelihood of years with high fire activity in tundra and high latitude boreal forests (Hu et al. 2015; Young et al. 2017; Masrur, Petrov and DeGroote 2018), and increased natural lightning ignitions (Veraverbeke et al. 2017; Hanes et al. 2019; Bieniek et al. 2020) have contributed to recent increases in area burnt by fire in the Canadian Arctic (Hanes et al. 2019). In contrast, economic development and related increased ignitions appear to be an important driver of the increasing area burnt in parts of north-west Siberia (Sizov et al. 2021). Analysis of these events strongly suggests climatic conditions govern the availability and flammability of fuels, and thus climate change can be expected to change Arctic fire regimes. Future climate changes are projected to become rapidly warmer and wetter on average, but inter-annual and regional variability, longer warm seasons, and the increased evapotranspiration associated with higher temperatures are likely to contribute to higher fire frequency and area burnt via direct effects on fuel flammability and availability (e.g., Flannigan et al. 2016). As extreme Arctic temperatures increase past historical thresholds, fire activity can be expected to increase, potentially very rapidly (Walsh et al. 2020). As an example, in 2020, fires in the north-central Russian Arctic burnt tens of millions of hectares. Figure 2.11 shows that the late winter/early spring and summer temperature anomalies in the region of these fires were 4–6°C above normal (1979–2000) and precipitation anomalies were below normal. While wind may have also been a contributing and related factor, these conditions indicate clear and ongoing regional changes like those described in Partain et al. (2016) for Alaska. The 2020 fire season caps a string of abnormally large high latitude fire years since the early 2000s, including but not limited to: 2019 (Alaska, Russia), 2018 (Fennoscandia, Russia), 2017 (Greenland), 2016 (Canada), 2015 (Alaska) and 2014 (Canada).

In recent years, the prevalence of latent winter fires (also called ghost or zombie fires because they can burn underground in peat soils undetected and emerge somewhere else weeks or even months later) in Arctic ecosystems has been well-documented (McCarty et al. 2020). As these fires have the potential to keep burning, they can flare up into wildfires when conditions are favourable (Figure 2.12).

Arctic wildfires have immediate to near-term impacts, including on local and regional air quality, carbon flux (see section 2.4), and vegetation changes or shifts, but the long-term effects on the cryosphere and ecosystems in a warming climate are complex. Air pollution from wildfires includes smoke, fine particulate matter ($PM_{2.5}$) and black carbon, or soot which can be transported long distances. These air quality problems have affected rural communities of the Arctic in recent fire seasons and will likely increase under climate change (Woo et al. 2020), but comprehensive regional analyses are lacking at this time. The increased deposition of black carbon on ice sheets and sea ice in summer and autumn could also potentially hasten melt as a result of increased heat absorption (Keegan et al. 2014).

Vegetation shifts in Arctic systems are implicated in increases in fire severity or frequency (e.g., Higuera et al. 2009; Higuera et al. 2011; Sizov et al. 2021). These increases, combined with climatic changes, can cause persistent energy balance changes that accelerate permafrost thaw (Jones et al. 2015; Ponomarev et al. 2020). Interactions among fire, permafrost thaw, and carbon fluxes (e.g., Estop-Aragonés et al. 2018; Walker et al. 2019) may increase the rate at which peatlands transition from carbon sink to carbon source. Feedback among temperature change, fire, permafrost thaw and atmospheric CO_2 in Arctic systems therefore have the potential to amplify regional changes to global impacts (Mack et al. 2011).

Taken as a whole, the current and projected changes in Arctic climate, ecosystems, fire regions, and socioecological systems suggest a future in which management of fire and adaptation to its impacts look fundamentally different than they did in the historical past. The immense and sometimes inaccessible areas are sparsely populated but are home to Indigenous peoples with strong cultural ties to the landscape and resources of the Arctic. Recent extreme fire years in all Arctic regions underscore the need for understanding the likely future impacts on the whole system, as well as the diversity of vulnerabilities, impacts and adaptive capacity across the Arctic.



2m Temperature Anomaly (ºC) FMA 2020 - 1979-2000 - ECMWF ERA5 (0.5x0.5 deg)

2m Temperature Anomaly (ºC) MJJAS 2020 - 1979-2000 - ECMWF ERA5 (0.5x0.5 deg)

Acc. Precipitation Anomaly (%) ONDJFMA 2020 - 1979-2000 - ECMWF ERA5 (0.5x0.5 deg)

Acc. Precipitation Anomaly (%) MJJAS 2020 - 1979-2000 - ECMWF ERA5 (0.5x0.5 deg)

Source: ClimateReanalyzer.org, Climate Change Institute, University of Maine

GRID-Arendal/Studio Atlantis, 2021

Figure 2.11. Temperature and precipitation anomalies in the Arctic. Upper left: February to March 2020 temperature anomalies. Upper right: March to September 2020 temperature anomalies. Lower left: October to March 2019–2020 precipitation anomalies. Lower right: May to September 2020 precipitation anomalies. All relative to 1979–2000 norms. Also available from Climate Reanalyzer (https://ClimateReanalyzer.org), Climate Change Institute, University of Maine, USA.)

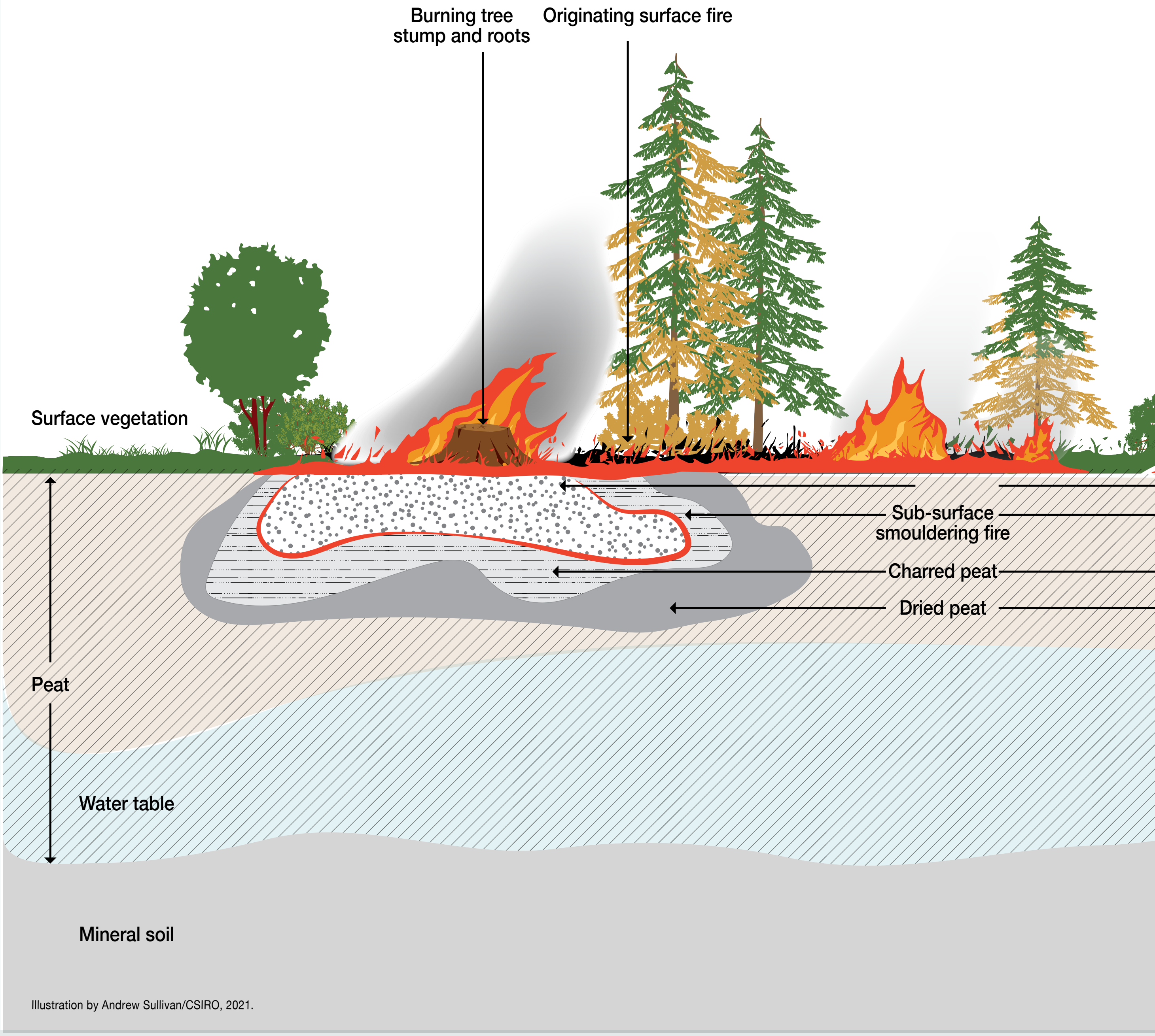
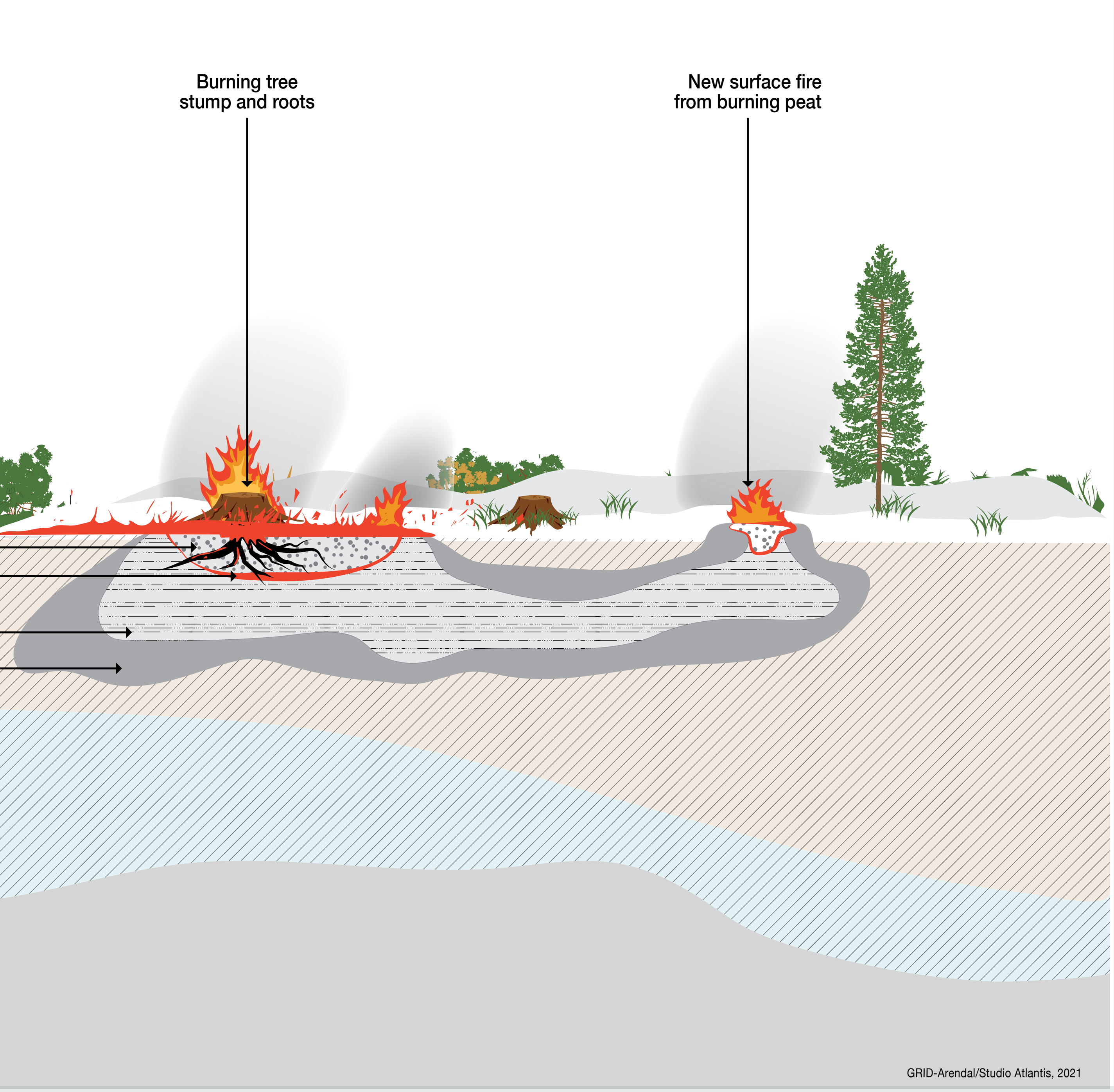

New surface fires from sub-surface fire propagation in peatlands

Illustration by Andrew Sullivan/CSIRO, 2021.

Figure 2.12. Surface fires burning in peatlands can transition to underground smouldering fires that can seem to self-extinguish only to reappear in another location, sometimes at a great distance from the originating fire. The primary combustion reaction is charring which is highly exothermic but does not require oxygen to sustain (Sullivan 2017). New fires can resurface anywhere from days to months after the original fire appears extinguished, even continuing to smoulder through the winter in the peat layer under layers of snow, leading to the name "zombie" or "ghost" fires.



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Chapter 3 — Impacts of wildfires on people

3. Introduction

Wildfires can threaten lives and livelihoods, affect national economies, and have other potentially long-lasting impacts on people. In addition to the potential loss of human life (see Portugal case study for an example), wildfires can cause acute and chronic health issues, destroy infrastructure, and degrade ecosystem services, such as water supply, nutrition, biodiversity, and carbon storage. In developing countries, an increase in damaging wildfires may reverse or delay progress towards the United Nations Sustainable Development Goals (SDGs, Figure 3.1), Paris Agreement and Sendai targets.

Impacts of wildfire on Sustainable Development Goals

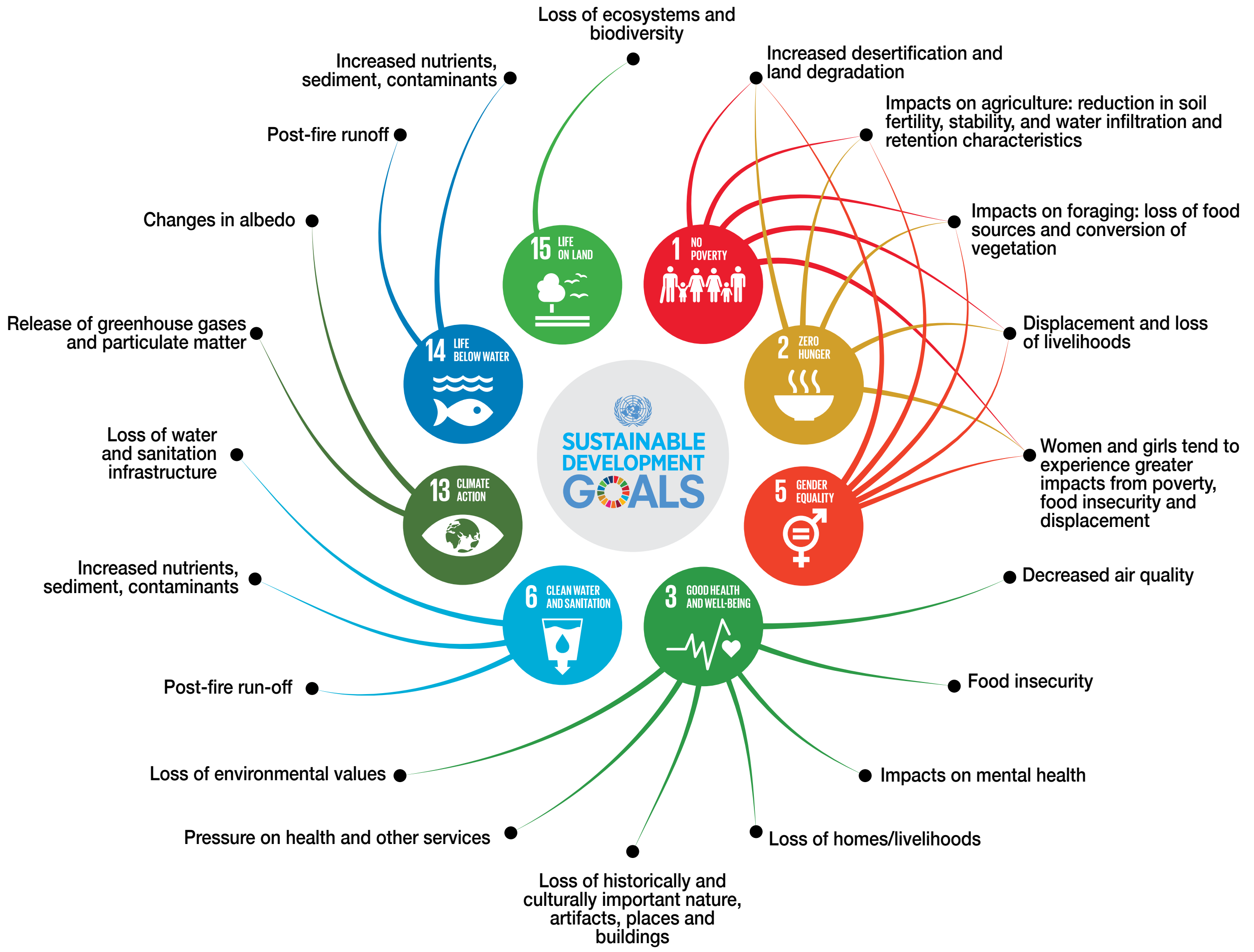

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Figure 3.1. Impacts of wildfire on the United Nations Sustainable Development Goals (SDGs). The changing scale and intensity of wildfires may impact achievements across several of the SDGs that impact human health and well-being (Martin 2019).

Case study: Portugal's 2017 firestorms and human life loss

Two unprecedented fire events took place in Portugal in 2017. That year, winter and spring had been unusually dry and included record-breaking heat waves that decreased vegetation moisture to unusual levels (Sánchez-Benítez et al. 2018; Turco et al. 2019). The first wildfire event occurred in Pedrógão Grande on 17 June. During a time of extreme fire weather conditions, two separate ignitions merged. Following a shift in wind direction and velocity caused by a gust front from a nearby thunderstorm, the fire escalated (up to an average rate of spread of 5 km h^{-1}; Guerreiro et al. 2017; Pinto et al. 2018). The resulting energy output combined with the highly unstable atmosphere to produce a large volume of uplifted air, forming a pyrocumulonimbus (pyroCb) cloud that reached 13 km in height. PyroCb are associated with strong inflow winds, vorticity (strong rotation), lightning and erratic, extremely dangerous fire behaviour, including profuse long-range spotting (Werth et al. 2016). In the early evening, a strong wind downburst occurred. This accelerated the fire spread resulting in the death of 66 people (31 female and 35 male), most of whom were in the process of fleeing from the fire (Haynes et al. 2020).

The second wildfire event occurred on 15 October. At the time, fuel dryness was at an historic low due to the nearly total absence of rain in the period since the Pedrógão Grande firestorm. As tropical storm Ophelia approached the Portuguese coast, advecting warm and dry air from northern Africa, several long and narrow wind-driven fires developed from south to north, matching or even exceeding the rate of spread observed in Pedrógão Grande (Castellnou et al. 2018). Fifty-one human fatalities (17 female and 34 male; men are often over-represented in fire fatalities, see section 3.3) resulted from eight fires (each greater than 10,000 hectares in size), and more than 200,000 hectares burnt in less than 24 hours – an unparalleled event in Europe. Additionally, 138 deaths were attributed to exposure to smoke from the October 2017 wildfires (Augusto et al. 2020).

These two fire events occurred in a context of moderately high population density, scattered in a flammable landscape. Contributing factors in the region include changing the land use and land cover trends over the last few decades, which have resulted in a decrease in farmland, the expansion of poorly managed forest plantations and the recovery of natural vegetation. The Mediterranean climate, with wet winters and dry summers, exacerbates fire hazard by favouring substantial biomass growth followed by its desiccation. These conditions are similar to those that prevailed during extreme wildfire events elsewhere, namely California and south-eastern Australia (Bowman et al. 2017), and Chile (see case study in chapter 2).

3.1 The costs of wildfire

Although the annual number of damaging wildfire events is small compared with the total number of fires, they are an enormous challenge for societies. They exceed the limits of suppression and therefore represent a heightened threat to firefighters, populations, assets, and natural values. Reports and news media emphasize visible wildfire damage and loss, while readily available data tends to focus on damage to human assets and often does not cover the costs related to death and injury, health impacts, ecosystem services, or firefighting (Doerr and Santín 2016). There is limited information about the global socioeconomic and environmental cost of wildfires over both the short and long term, but local estimates of costs suggest they are significant (Bowman 2018; Johnston et al. 2021).

Wildfires can cause damage to infrastructure including power and communication lines, water supply, roads, and railways. There can also be huge clean up and rebuilding costs after a major fire event. Changes in the availability of fresh water and degradation of water supplies as a result of wildfires in watersheds can be costly and have the potential to affect large numbers of people (see section 3.4).

A fire can force business closures and disrupt transport and supply chains. This can decrease tax revenues and affect property values. Interruptions can result in customers taking their business elsewhere, causing further losses to local businesses. Whole communities can be impacted if workers are laid off and decide to relocate. Damaging wildfires pose risks to banks and insurers who can incur significant losses from catastrophic events.

The amount of money spent each year globally on fire management has been increasing in recent years. In the lead-up to the 2021 fire season in Chile, the President unveiled the National Forest Fire Prevention Plan, a mix of government and privately funded firefighting resources with a value of US$180 million – a record amount for the country

Fire-related loss – Brazilian Amazon

An early study on economic loss due to fires in the Brazilian Amazon was undertaken by da Motta et al. (2002). The study covered the period from 1996 to 1999, a period which included a severe drought in 1997–1998. The authors estimated that the destruction of pasture, fences, forests, and impacts on human health related to increased particulate matter resulted in annual average costs of over US$100 million (nearly 9 per cent of the region's GDP).

More recent estimates have been carried out at local scales. For example, Brown et al. (2006; 2011) estimated that during the 2005 drought in the state of Acre in the south-western Brazilian Amazon, over 300,000 hectares of forests burnt and more than 400,000 people were affected by fire-related air pollution. In the southern Amazon, fires have been shown to cause an increase in hospital admissions for respiratory disease in children and the elderly (Ignotti et al. 2010; Do Carmo et al. 2013; Machado-Silva et al. 2020).

Butt et al. (2020) looked at the health impact of fine particulate matter from vegetation fires across the Amazon Basin. They focused on 2012 (a year with emissions similar to the ten-year average between 2008 and 2018). They found that preventing fires during that year would have avoided 16,800 premature deaths and 641,000 disability-adjusted life years across South America. Direct losses amounted to more than US$50 million, and indirect losses (economic, social, and environmental) reached US$ 100 million. Campanharo et al. (2019) looked at the same region during 2008–2018, a period which included a severe drought in 2010. They estimated losses of US$243.36 million (± 85.05) for the drought year and US$307.46 million (± 85.41) for the whole period (Campanharo et al. 2019). These values represent a significant loss to the region, about 7 per cent of GDP.

Fire-related loss – Indonesia

Since the early 1980s, Indonesia has experienced increasing fire risk and fire incidence, associated with forest degradation, the expansion of commodity crops, including oil palm, and drainage of fire-prone peatlands. Extensive forest fires in 1997–1998 that resulted in forest loss were estimated to have cost in the range of US$1.62–2.7 billion and smoke haze pollution added an additional US$675 million (Tacconi 2003). Seven years later in 2015, economic losses from the severe fires escalated to US$16.1 billion, which is equivalent to 1.9 per cent of the country's GDP (Glauber et al. 2016). Measurable costs included fire-related damage to timber and agriculture, including loss of plantation crops, and from smoke-related impacts on tourism and transport. In just 2 months, from September to October, the transportation sector lost an estimated US$372 million. Smoke contributed to the death of 19 people and more than 500,000 acute respiratory infections - with immediate health costs totalling US$151 million (Glauber et al. 2016). There may also be other longer-term impacts, such as education outcomes, as 5 million children missed school due to closures during the year. Although not as catastrophic as 2015, fires in 2019 caused significant economic losses, estimated at US$5.2 billion or 0.5 per cent of GDP (World Bank 2020).

(Chile 2020). Annual firefighting expenditure by USA federal agencies has increased to US$1.9 billion (a rise of more than 170 per cent in a decade; National Interagency Fire Center 2021). In Canada, total annual expenditure for national wildland fire management activities ranges between CAD 500 million and CAD 1 billion, an increase of about CAD 120 million per decade since the 1970s (Hope et al. 2016; Stocks and Martell 2016). In many regions the actual cost of suppression is difficult to estimate due to extensive reliance upon volunteer firefighters. While much of the increase in fire management expenditure is due to increased need for wildfire suppression, increases in the cost of suppression resources also plays a significant role. In particular, the reliance on large and expensive aircraft for direct and indirect attack is a relatively recent development that has greatly increased fire management expenditure in some locations (Plucinski 2019).

3.2 Global health impacts of wildfires

Wildfire smoke is chemically complex, comprising a range of potentially toxic combustion products and fine particulate matter (Groot et al. 2019). The composition of wildfire smoke, and resulting lung toxicity and other human health effects, depends on factors such as fire behaviour, vegetation type, season, burn conditions, available fuel, combustion phase and exposure (Kondo et al. 2019; Figure 3.2). Particulate matter (especially < 2.5 µm, known as $PM_{2.5}$) is the major smoke product of concern, with daily levels during wildfires often exceeding safe levels as recommended by air quality guidelines (for example, the World Health Organization Global Air

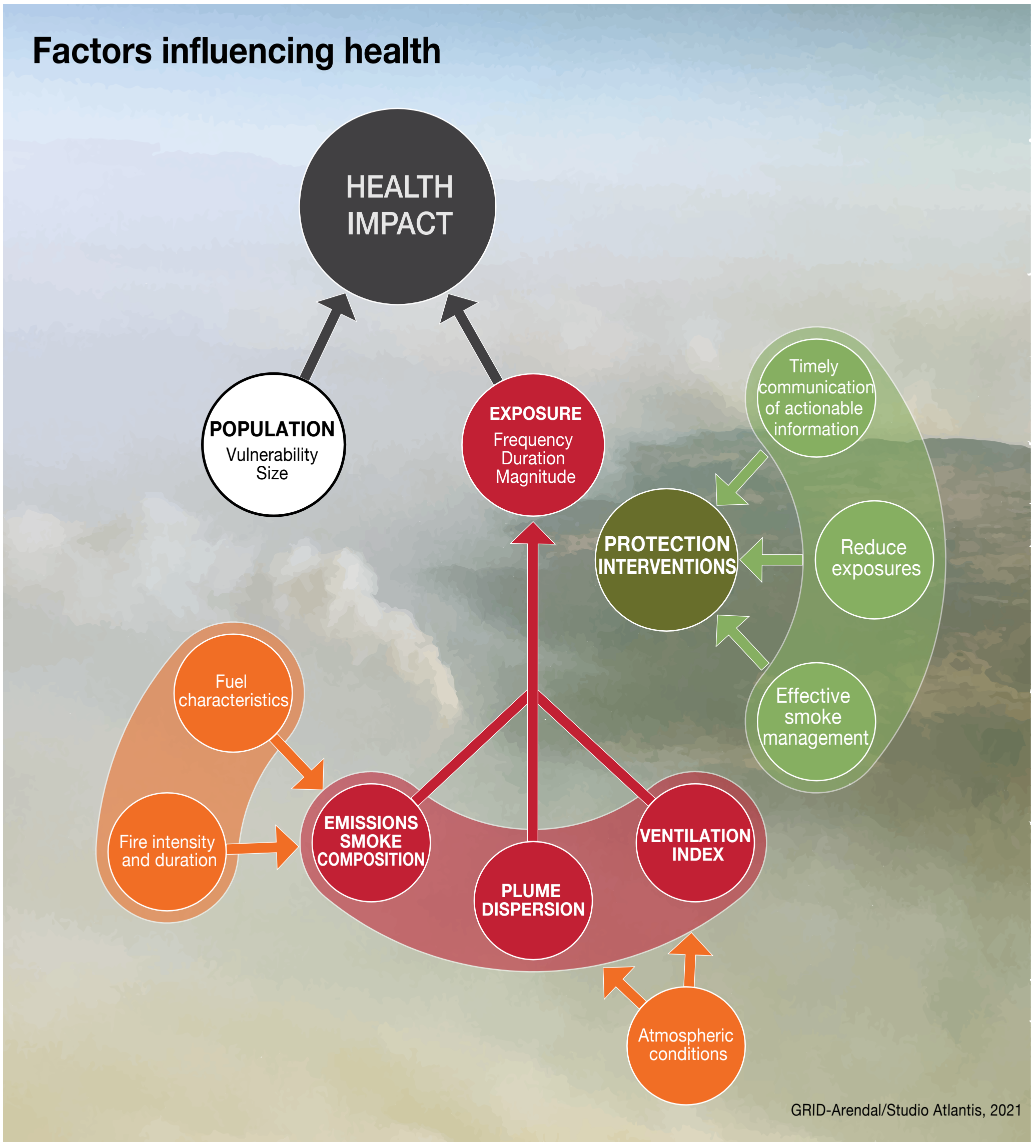

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Figure 3.2. Wildfire smoke contains fine particulate matter and potentially toxic combustion products (the latter can be particularly harmful at the wildland-urban interface where waste and rubbish, materials used in buildings and vehicles are often burnt; Hallema et al. 2019).

Quality Guidelines 2005; Figure 3.3). Epidemiological studies consistently show associations between exposure and adverse respiratory health outcomes (Liu et al. 2015; Kondo et al. 2019) and there is also growing evidence of adverse cardiovascular effects (Jones et al. 2020).

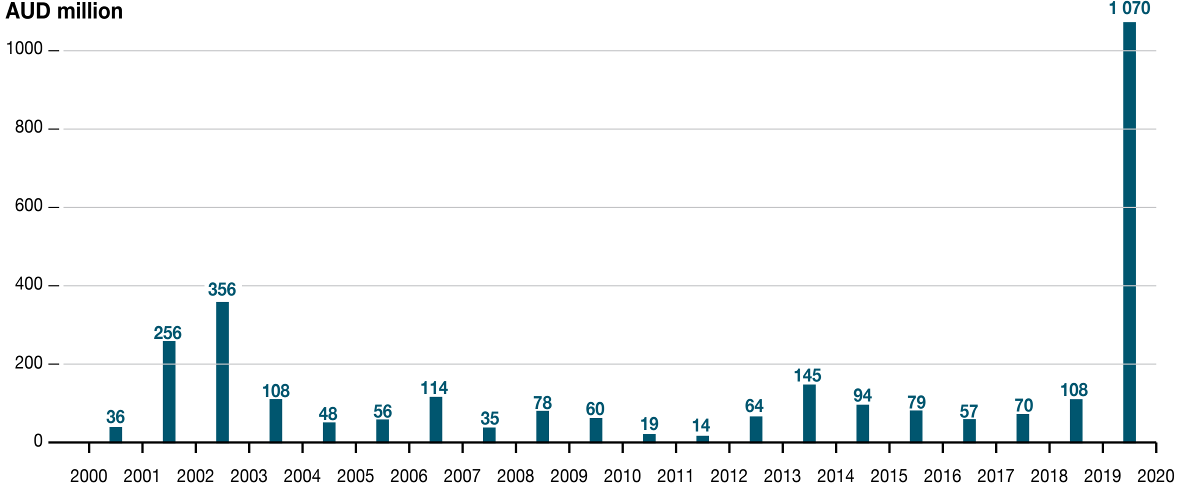
Occupational exposure in firefighters occurs while fighting wildfires and during prescribed burns (Figure 3.4). Risks are heightened due to the combination of exposure to high smoke concentrations and strenuous physical activity (which increases respiratory and heart rates; Groot et al. 2019). Respiratory effects include small, but statistically significant, declines in lung function. Hypertension is positively associated with firefighter career length (Groot et al. 2019). Significant adverse associations have been reported between wildfires and systemic inflammation (Huttunen et al. 2012), bone marrow content (Tan et al. 2000), lung cancer (Navarro et al. 2019), and physical strength and overall health (Frankenberg et al. 2005). Other occupational effects include mental health issues, with some firefighters showing chronic post-traumatic

→

Figure 3.3. The top panel illustrates the historical $PM_{2.5}$ fire smoke-related health costs and the health costs incurred during the Australian 2019–2020 Black Summer fires (more than AUD 1 billion). The middle panel shows the average annual level of exposure of New South Wales's population to $PM_{2.5}$ fire smoke. The bottom panel shows a more detailed breakdown of $PM_{2.5}$ smoke exposure during the Black Summer fires. On many occasions the population experienced concentrations of $PM_{2.5}$ smoke above the Australian national ambient air quality standard of 25 µg/m^3 – 24-hour $PM_{2.5}$ (Arriagada et al. 2020).

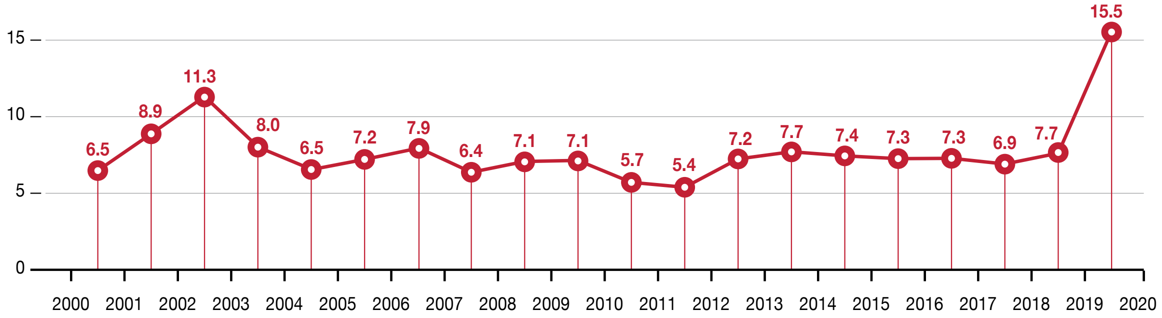
Historical wildfire impacts in New South Wales, Australia

Fire smoke $PM_{2.5}$ related health costs



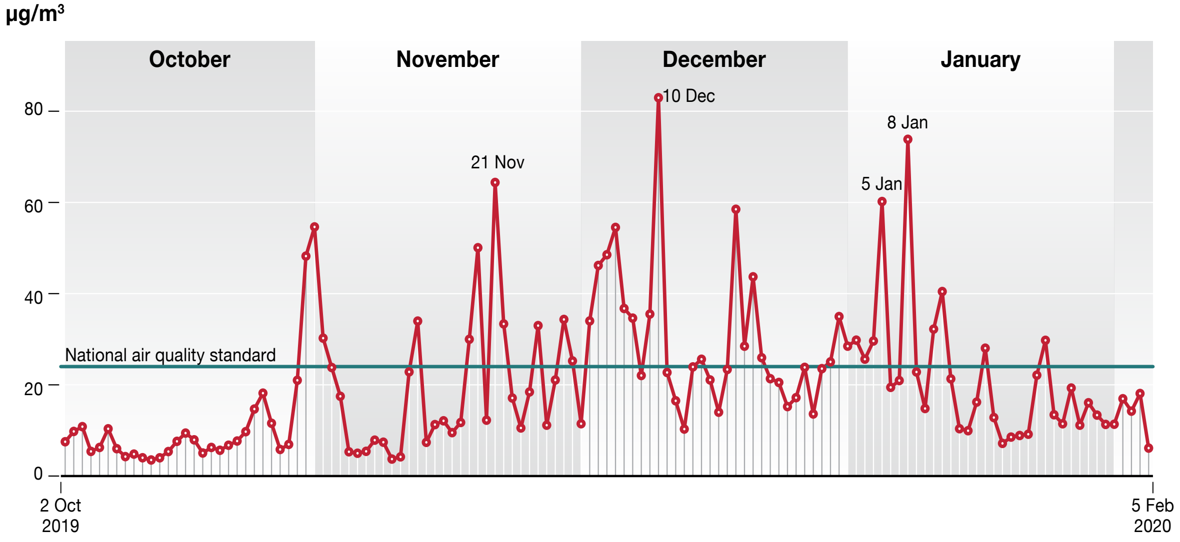
Median population-weighted fire smoke $PM_{2.5}$

µg/m³ Population weighted $PM_{2.5}$ is a measure of the average level of exposure of a population to particles with an aerodynamic diameter less than 2.5 microns ($PM_{2.5}$), otherwise known as fine particulate matter. These particles, being so small, are capable of entering the respiratory system and bloodstream, causing different adverse health effects. $PM_{2.5}$ exposure levels estimated across the state are aggregated into one measure (population-weighted PM2.5) by estimating a weighted mean where larger populations have higher weights (importance) than smaller ones.



Median population-weighted fire smoke $PM_{2.5}$

2019-2020 fire season



Source: Arriagada et al. 2020.

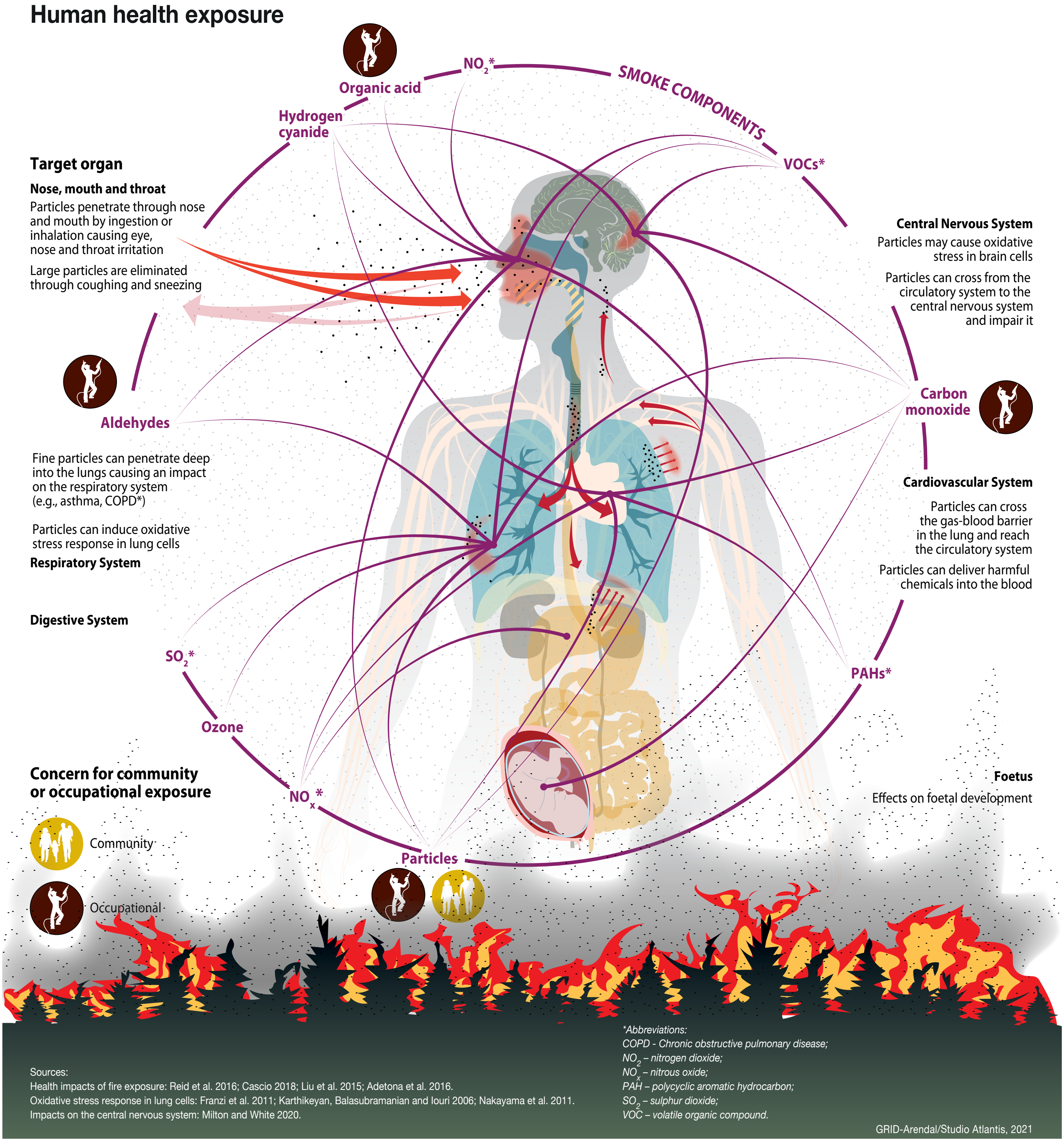
GRID-Arendal/Studio Atlantis, 2020

stress symptoms. These can be more prevalent in younger, seasonal firefighters with less experience and therefore higher anxiety. Symptoms have been found to persist for up to seven years (Groot et al. 2019).

When it comes to impacts on the health of the general public, Australia and the USA are the most frequently studied countries, with most studies focusing on areas close to fires (Figure 3.3; Liu et al. 2015). Like occupational exposure, respiratory effects are the most frequently and consistently observed impacts. Studies indicate that wildfire smoke is significantly associated with risk of respiratory morbidity. Respiratory morbidities are more frequently observed in people middle-aged and older (Henderson et al. 2011). The risk of respiratory-related hospital visits is also higher among children (under five years old) compared with other age groups (Liu et al. 2015). There is evidence that exposure to wildfire smoke during pregnancy (especially the last trimester) is linked to an increased risk of pre-term birth (Heft-Neal et al. 2022).

In addition to respiratory morbidity, wildfire exposure is possibly associated with all-cause mortality (Reid et al. 2016; Groot et al. 2019). There are a significant number of epidemiological studies that demonstrate a link between air pollution and cardiovascular-associated morbidity and mortality. However, the relationship between wildfire smoke and cardiovascular-related illness is less clear (Jones et al. 2020). To date, the most consistently observed impact appears to be the link between wildfire smoke and out-of-hospital cardiac arrest (Dennekamp et al. 2015; Haikerwal et al. 2015; Ho et al. 2018; Jones et al. 2020). Shorter total sleep times and sleep disruption have also been reported (Rifkin et al. 2018).

Smoke associated with deforestation fires in the Brazilian Amazon has been found to be responsible for the premature death of almost 3,000 people annually (95 per cent percentile confidence interval: 1,065–4,714), demonstrating the regional scale of fire impacts (Reddington et al. 2015). A comprehensive analysis from 2001 to 2016 showed an increase of 27 per cent in respiratory disease hospitalizations related to drought and fire occurrence in the state of Roraima, located in the south-western Brazilian Amazon (Machado-Silva et al. 2020). A recent report assessing the 2019 Amazonian fires estimated 2,195 hospitalizations due to respiratory diseases during that year, with 21 per cent being children under one year old and 49 per cent over 60 (Amazon Environmental Research Institute [IPAM] 2020). In 2020, 141,055 fires were detected in the Amazon – an increase of 24 per cent over the previous year (13,620 fires; Centro de Previsão de Tempo e Estudos Climáticos/Instituto Nacional De Pesquisas Espaciais [INPE/



Target organ

Nose, mouth and throat

Particles penetrate through nose and mouth by ingestion or inhalation causing eye, nose and throat irritation

Large particles are eliminated through coughing and sneezing

Fine particles can penetrate deep into the lungs causing an impact on the respiratory system (e.g., asthma, COPD*)

Particles can induce oxidative stress response in lung cells

Respiratory System

Digestive System

Central Nervous System

Particles may cause oxidative stress in brain cells

Particles can cross from the circulatory system to the central nervous system and impair it

Cardiovascular System

Particles can cross the gas-blood barrier in the lung and reach the circulatory system

Particles can deliver harmful chemicals into the blood

Foetus

Effects on foetal development

Concern for community or occupational exposure

Sources:
Health impacts of fire exposure: Reid et al. 2016; Cascio 2018; Liu et al. 2015; Adetona et al. 2016.
Oxidative stress response in lung cells: Franzi et al. 2011; Karthikeyan, Balasubramanian and Iouri 2006; Nakayama et al. 2011.
Impacts on the central nervous system: Milton and White 2020.

**Abbreviations:*
COPD - Chronic obstructive pulmonary disease;
NO$_2$ – nitrogen dioxide;
NO$_x$ – nitrous oxide;
PAH – polycyclic aromatic hydrocarbon;
SO$_2$ – sulphur dioxide;
VOC – volatile organic compound.

GRID-Arendal/Studio Atlantis, 2021

Figure 3.4. Smoke particulate exposure pathways and impacts. Smoke exposure is most commonly measured from land-based air pollutant monitors, followed by satellite-based imagery models, with fewer studies measuring personal exposure to smoke (Liu et al. 2015).

CPTEC] 2021). The poor air quality compounded the respiratory stress of COVID-19 patients and put additional pressure on the health system. It has been reported that Indigenous peoples are particularly vulnerable with COVID-19 mortality rates 1.5 times the Brazil-wide average (Fellows et al. 2020). Similarly, it has been suggested that smoke pollution from tropical peatland fires can increase the vulnerability of populations exposed to COVID-19 (Harrison et al. 2020). Fire disturbance in peatlands, which are home to many potential animal vectors, could also encourage the emergence of further zoonotic infectious diseases like COVID-19 (Harrison et al. 2020).

A wildfire event can cause feelings of confusion, anger, fear, and loss, all of which can have long-term impacts on the people and communities that experience them. Loss, not just of property such as houses, but also pets and valued objects, can devastate individuals and communities. There are indications that people can also suffer psychological distress from losing a valued landscape due to a wildfire (“solastalgia”; Eisenman et al. 2015) or as a result of long periods of smoke exposure (Dodd et al. 2018; Burkhardt et al. 2020).

3.3 Wildfire risk perception and impacts on the built environment

There are several actions that people can take during a fire, but the main options are either to evacuate, or to stay and take appropriate action while the fire passes (sheltering or active defence). A significant number of wildfire fatalities tend to be from people trying to evacuate late – just prior to, during or even after the fire has arrived (Blanchi et al. 2014; Haynes et al. 2019; Molina-Terrén et al. 2019). Europe and the USA generally favour mass evacuation protocols, with the assumption that it is the most effective means of protecting life, whereas in the past Australia, recognizing the potential dangers of evacuation and potential benefits of individuals defending their property, developed a policy that left the decision to evacuate primarily up to the individual (McCaffrey and Rhodes 2009). However, following the Black Saturday fires in southern Australia in 2009, in which 173 people died (mostly within or adjacent to their homes), messaging shifted to a general leave-early recommendation (i.e., prior to a fire outbreak based on forecast fire potential), although the general policy remains in place.

Research in Australia suggests that men are more likely than women to remain and defend property during a wildfire (e.g., Haynes et al. 2010). This gendered response is reflected in fatalities, with men more likely to die protecting assets while women and children are more likely to perish inside or while trying to escape (Haynes et al. 2010; Eriksen 2013). A recent study in the USA found that the decision to stay and defend was influenced by preparedness, with residents who have implemented fuel reduction and fire mitigation strategies more likely to stay (Stasiewicz and Paveglio 2021).

When a wildfire spreads into an urban area, houses, the surrounding infrastructure (e.g., fences, retaining walls, cars, and sheds) and gardens become the dominant wildfire fuel and the main source of embers that ignite adjacent structures (Cohen 2000). Building vulnerability depends on its design, construction, use of material and maintenance, and the condition of the surrounding area. In many cases, common housing construction techniques are not specifically built with fire in mind and can contribute to an increased risk (Blanchi et al. 2012). Mechanisms by which wildfire ignites buildings include direct flame contact, radiant and convective heat, and burning debris (the dominant mechanism, alternatively referred to as "firebrands" or "embers" – Figure 3.5).

3.4 Wildfire impacts on water security

Forest ecosystems are an essential part of the water cycle. By absorbing (i.e., sponge effect) and filtering water, they regulate the delivery of good-quality water to aquatic ecosystems and downstream communities (e.g., Ellison et al. 2012). Forests thus sustain several direct and indirect downstream water uses, from drinking water and irrigation to fishing, recreation and flood mitigation, among others (Brauman 2015; Jenkins and Schaap 2018; United Nations Economic Commission for Europe [UNECE] and Food and Agriculture Organization of the United Nations [FAO] 2018). Worldwide forest degradation,

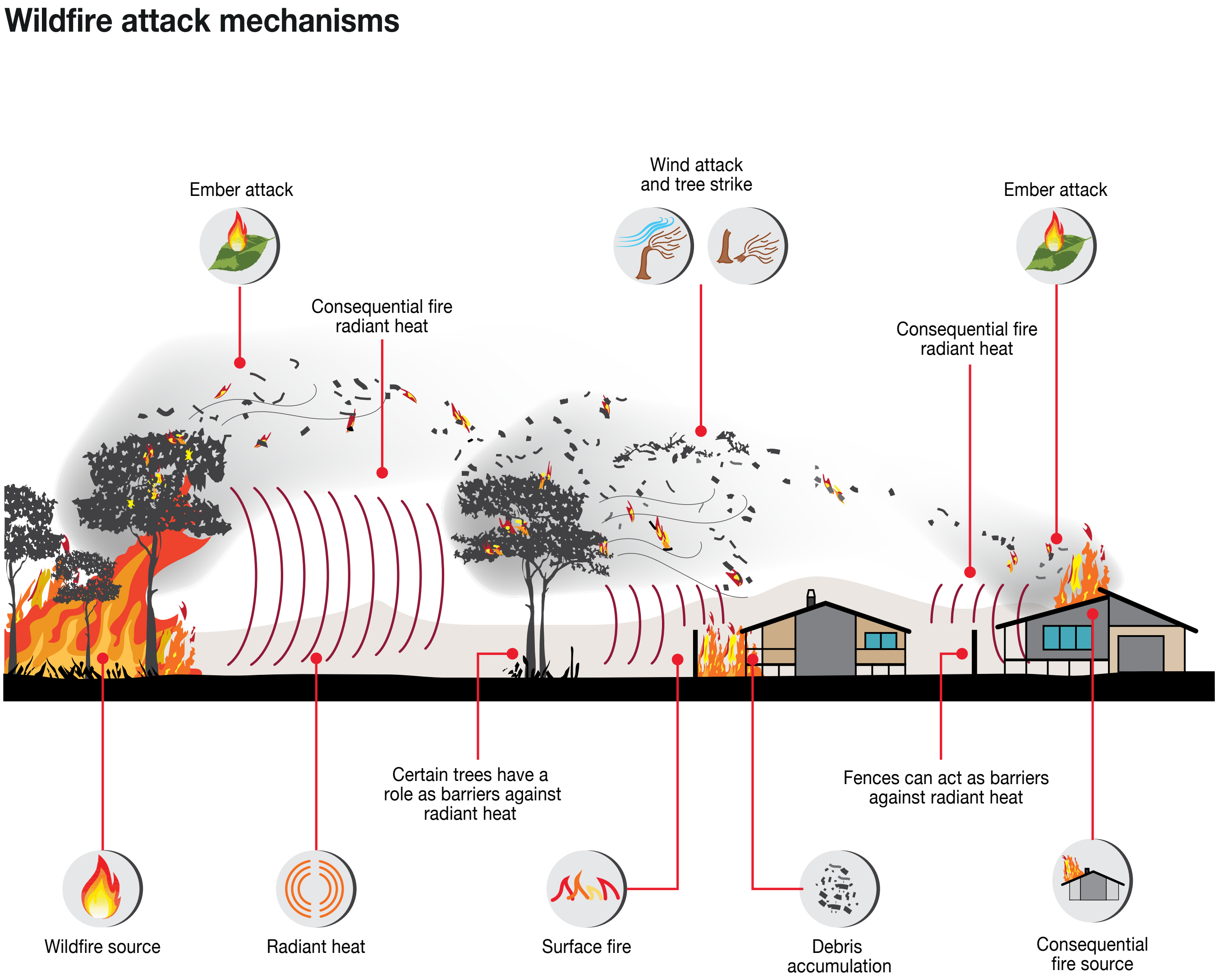


Illustration by Raphaele Blanchi/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

Figure 3.5. Wildfire attack mechanisms.

extreme climate events, and human development already put watershed health at risk, and water resources for billions of people are now threatened by changing fire regimes (Hallema et al. 2018; Robinne et al. 2018; Swain et al. 2020; Robinne et al. 2021).

After high-severity fires, the sponge and filter effects regulating downstream water quality and availability become limited (Shakesby and Doerr 2006) due to the combustion of vegetation (Figure 3. 6). Water from rainfall or snowmelt, or both, runs faster on bare, unstable ground, thereby enhancing its erosive power. This combination of higher water velocity and enhanced erosion commonly lead to sudden changes in water quantity and quality in rivers and lakes downstream of burnt areas, particularly after extreme precipitation events (Figure 3.7) (Shakesby and Doerr 2006; Bodí et al. 2014; Dahm et al. 2015; Moody et al. 2015). Such changes can lead to cascading effects on aquatic life due to the increase in nutrient concentrations and temperature, and the decrease in light and dissolved oxygen. It is not uncommon to see a sudden drop in fish populations post-fire (Bixby et al. 2015; Robinne et al. 2020; Silva et al. 2020b). At the same time, if there is some level of connectivity between populations, aquatic populations usually recover within a few years of single fires (Dunham et al. 2003).

When wildfires burn human settlements, toxicants derived from plastic (such as benzene) or industrial leachates (such as mercury or arsenic) can also accumulate in run-off and threaten aquatic life and human health (Burke et al. 2013; Murphy et al. 2020; Proctor et al. 2020). Pregnant women should take precautions as there is a risk of foetal exposure. Breastfeeding infants are also vulnerable to toxicity (Axelrad et al. 2007). These effects might last from a few years after the fire to several decades depending on the indicator of hydrologic alteration (for example, nutrient concentration or summer low flows) and the social, economic, and ecological settings of the area of interest (Feikema, Sherwin and Lane 2013; Niemeyer, Bladon and Woodsmith 2020).

The sudden increase in run-off and erosion that can occur after a wildfire can lead to sediment-laden flows that threaten human life (Figure 3.8) and infrastructure damage. Such impacts have significant social and economic costs. Massive

Figure 3.6. Water runs faster on bare, unstable ground in severely burnt forests, increasing soil erosion

Figure 3.7. Increased turbidity is one of the changes in water quality downstream of burnt areas.

sediment influx can reduce reservoir lifespan, and sediment dredging is costly (for example, it costs US$60 million to dredge the Strontia Springs reservoir that supplies the city of Denver, Colorado – Bladon et al. 2014). Changes in the timing and the range of high and low flows also pose challenges for reservoir management yet impacts on the efficacy of flood control and power generation remain to be documented. Comparable challenges for drinking water supply also arise, with other specific concerns related to changes in water quality and the efficacy of the water treatment process (Hohner et al. 2017; 2019). In places without access to improved and/or clean water sources for drinking and hygiene needs, vulnerability to post-fire water pollution might make the water unfit for consumption, thereby further compromising water security.

While there is a major focus on wildfire-watershed risks to large cities, they are likely less vulnerable to post-fire water issues than smaller communities – often rural or disenfranchised, or both – that usually have neither the financial nor technological resources to deal with the risk and the aftermath of a disaster (Abell et al. 2017; Hoekstra et al. 2018).

Figure 3.8. Heavy rains one month after a series of major wildfires in 2018 resulted in a mudslide in Montecito, California that caused 23 deaths.

Chapter 4 – Impacts of wildfires on the environment

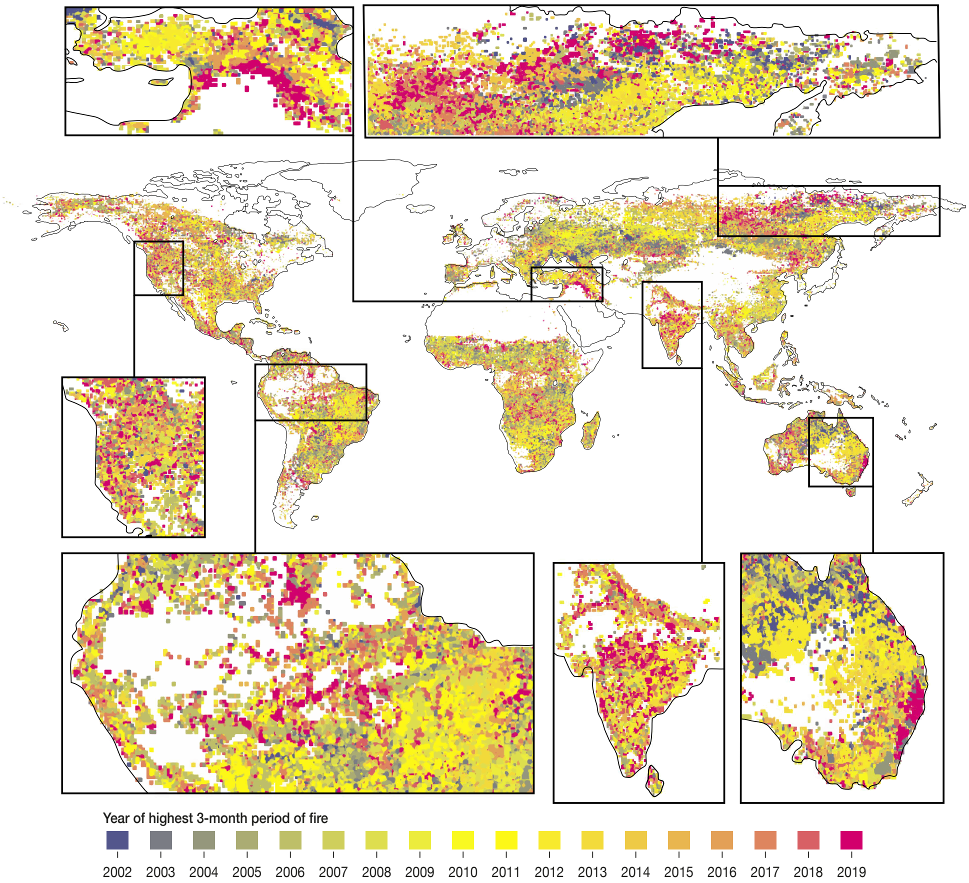
4. Introduction

Fire is an important factor shaping global ecosystems. It can affect vegetation structure at multiple scales (landscape, community, and individual plants) and to different degrees, depending on the fire regime, climatic conditions pre-and post-disturbance, and ecosystem type (Keith et al. 2009). In many fire-adapted biomes, fire has an essential role in maintaining ecosystem health, vegetation structure, biodiversity, reproduction, and ecosystem function. In dry, fire-prone regions, fire can play a major role in the decomposition of biomass (e.g., Throop et al. 2017; van Wagtendonk et al. eds. 2018) and creation of refugia for fauna. Many open-canopied woodlands, grasslands and sedgelands are maintained by frequent fire (Jackson 1968; Peterson and Reich 2001; Bond et al. 2005; Burton et al. 2019), sometimes in concert with other factors such as animal browsing (van Langevelde et al. 2003).

However, multiple fires at short intervals can lead to short- and long-term impacts on biodiversity that can be negative or positive, depending on the historical fire regime, and the extent and intensity of each fire (Haslem et al. 2011; Hradsky et al. 2017). Some ecosystems tolerate or even require frequent fires (once every 1–10 years) with little long-term change in the composition of species (e.g., some fire-prone Mediterranean ecosystems of south-western Australia and tropical and temperate savannas), whereas a high frequency of burning can have a catastrophic effect on biodiversity in fire-sensitive ecosystems (e.g., old-growth montane forests; Bradshaw et al. 2018). Ecosystems that now burn more frequently than they did historically include 70 per cent of the world's fire-sensitive tropical habitats (Shlisky et al. 2009), the North American steppe (Davies 2011), and many peri-urban areas (Keeley et al. 1999).

Figure 4.1 shows a global map of the year in which the highest three-month moving average period of fire occurred (2002–2019), broken down by biome. For many of the most fire-prone savanna regions, the highest three-month period of fire occurred more than 10 years ago, and they are now experiencing a decline in burnt area (chapter 2). Many regions experiencing their highest three-month period of fire more recently are not typically associated with frequent wildfire occurrence (for example, central Amazonia, India, and the Middle East). Some of these are fire-prone areas where recent fire events are much more extensive than typically experienced (the Siberian Arctic, south-eastern Australia, and south-western USA). Increased fire frequency is especially damaging for long-lived plants which require decades without high-intensity disturbance to mature and reproduce (Bradstock et al. 1996; Tulloch et al. 2016). For example, increased fire frequency and fire season length in the Amazon (Aragão et al. 2018) has led to an approximately 25 per cent decrease in tree biomass and carbon stocks (Silva et al. 2020a).

Year of highest three-month period of fire



Sources: Chuvieco et al. 2018; Laboratório de Aplicações de Satélites Ambientais [Laboratory for Environmental Satellite Applications – LASA], 2020; Libonati et al. 2020.

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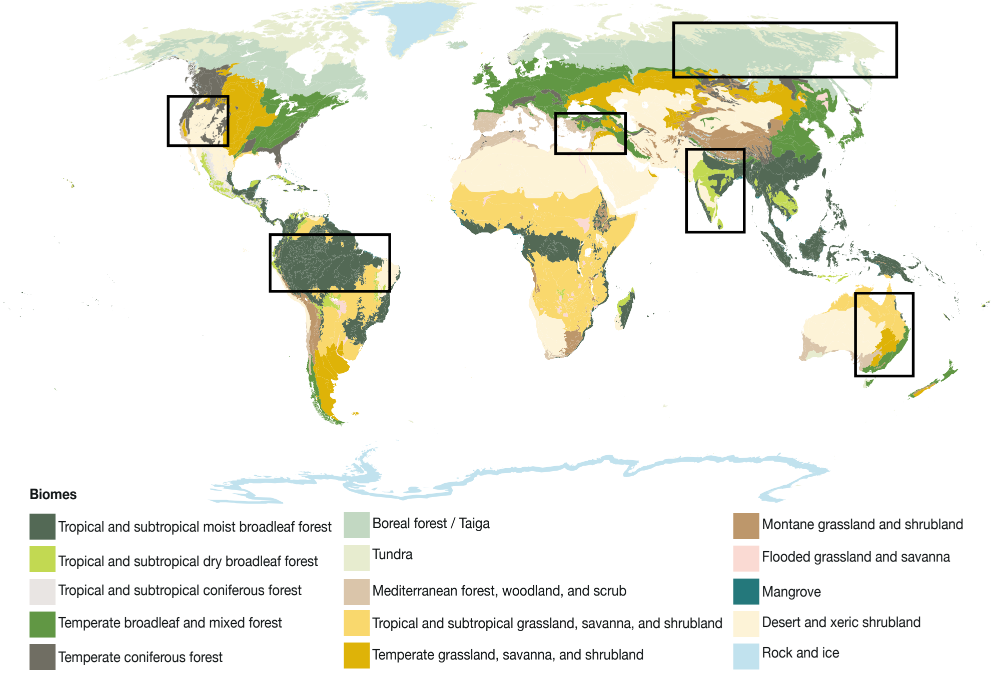
The main biomes of the World



Source: Lambrechts et al. 2020.

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Figure 4.1. The top map shows the year in which the highest moving average three-month period of fire occurred (2002–2019) based on MODIS Fire_cci Burnt Area product (Chuvieco et al. 2018). Cooler colours (blues and greens) indicate that the highest three-month period of fire occurred further in the past. Hotter colours (oranges and reds) indicate that the highest three-month period of fire occurred more recently. Regions of interest with recent fire activity are highlighted (clockwise from top left): western USA/ Canada, the Middle East, India, Siberia, eastern Australia, and Amazonia. The bottom map shows the major biomes of the world with the same regions of interest highlighted. The regions where the highest three-month period of fire have occurred most recently span a wide range of biomes, locations, and latitudes.

4.1 Response of ecosystems to fire

Ecosystems and species are not adapted to fire per se but rather to certain fire regimes. Those with little or no evolutionary exposure to fire will often suffer harm of some sort when burnt. Those with a history of fire experience selective pressures that cull and shape the species pool and the environment in concert with the peculiarities of the reigning fire regime (Figure 4.2). Major and persistent changes to fire regimes can have significant impacts on ecosystems and species (Bowman et al. 2014: Pausas and Keeley 2014; Keeley and Safford 2016).

Fire frequency and fire severity are the two fire regime characteristics whose impacts on biota are best understood. Because fire depends on fuel (i.e., vegetation) to burn, there is an important interaction between fire severity and frequency (Pickett and White eds. 1985; Huston 2003). Frequent fires reduce fuel loads, which results in lower heat release. In some systems, the long-term absence of fire permits the accumulation of heavy fuel loads and thus high fire intensity when burning conditions are right. The effects of both frequency and severity (and other fire regime characteristics) must be considered as a function of the biology of the affected biota. In other words, "high frequency" and "high severity" mean different things to different taxa. In order for a species population to be resilient to fire, the characteristics of fire must fall within some "natural range of variation" that encompasses the conditions that permit population persistence (Landres et al. 1999).

Disturbance frequency is a major driver of species response and ecosystem impacts and there has been much consideration of its ecological and evolutionary role (Connell 1978; Pickett and White eds. 1985; Huston 1994). When fire disturbance is extremely frequent – for example an annual occurrence – almost no woody plant species can survive, and the vegetation will be dominated by herbaceous and often annual plants. When fire is extremely infrequent – for example, once every few centuries or millenninia – the community will support longer-lived species and adaptations to fire will be rare. Between these generalized extremes, fire frequency and changes in frequency can have major ecological and evolutionary impacts on plants. For example, many moist tropical forests support almost no lightning-ignited fire, even where there is a defined dry season, because lightning is most common in the wet season and is generally accompanied by rain. However, anthropogenic fires are set during the dry season and when they enter forests they kill some trees and open the canopy, permitting the invasion of tall (mostly exotic) grasses that are absent in the dense shade of the unburnt forest. Logging before or after fire can accelerate and intensify grass invasion. Grasses are extremely flammable in the dry season, and their presence in the forest understory leads to subsequent dry season burning. This progressively dries out and opens up the forest canopy, leading to a positive feedback loop that is known as the "grass-fire cycle" (D'Antonio and Vitousek 1992). The grass-fire cycle is a major source of tropical forest degradation worldwide (Hughes et al.1991; Silvério et al. 2013).

Changing fire frequencies can also have deleterious impacts on fire-adapted ecosystems. For example, notably decreasing the time between fires in Mediterranean-climate shrublands can decrease abundance and increase extinction risk of obligate-seeding shrubs and trees (taxa that are killed by fire and regenerate via germination of soil- or canopy-stored seed banks) as it reduces the viable seed pool due to plant immaturity (Zedler et al. 1983; Bradstock et al. 1998, Tulloch et al. 2016).

Fire intensity is also a major driver of species response and ecosystem impacts (DeBano et al. 1998). In ecosystems where fires are typically hot but infrequent enough to permit the domination of woody vegetation – such as in many Mediterranean-climate shrublands like chaparral (North America), fynbos (South Africa), or kwongan (Australia), as well as in ecosystems supporting serotiny – many species have evolved reproductive processes that require the heat of fire to cue germination or seed release (Keeley 1987; Bell et al. 1993). Species vary in their response to heating with some having a greater requirement for heat to stimulate germination or trigger release of seed from cones and fruit. Fires outside of optimal heat exposure (duration

Adaptive strategies of biota to fire

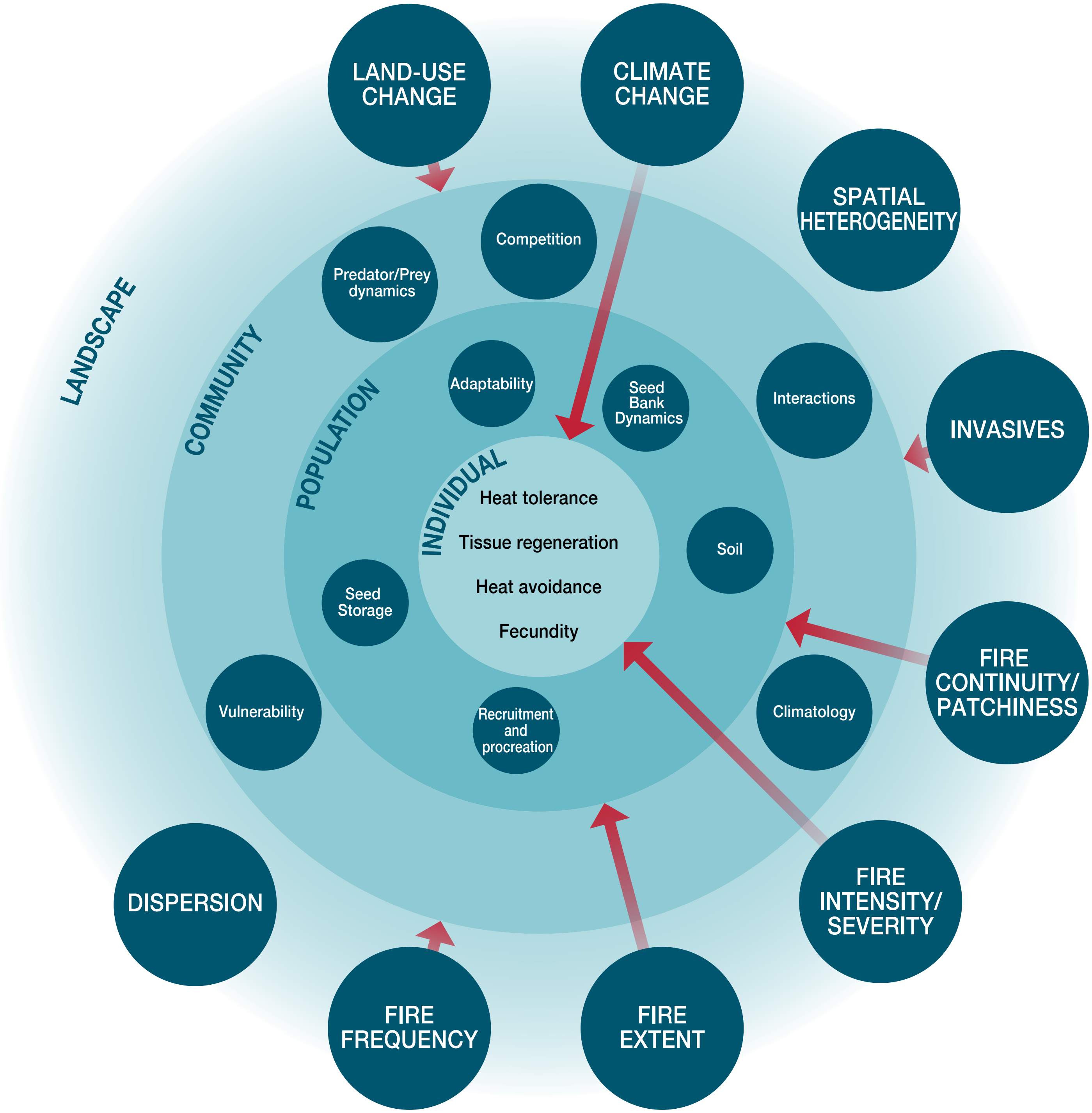


Illustration by Andrew Sullivan/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

Figure 4.2. Adaptive strategies of biota to fire at the landscape, community, population, and individual scale. Depending on the conditions driving the fire, its intensity and extent, and the frequency with which such events occur, some strategies may work better than others at different scales. Fauna strategies are largely based on mobility and refugia. Flora strategies are largely based on either increased heat tolerance and tissue regeneration or increased seed production.

or intensity) for a species may either kill seeds or fail to germinate sufficient seeds to guarantee population recovery (Keeley 1987; Bond and van Wilgen 1996; Penman and Towerton 2008). Other fire-related mechanisms that can cue germination include chemicals found in smoke or charred wood (Keeley and Bond 1997).

In western North America, seasonally dry conifer-dominated forests experienced frequent low-intensity fires before Euro-American arrival and the dominant species were characterised by adaptations to this fire regime. These included thick bark, self-pruning of lower branches, and high litter flammability (Safford and Stevens 2017). Exclusion of fire from these forest types and selective logging since the early twentieth century have resulted in forest densification, successional trends favouring fire-intolerant species, and accumulation of live and dead fuels. As a result the fire regime has changed to one characterized by infrequent but highly intense fires that kill most of the forest canopy over large areas. This results in regeneration failures and potentially permanent conversions to shrub- and grass-dominated vegetation (Welch et al. 2016; Coop et al. 2020).

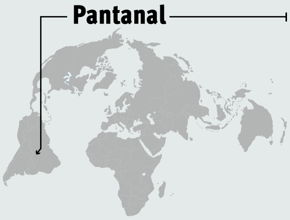
The effects of fire on animal communities at the global scale are poorly understood compared with plants (Pausas 2019; Foster et al. 2016). There are however, some advantages for fire-dependent species that are either affected by fire-induced direct mortality during an event (or shortly after) or by changes that occur in the longer-term and affect their persistence in those landscapes (Nimmo et al. 2019). The limited information is partly due to the movement capacity of animals, making their study and tracking difficult, costly, and time-consuming, particularly in remote ecosystems with dense vegetation such as tropical forests (González et al. 2017). This is compounded by limited access to high-resolution spatial fire satellite data, lack of long-term occurrence monitoring data or genetic data (Nimmo et al. 2019). The other key challenge has been to identify consistent traits that could be useful for developing a predictive framework for animals (Langlands et al. 2011; Driscoll et al. 2020).

We know that fire can indirectly affect fauna through changes in the structure and composition of the vegetation in their habitats (Litt and Steidl 2011; Mowat et al. 2015). For example, recent research in North America has documented important impacts of spatial heterogeneity in fire severity (which refers in this case to the impacts of intensity on vegetation) on population sizes and trends for a variety of bird and mammal species (Jones et al. 2020; Stillman et al. 2021; Steel et al. in press). Smoke and flames also directly affect animals (Peres 1999). The capacity to escape fire can have a strong influence on the likelihood of survival (Pausas 2019) – animals with good dispersal ability or local refuges such as burrows or rock crevices (Roznik and Reichling 2021; Selwood and Zimmer 2020), are able to escape or avoid the fire (Pausas 2019) and take refuge until it is over (Brotons et al. 2008; Kelly et al. 2012; Robinson et al. 2013). These movement-related behavioural responses not only influence the survival of animals, they also allow species to recolonise burnt areas (Chia et al. 2016), either from areas outside of the fire area or from refuges within the burn boundary. Movement capacity, and presence of refuges, therefore, have a strong influence on the recovery of animals in fire-affected ecosystems (Chia et al. 2015).

Case study: Burning tropical forests of Latin America – Amazon region

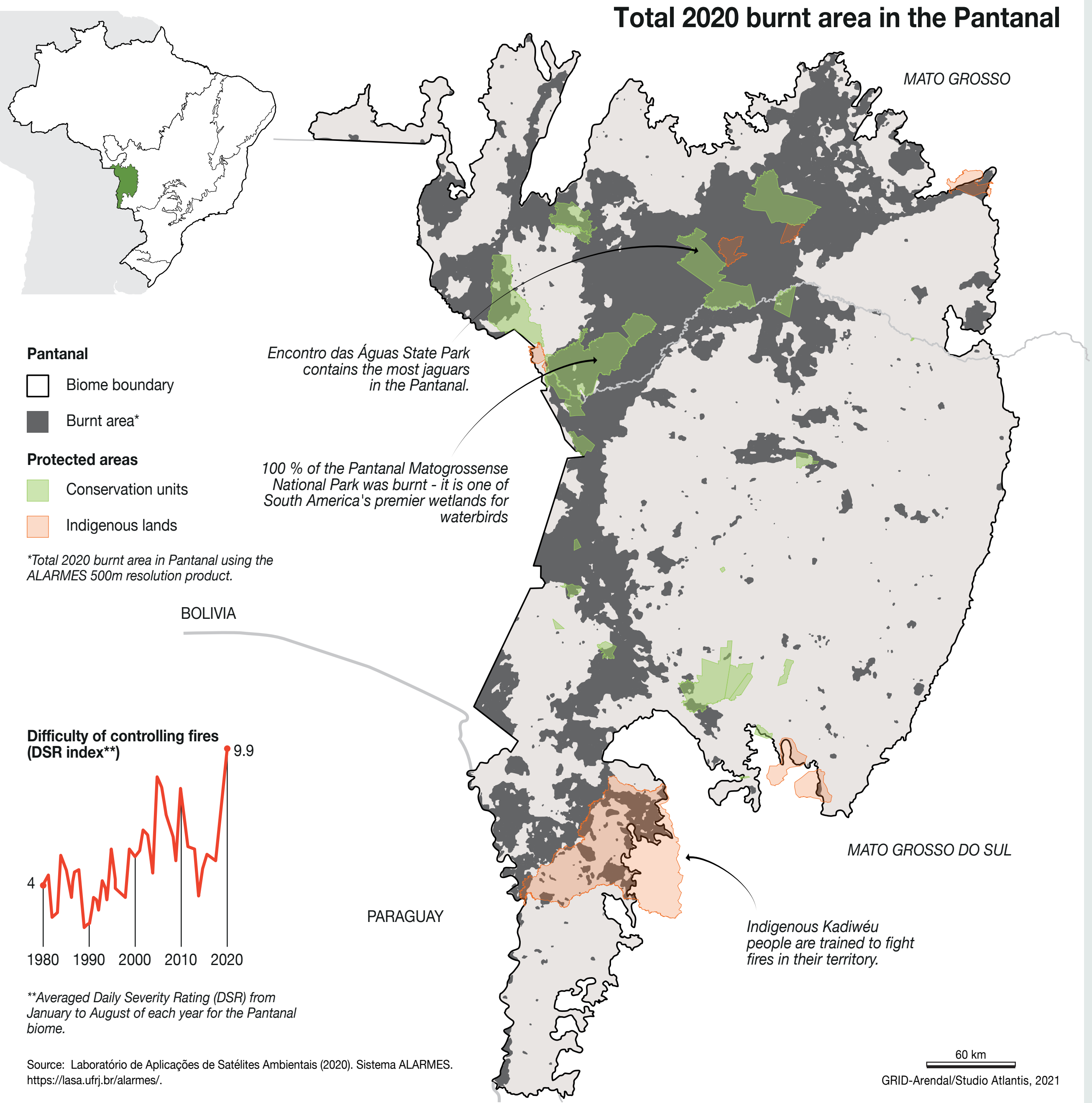
The forests of the Amazon region occur between 8.5°N and 19°S and make up approximately 50 per cent of the Earth's tropical forest and 23 per cent of all forests. This biome has been subject to large-scale forest conversion driven by deforestation for pasture and cropland, road construction, and mining. These activities are responsible for introducing fire into an area that contains a myriad of fire-sensitive wet forest ecosystems, (for example, tropical evergreen forests, gallery forests and rainforests with naturally high moisture and low fuel loads) (Cochrane 2003). These fires are also modulated by climate factors such as droughts, although 2019 and 2020 were examples of how nowadays fire can be widespread in the Amazon even if there is not an extreme drought (Silveira et al. 2020; Libonati et al. 2021). This is partly because many of these ecosystems have been profoundly fragmented, resulting in a mosaic of forest edges, human-disturbed forest (often by selective logging) and adjacent pastures (Armenteras et al. 2013; Silva Junior et al. 2018). The disturbed habitats have high fine fuel loads that can burn after only a few days without rain. Logging opens canopy patches that enhance vegetation and soil drying and allow greater penetration of wind, which facilitates fire spread (Gerwing and Uhl 2002; Berenguer et al. 2014; 2018). Fires that enter disturbed forest kill trees and further reduce the canopy cover. The tree mortality causes increased accumulation of surface litter fuels, which can make subsequent fires more intense (Cochrane 2003). Recent estimates have supported a fourfold increase of dead trees (Silva et al. 2020a) and daunting estimates of biodiversity loss after fire (Barlow et al. 2016).

Case study: Burning a wetland – the Pantanal



Stretching across Brazil and parts of Bolivia and Paraguay, the Pantanal is the world's largest tropical wetland, covering around 15 million hectares. Parts of the Pantanal have been designated a biosphere conservation area and recognised as a United Nations Educational, Scientific and Cultural Organization (UNESCO) World Heritage site. The area is home to thousands of endangered species such as the jaguar (*Panthera onca*), the giant otter (*Pteronura brasiliensis*), the marsh deer (*Blastocerus dichotomus*), and the hyacinth macaws (*Anodorhynchus hyacinthinus*), and has the greatest concentration of wildlife in South America. The Pantanal is also a key migratory route of terrestrial and aquatic bird species.

Since 2019, the Pantanal has experienced a severe drought (Marengo et al. 2021). In 2020, the coincidence of hot and dry conditions pushed vegetation combustibility thresholds to their highest since 1980 (Libonati et al. 2020a). These conditions, combined with a lack of appropriate management, resulted in the intense and widespread fires of 2020 – the highest fire year recorded between 2001 and 2020 (Garcia et al. 2021). The fires, which in most cases were deliberately lit, consumed almost one-third of the biome – approximately 4 million hectares (Figure 4.3; Libonati et al. 2020a). Large areas of Indigenous lands and converted areas were extensively burnt, devastating the habitat of many endangered species. Protected areas such as the "Encontro das Águas" (the Meeting of Waters) State Park, an area with the highest feline density in the world, burnt entirely (Libonati et al. 2020b). It will take several months to assess the total extent of plant and animal loss across the area, but already there are indications that the impact will be extensive and long-lasting, giving rise to concerns that this biodiversity hotspot may not be able to fully recover from these extreme fires (Mega 2020).



Total 2020 burnt area in the Pantanal

Encontro das Águas State Park contains the most jaguars in the Pantanal.

100 % of the Pantanal Matogrossense National Park was burnt - it is one of South America's premier wetlands for waterbirds

Indigenous Kadiwéu people are trained to fight fires in their territory.

**Total 2020 burnt area in Pantanal using the ALARMES 500m resolution product.*

***Averaged Daily Severity Rating (DSR) from January to August of each year for the Pantanal biome.*

Source: Laboratório de Aplicações de Satélites Ambientais (2020). Sistema ALARMES. https://lasa.ufrj.br/alarmes/.

GRID-Arendal/Studio Atlantis, 2021

Figure 4.3. Total 2020 burnt area in the Pantanal using the ALARMES 500m resolution product. Conservation units and Indigenous lands are shown in green and orange, respectively. The bottom left graph shows Pantanal's average Daily Severity Rating (DSR) from January to August each year, estimated using the ERA5 reanalysis product (Libonati et al. 2020a). DSR is a numeric rating of the difficulty of controlling fires.

Case study: Changing fire regimes in boreal forests

The boreal forest, or taiga, is a region dominated by coniferous trees that occurs between 50° and 70°N and makes up approximately 29 per cent of the Earth's forested regions (Kuusela 1992; Brandt et al. 2013). These forests are characterised by a continental climate with extremely cold winters that last for six to eight months each year (mean temperatures from -10 to +2°C, with temperatures as low as -60°C). The growing season is short (100–150 days) with warmer temperatures (mean temperatures from 10 to 20°C) (Kuusela 1992). These harsh conditions have led to the development of relatively species-poor forests when compared with other biomes, with as few as one to six tree species typically present (La Roi 1967). The area is experiencing some of the fastest rates of climate change globally (an estimated 1–2.5°C warming from 1901–2012 (IPCC 2013); Chapter 2), with cascading impacts on fire regimes, permafrost, and biodiversity. In the coming century, this region is expected to continue to warm at up to twice the rate of the rest of the Earth.

Fire is the chief natural disturbance in the boreal forest (Safford and Vallejo 2019) but recent fire regimes in parts of the boreal forest are more extreme than fire regimes of the past. Many boreal species are well-adapted to fire. For instance, black spruce (*Picea mariana*) has serotinous cones that only open with exposure to high temperatures during fire. Other species like larch (Larix spp.) have much higher rates of regeneration following fires that reduce soil organic layer depths (Sofronov and Volokitina 2010; Alexander et al. 2018) or decrease competition from other plants (Mateeva and Mateev 2008). But as the climate warms and permafrost thaws, the frequency, severity and extent of forest fires is increasing across the northern-hemisphere boreal forests (Kasischke and Turetsky 2006; Liu et al. 2012; Shvidenko and Schepaschenko 2013; Ponomarev, Kharuk and Ranson 2016). The major changes in fire regimes disrupt the adaptive strategies of plants (see Figure 4.2) and can reduce ecosystem resilience and negatively affect biodiversity (Johnstone et al. 2016; Safford and Vallejo 2019; Whitman et al. 2019).

Siberian taiga

The boreal forests of Siberia are classified as either "light" or "dark" coniferous taiga. They have different species composition and are characterised by different fire regimes. Light coniferous taiga is dominated by Scots pine (*Pinus sylvestris*) and larch (primarily *Larix cajanderi* and *Larix gmelinii*), while the dark coniferous taiga is dominated by spruce (*Picea* spp.) and fir (*Abies* spp.) (Shorohova et al. 2009). Across the region, fires are caused by both lightning and anthropogenic intervention (Figure 2.1, chapter 2). However, there is a greater prevalence of lightning-caused fires in the most remote, northern latitudes, and a greater prevalence of human-ignited fires in the more populated, southern latitudes (Kharuk et al. 2011; Liu et al. 2012).

The typical fire return interval (FRI) in the light coniferous taiga varies with latitude. The historical FRI ranges from 350 to 80 years with longer FRIs in the larch-dominated forests of the north and shorter FRIs in the pine-dominated forests of the south (Kharuk et al. 2011; Ponomarev et al. 2016). Most fires in the north are stand-replacing surface fires (Krylov et al. 2014) that cause larch mortality through damage to the roots (Sofronov and Volokitina 2010; Volokitina 2015). Even-aged stands develop following these stand-replacing fires and remain even-aged up until approximately 180 years post-fire when eventual windthrow or gap dynamics influence the development of uneven-aged stands (Shorohova et al. 2009). In the southern portion of the light coniferous taiga, the larger-diameter larch trees and pine trees can withstand surface fires (Krylov et al. 2014), and fires provide an environment suitable for larch regeneration, leading to the development of multimodal stand structures (Shorohova et al. 2009). In portions of central and southern Siberia, successional dynamics also occur, with birch (Betula spp.) and aspen (Populus tremula) dominating post-fire stands and eventually being replaced by larch (Shorohova et al. 2009).

Forest regeneration in the light coniferous taiga is strongly limited by both deep soil organic layers (Sofronov and Volokitina 2010; Alexander et al. 2018) and seed source availability (Cai et al. 2013), both of which are affected by changing fire regimes. Surface fires consume a portion of the soil organic layer, with the most severe fires leading to the greatest reduction in soil organic layer depths, generally improving seedbed conditions. When seeds are not limiting, the highest levels of larch regeneration tend to occur in these areas with the highest soil burn severities (Alexander et al. 2018). Thus, changing fire regimes that lead to increase soil burn severity could promote increased larch regeneration (Alexander et al. 2018). In contrast, seed source availability becomes increasingly limiting with increases in the frequency and extent of fires. Cajander larch (Larix cajanderi) produce wind-dispersed seeds, have masting every two to three years, and do not produce a seed bank (Abaimov 2010). Therefore, successful larch regeneration following a stand-replacing fire requires wind dispersal of seeds from nearby unburnt areas, or from surviving trees within the burnt area (Greene and Johnson 1995; Figure 4.4). If fire extent is greater than the distance that larch are able to disperse, or if fire frequency becomes so short that mature larch trees are not able to grow between fires, then forest loss or declines in forest density may occur. In these cases, forests can convert to low-density forests or to grassland or shrub-dominated communities (Sofronov and Volokitina 2010; Scheffer et al. 2012; Cai et al. 2013; Alexander et al. 2018). In the southern portion of the light coniferous taiga, there is also evidence of conversion of formerly Gmelin's larch-dominated (*Larix gmelinii*) forests to increased dominance by birch (*Betula* spp.) or aspen (*Populus tremula*) due to rapid FRIs (Zyryanova et al. 2007; Cai et al. 2013). These changes in the overstory structure and composition of larch forests can have consequences for the understory plant community, as different plant communities are associated with variation in tree density, light availability, and overstory tree composition (Ma et al. 2016; Zhang et al. 2017; Kumar et al. 2018). These shifts in forest composition also have consequences for albedo (Loranty et al. 2014), above-ground carbon storage (Alexander et al. 2012), nutrient cycling (Nilsson and Wardle 2005; Campioli et al. 2009), and permafrost stability (Abaimov et al. 2002).

In the past, fires rarely occurred in central Siberia's dark coniferous taiga, with FRIs ranging from 300–900 years (Mollicone et al. 2002; Feurdean et al. 2020). When a fire does occur, it often results in high levels of tree mortality (Tautenhahn et al. 2016). The current fire regime in the dark coniferous taiga is now outside the historical range of variability of the past 5,000 years, with increases in both fire frequency and severity (Feurdean et al. 2020).

In the dark coniferous taiga, deciduous hardwoods like aspen or birch are the first species to gain dominance following fire, due to their ability to resprout from underground reserves, long-distance seed dispersal, and fast growth rates (Furyaev et al. 2001; Schulze et al. 2005; Tautenhahn et al. 2016). Evergreen conifers like *Abies sibirica*, *Abies nephrolepis*, *Picea abies*, and *Picea obovata* establish around the same time as deciduous hardwoods but have much slower growth rates and do not gain dominance until more than 70 years post-fire (Schulze et al. 2005; Shorohova et al. 2009). Seed availability is strongly limited by dispersal for *Abies* spp. and *Picea* spp., and therefore, increases in fire activity that limit the availability of nearby seed sources can spark a transition in forest successional trajectories to continued dominance and self-replacement by deciduous hardwoods following fire (Tautenhahn et al. 2016). At the landscape scale, this could lead to a decline in the evergreen conifer fire "avoider" species like *Abies* spp. and *Picea* spp., that promote cool, moist conditions in the understory (Tautenhahn et al. 2016; Feurdean et al. 2020). As with compositional and structural changes in the Siberian light taiga, the shifts in species composition that result from intensifying fire regimes could lead to changes in albedo (Loranty et al. 2014) and above-ground carbon storage (Alexander et al. 2012; Alexander and Mack 2016).

Figure 4.4. Light coniferous taiga near Chersky in the Sakha Republic, Russia, burnt in 2001, with complete tree mortality. This forest has only a single tree species – Cajander larch (*Larix cajanderi*). There is abundant regeneration of larch seedlings close to the edge of the burn, but very little regeneration further into the burnt area where seed sources are more limiting.

Case study: Mediterranean climate regions, with focus on the North American Mediterranean Climate Zone

The Mediterranean climate regions (MCRs) occur between 30° and 45° latitude on the west coasts of Africa, the Americas, Australia, and Europe. Forests cover only a fraction of the MCR land base and collectively make up less than 1 per cent of the Earth's forests (Safford and Vallejo 2019). The MCRs are characterised by cool, wet winters and warm or hot, dry summers. Generally, there is sufficient precipitation in the winter and early spring to sustain annual vegetation growth that contributes to the accumulation of fine fuels which dry out and become combustible during the dry late spring/summer/early-autumn fire season. As a result, Mediterranean vegetation is among the most fire-prone and fire-shaped on the planet, with fire acting as a major control on speciation in four of the five MCRs (western North America, south-western Australia, Mediterranean Basin and South Africa). MCR flora exhibit a wide range of adaptations to fire. In high severity fire regimes, these include seed banking, serotiny (seeds released by fire), and fire-cued germination. In low severity fire regimes, fire-resistant characteristics such as thick bark and self-pruning of lower branches occur (Keeley et al. 2011).

Fire-initiated reproduction in high-severity fire regimes – which are typified by hard-leaved shrublands such as chaparral (North America), kwongan (south-western Australia), matorral (Chile), garrigue and maquis (Europe), and fynbos (South Africa) – is vulnerable to major and persistent changes in fire frequency in all five MCRs (Keeley et al. 2012). In the North American MCR (NAMCR) before Euro-American settlement, chaparral ecosystems were characterised FRIs of between 30 and 100 years (Van de Water and Safford 2011). As a result of climate and land-use change, many areas dominated by chaparral

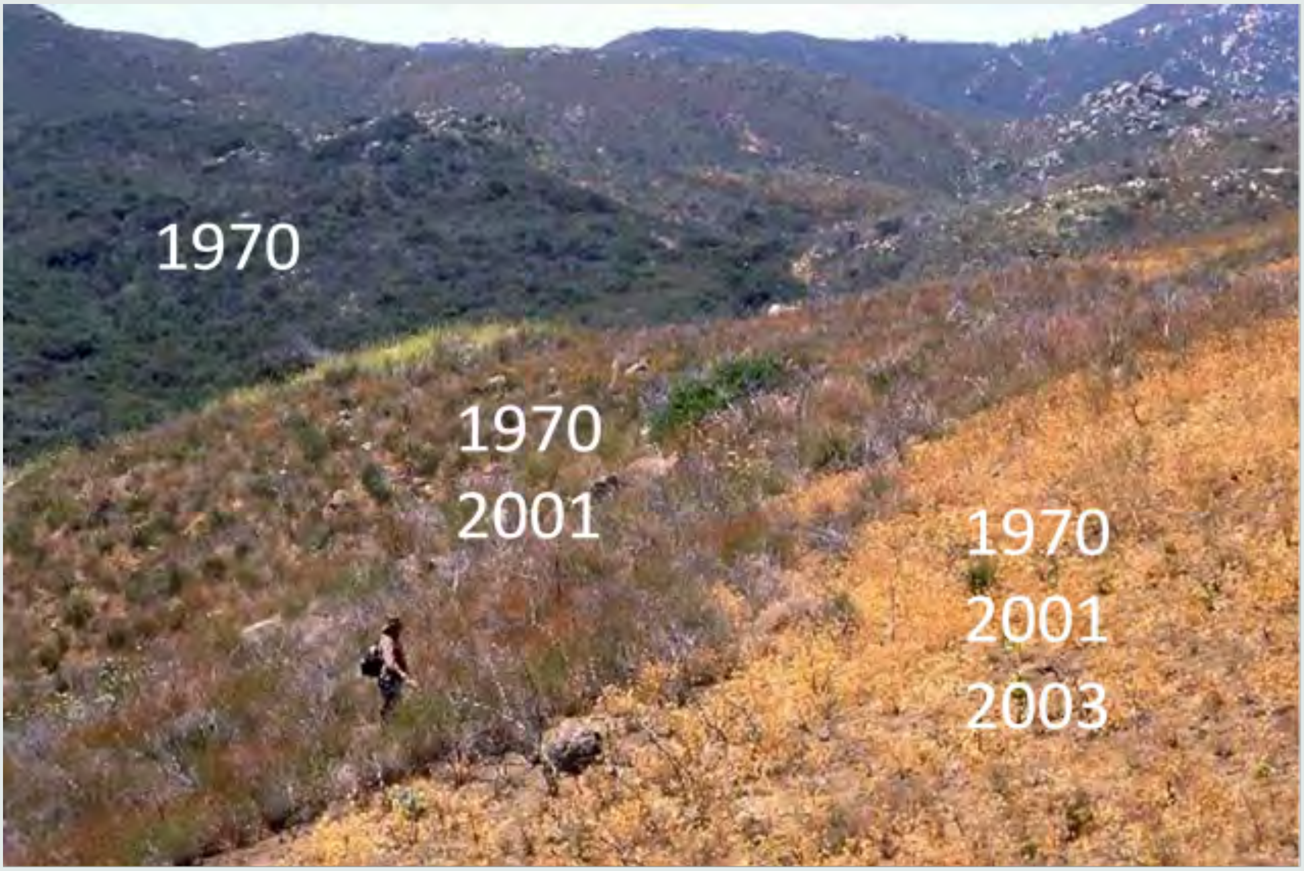


Figure 4.5. Effects of high fire frequency on chaparral in the North American Mediterranean climate region. The entire photographed area near San Diego was burnt in 1970, then two fires occurred in 2001 and 2003. The area in the foreground, dominated by exotic annual grasses and limited native shrub regeneration, has an FRI of 16.7 years over the last 50 years. Ecosystem resilience has suffered tremendously.

are now experiencing FRIs of <15 years, which is a rule of thumb threshold for the loss of obligate-seeding species – both shrubs and serotinous conifers – which require time to reach sexual maturity and produce and store seeds (Keeley and Safford 2016). Fire frequencies are so high in some locations that even resprouting species are failing to regenerate. This has led to severe environmental degradation of chaparral ecosystems in areas subject to high ignition densities, with dense woody native cover replaced by grassy scrublands dominated by invasive exotic weeds, disruption of hydrological cycles and increased overland flow and flooding, and major erosion events (Underwood et al. eds. 2018; Figure 4.5).

High-frequency/low-severity (HFLS) fire regimes in the MCRs are concentrated in woodlands and open forest systems (Figure 4.6). In the NAMCR and the Mediterranean Basin, these are dominated by oaks and fire-resistant pine species and in Australia by certain *Eucalyptus* spp. In these ecosystems, there is a general lack of species with reproductive strategies associated with high-severity fire, such as serotiny or fire-initiated germination (Keeley and Safford 2016). In these HFLS ecosystems, the fire regime before Euro-American settlement was fuel-limited (as opposed to climate-limited). The high frequency of low-intensity fire limited fuel accumulation and maintained conditions that supported low- to moderate-severity fire (Steel et al. 2015; Safford and Stevens 2017).

Changes in the fire regime of forests and woodlands once characterised by HFLS fires are impacting biota in the NAMCR and Australia. This is true for areas that have been long-unburnt as well as areas experiencing uncharacteristically high-severity burning. Bird communities are heavily influenced by fire, and many species are scarce in long-unburnt areas (White et al. 2016). Trends of increasing fire severity benefit some animals (for example, certain woodpecker and bat species), but imperil those that depend on older forest environments, such as – in the NAMCR – lichens, the spotted owl (*Strix occidentalis*), the northern goshawk (*Accipiter gentilis*), and the Pacific fisher (*Pekania pennanti*) (Jones et al. 2016; Miller et al. 2018; Steel et al. 2019). In drier pine-dominated forests, the lack of fire has negatively influenced local plant diversity (Richter et al. 2019; Miller and Safford 2020). Aquatic organisms can be strongly affected by physical changes to watercourses and decreases in water quality triggered by high-severity burning (Oliver et al. 2012). Terrestrial invertebrates and small mammals do not appear to be greatly impacted by variation in fire severity (Fontaine and Kennedy 2012), but some species are strongly tied to open canopies and low-surface fuels (e.g., Dalrymple and Safford 2019). Overall, studies in HFLS ecosystems show that a diversity of burning conditions – but strongly skewed towards low- and moderate-severity burning – is important to community diversity in nearly every taxonomic group studied.

Figure 4.6. Comparison of forest conditions in the North American Mediterranean climate zone's yellow pine mixed conifer forest under continued frequent fire and absence of logging (left) versus >100 years of fire suppression and logging of fire-tolerant species (right). Left: Mixed conifer forest, Sierra de San Pedro Mártir, Baja California, Mexico. The area was not logged and fire exclusion has been practised only since the 1980s. Right: Typical stand of mixed conifers following nineteenth- and twentieth-century logging and a century of fire exclusion, Sierra Nevada, California.

Case study: The 2019–2020 Black Summer in Australia

The 2019–2020 wildfires in Australia were exceptionally severe, burning ecosystems that typically do not burn, including the World Heritage-listed Gondwana rainforest (Nolan et al. 2020). The most extensive of the wildfires occurred in the south-eastern corner of the continent, although south-western ecosystems and northern rainforests were also affected. It is estimated that 3 billion animals were killed or displaced, including an estimated 143 million mammals, 180 million birds, 51 million frogs and a staggering 2.5 billion reptiles (van Eeden et al. 2020).

The fires burnt the habitat of 832 of Australia's native animal species (Ward et al. 2020) including 21 "threatened with extinction" under Australia's Environment Protection and Biodiversity Conservation (EPBC) Act. These include the Kangaroo Island dunnart (*Sminthopsis griseoventer aitkeni*) and the long-footed potoroo (*Potorous longipes*), which had more than 80 per cent of its habitat impacted by the fires. Furthermore, almost a quarter (272) of the 1,180 listed threatened plant species had >10 per cent of their known distribution within the fire footprint (Wintle et al. 2020).

Many native Australian species have been historically maintained within specific fire regimes. Altered frequency, severity, or timing of extreme events such as wildfire can increase their extinction risk and encourage threats such as invasive species. Many species and ecosystems are known to be at risk because of multiple, interacting disturbances (Didham et al. 2007; Foster et al. 2016).

Fire effects on soil and erosion

Fire is often considered an important soil-forming factor, influencing soil development and sediment production (Certini 2005). The heat transfer from the combustion of biomass directly impacts soil properties, and indirectly affects erosion rates and sediment production (Robichaud et al. 2016; Pingree and Kobziar 2019). At lower temperatures (below 200°C), essential biological properties are affected, in particular there is a significant reduction in the microbial community, biomass, and seed bank. At higher temperatures (above 200°C), physico-chemical properties of soil are modified through the combustion of soil organic matter and the production of pyrogenic compounds. Physical transformations include the breakdown in soil structure and aggregate stability, reduced moisture retention capacity, and development of soil hydrophobicity (soil repellency to water that hampers soil wetting). Chemically, fire-affected soils undergo changes in nutrient cycling rates and pH. These changes typically lead to more brittle and erodible soil (Shakesby 2011; Wittenberg 2012). This may cause the accelerated loss of topsoil after the fire, with published rates of 0.1–41 Mg ha^{-1} per year after moderate to severe fires compared with 0.003–0.1 Mg ha^{-1} in unburnt landscapes (Shakesby 2011; Santín and Doerr 2016). It is estimated that more than 70 per cent of the total annual erosion in the MCRs is caused by wildfires (Swanson 1981). Using a combination of climate, fire, and erosion models, one study in the western USA estimated that by 2050 post-fire sedimentation rates would increase by more than 100 per cent in over 30 per cent of the watersheds due to increased fire activity (Sankey et al. 2017).

Soil erosion is a problem worldwide. The loss of topsoil – where organic substances and vital nutrients are stored – results in decreased soil fertility. However, post-fire increases in erosion rates are often limited to a short period following the fire. They usually decrease at larger spatial scales (basins) due to local redeposition of sediment and rapid regrowth of vegetation (Zituni et al. 2019).

Chapter 5 – Risk mitigation and wildfire management

5.1 Introduction

There are five integrated phases of emergency management of wildfires: review and analysis, risk reduction, readiness, response, and recovery, also known as the 5Rs.[3]

- *Review and analysis* – the collection of data and information on past events. Understanding critical factors (e.g., fuels, weather, fire behaviour, ecological response, fire management response, general-public response, post-fire recovery processes, etc.) and causal relationships (e.g., fuel management and fire mitigation effects) helps improve wildfire mitigation and management.
- *Risk reduction* – includes many possible actions aimed at reducing the likelihood and consequences of wildfire. For example, fuel management (at spatial scales from householder to wilderness), resilient building design, land use planning, and reducing the incidence of arson and accidental ignitions.
- *Readiness* – even with effective risk reduction measures in place, fires will still occur. Communities and fire services need to be prepared. For residents this may be having an evacuation plan or a well-conceived plan for remaining in place to protect assets. No matter what preparations are made, they need to be based on an awareness of the nature and risk of wildfires. Fire services and other relevant organisations also prepare for wildfires by having trained personnel and appropriate technology, systems, and process in place.
- *Response* – relates to the actions taken to manage a wildfire when it does occur, including resource allocation and management (personnel and equipment) for safe suppression efforts, wildfire alerts and fire status updates, incident management, and evacuations.
- *Recovery* – includes all remediation efforts during and after a wildfire disaster.

Because it is not possible to prevent all fires, nor desirable from an ecological standpoint, this chapter focuses on mitigation, actions that can help to reduce the potential adverse impacts a wildfire might have.

Wildfire management is challenging and will become even more so with the increasing risk and threat of wildfires. Worsening risk profiles are increasing the demand for detailed and timely intelligence on fire likelihood and threat. In many regions of the world the majority of fire management expenditure is on fire response – that is, direct suppression of wildfires as they occur. For this to be effective, understanding of the ecosystem in question, its vulnerability or adaptation to fire, amount of available fuel, the assets, infrastructure and lives at risk, and the likelihood of a fire outbreak developing into a wildfire and its likely behaviour is essential.

Even if it was desirable to remove the risk of wildfire entirely, there will always remain a residual risk of fire that cannot be avoided. We therefore must learn to live with fire. In some areas under some conditions, this may involve developing flexible fire response strategies and planning tools that facilitate fire management decisions to not immediately suppress a wildfire because it is likely to have desirable ecological benefits without risking key human values (North et al. 2012; Boisramé et al. 2017; Rakhmatulina et al. 2021). In other areas, this may involve focusing on reducing wildfire outbreaks through concentrated actions to reduce potential accidental ignitions (Abt et al. 2015; Collins et al. 2015). Response to outbreaks that do occur is often required. However, the emphasis in many regions where risk remains high should be on rapid, appropriate, and effective initial attack on unwanted fires before they become established or escalate into an emergency (Thompson et al. 2016). This requires improved ability to quickly assess which fires may not need immediate suppression, as discussed above, and less emphasis on sustained action on large fires that are predominantly beyond control. Without a complete shift in our standard operating practice in many parts of the world we will continue with politically motivated decisions (public opinion in many places favours putting out fires at all costs) that currently dominate wildfire management. Moving wildfire management from an automatic fire exclusion response to allow more adaptive management of fire in the landscape where appropriate could, in some regions, have multiple benefits across the breadth of social, economic, and ecological concerns (Otero and Nielsen 2017; Thompson et al. 2018; Moreira et al. 2020). Despite the tendency to focus on fighting fires, in many situations, prevention (in the form of reducing the probability of wildfires breaking out), and mitigation activities (reducing the potential impact of wildfires when they do occur) are more likely to offer opportunities for long-term, cost-effective reduction of wildfire disasters (Multihazard Mitigation Council 2018).

Integrated fire risk reduction is central to adapting to current and future changes in global fire risk. The quantification of negative impacts is crucial in illustrating the importance of investing in effective prevention and mitigation actions

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[3] The stages are also sometimes referred to as planning, prevention, preparedness, response, and recovery.

as well as necessary and appropriate response and recovery (Figure 5.1). Managing wildfire risk requires integrated region-specific approaches that factor in population awareness and preparedness, ignition likelihood, fire spread forecasting, surveillance and early-warning systems, adaptive suppression strategies, fire-regime restoration and management, landscape-scale fuel management, changes in land-use practices, and active restoration of landscapes. In this, we have much to learn from many Indigenous peoples who have historically effectively coexisted with fire-prone ecosystems and have often used fire as tool for sustainable land-management (Huffman 2013).

5.2 Risk mitigation

When socio-environmental hazards begin to have significant impacts, the initial societal response tends to focus on mitigating the risk with structural interventions or technology to eliminate or modify the hazard itself (Burton et al. 1993). In many regions, for wildfires this has involved regulating activities that can lead to wildfire ignitions (e.g., total fire bans). Where ignition prevention has had limited success, the response has been to build better fire detection or fire suppression capability, often reflecting a belief that fire risks can be eliminated, given enough investment.

In Canada and the USA, for example, well-targeted and appropriate suppression resources are successful in containing most fires, restricting the number of fires that escape initial containment to less than 10 per cent of ignitions. However, it has been found that those few fires that escape initial attack and become large generally contribute the most to annual areas burnt (e.g., Stocks et al. 2002). Many of these wildfires cannot be directly suppressed while they burn under extreme weather conditions as the fire behaviour exceeds the limits of suppression (Tedim et al. 2018). For these fires, no amount of additional equipment, resources, data, information, or technology will be sufficient to extinguish or control the fire until the weather changes. However, other firefighting actions may be undertaken to facilitate containment when the weather does change.

In developing countries, many fire response organisations are under-staffed and under-equipped and have a serious lack of fire detection technologies. Moreover, the identification of priority areas to which the resources to mitigate or suppress fires should be allocated requires technical information, such as the potential fire behaviour and impact, the fire risk uncertainty, distances, and accessibility to the outbreaks, among others, that in most cases are not available or not organised to enable effective planning and decision-making.

Beyond practical technological limitations, large-scale structural approaches to risk mitigation may decrease short-term risk while increasing the long-term risk. As the process that creates a natural hazard can rarely be eliminated, structural approaches only serve to postpone and effectively raise the hazard bar – an event might occur less often but when it does, it will overwhelm the structural fix and often lead to greater damage.

5.3 Ignition mitigation

A foundational component of effective wildfire management is ignition mitigation, specifically reducing the likelihood of wildfire outbreaks from intentional, accidental, and natural causes. Stopping or reducing wildfire outbreaks is the surest way of mitigating the detrimental impacts of wildfires. In many regions, this takes the form of community education and awareness-raising programmes, informing populations of the risk of wildfire ignition posed by day-to-day actions and activities, particularly under conditions of elevated fire weather. In some jurisdictions, laws are enforced during high fire risk periods to prohibit risky activities (such as use of naked flames or angle grinding in the open, or mechanical harvesting of crops, etc.) or restrict public access to high-risk areas such as nature reserves or national parks.

There is increasing recognition of the role of accidental failures in electricity distribution networks in wildfire ignitions. Reducing this risk has required operators to either install specialised equipment or place networks underground. Natural causes of wildfires, such as dry lighting, are not preventable and despite all best efforts to remove or minimise intentional and accidental ignitions, some risk of ignition will always remain.

Figure 5.1. Integrated wildfire management consists of five interlinked and often overlapping phases: review and analysis, risk reduction, readiness, response, and recovery. Review and analysis and risk reduction generally take place prior to the onset of the fire season and are focused on preventing avoidable outbreaks (e.g., ensuring fire ban compliance, improving understanding of arsonist behaviour, power line maintenance, and general fire safety awareness) ensuring communities and authorities are aware of the risk of wildfire and have thought about what needs to be done in the event of a wildfire. Readiness focuses on the actions to be carried out during the fire season before a wildfire breaks out and includes preparing properties, early detection of outbreaks, and having suppression resources on standby. Response includes the actions undertaken during a wildfire and focuses on reducing the potential impact of the wildfire through active suppression or relocating communities or assets under threat. Recovery (the planning for which should begin during the response phase) focuses on those actions to minimise disruption to the community, repairing infrastructure and restoring landscapes.

Integrated Fire Management - the 5Rs

Review and analysis

Collection of data and information	**Review of policies, procedures, and approaches to integrate fire management**	**Analysis of data and stakeholder engagement**
Development of fire behaviour models	Integrate fire management	Development of integrated fire management plans
Post-fire assessment and analysis		
Identify critical areas where intervention and investment are needed to support risk reducion		

Risk reduction

Awareness and education	**Landscape management**	**Fire use laws and enforcement**	**Community based fired management**
Ignition avoidance/restriction of high-risk activity	Fuel management	Building codes	Promote the safe management of fire through education
Personal evacuation plans	• Hazard reduction • Indigenous/traditional approaches • Grazing/mowing • Support ecological needs	Regulate fire use	Homeowner actions
Asset protection		Ignition reduction strategies	
Training	Firebreak creation and maintenance		
	Land use planning		
	Fire regime restoration and management		

Readiness

Fire surveillance and detection	**Threat/danger forecasting**	**Pre-suppression readiness**
Early warning systems	Fire danger rating systems	Firefighters on standby
Public notification		Personnel and equipment resourcing (capacity)

Response

Adaptive suppression	**Suppression resource**	**Community health and safety**	**Post-fire impact planning**
Safe	Capacity maintenance	Evacuation	Recovery assistance plans
Adequate • Rapid initial attack	Resource sharing/requests	• Emergency food, water, and shelter • Emergency health care	Loss assessment tools
Appropriate • Right resource mix		Support mobilization	
Effective • Contained and control, if possible			

Recovery

Community aid	**Environment**	**Infrastructure**
Emergency housing	Wildlife rescue	Loss assessment
Finance	Environmental restoration	Repair
Longer term recovery assistance plans	Fire regime restoration	

Illustration by Andrew Sullivan/CSIRO, 2021.

GRID-Arendal/Studio Atlantis, 2021

5.4. Firefighting and limits of suppression effectiveness

Firefighting incorporates all activities concerned with controlling, containing, and extinguishing a wildfire. It may also include the protection of threatened assets, such as homes, infrastructure, or natural values. While suppression is the focus of the "response" phase of fire management, it is also an important aspect of the "readiness" phase, as to be effective and efficient it must be well-planned and resourced with trained firefighters. Wildfires are suppressed using a variety of firefighting resource types with a range of capabilities that are suited to different environmental conditions, firefighting tactics, and strategies. These include ground crews using hand tools, earth-moving machinery, hoseline crews, and aircraft. Each resource may be used with a range of tactics in isolation or in combination. Most suppression tactics are undertaken to contain the spread of a fire and keep it from escaping containment. The aim is to stop fire spread either directly through action at the fire edge or indirectly through the manipulation of fuels ahead of a fire (Plucinski 2019b; Simpson et al. 2019). Direct attack actions may involve the application of water or water with additives onto burning fuels or the removal of unburnt fuels adjacent to burning fuels. This is only an option when the fire edge is accessible and fire behaviour is mild enough for firefighters to safely access. Indirect suppression is mostly achieved through removal of fuel at some distance from the fire edge, such as during burning-out operations, backburning or, the application of retardant chemicals in the path of a fire.

The effectiveness of all forms of wildfire suppression, including aircraft, reduces as fire behaviour becomes more intense. Firefighting resources are overcome by intense fire behaviour, usually due to spotting and radiant heat and their influence on firefighter safety. The limits of effective suppression have been expressed in terms of fireline intensity, with some variation between resource types, but all resource types are restricted to conditions below those experienced during extreme fire danger (Hirsch and Martell 1996). During these conditions, the deployment of suppression resources to directly attack the fire edge is futile and endangers the safety of firefighters. As a result, firefighters are restricted to tasks such as the protection of well-prepared houses and preparing containment lines in advance of the fire front. The containment of large and impactful wildfires is mostly achieved during periods when weather conditions (and fire behaviour) have moderated (Finney et al. 2009). There is still much to be learned about the effectiveness of wildfire suppression operations (Thompson et al. 2017; Plucinski 2019a).

5.5 Living in a fire-prone area: human behaviour and risk perception

When considering the role of human behaviour in creating and mitigating wildfire risk, it is important to recognise that there is no single action that leads to increased fire risk. In some locations, risk is increased by more people moving into a fire-prone area, placing more values at risk (i.e., increased vulnerability) (Radeloff et al. 2018). Conversely, in the African landscape, as more people move into an area, fires decrease as fuel is removed or fragmented (see Archibald et al. 2012). In other areas, the fire hazard is increased due to the loss of traditional fire management practices as people are excluded from or move away (often for economic reasons) from an area (Moreira et al. 2011). Elsewhere, increased fire risk may have little to do with population change but may instead be associated with changes in land management practices affecting the amount, type, or arrangement of vegetation available as fuel. Attempts to improve fire management outcomes that do not take these factors into account are unlikely to be effective. For example, decreasing fire risk may be more effectively achieved in the long run by supporting the continued use of traditional practices rather than creating centralised fire organisations or increasing spending on suppression capability.

Often a key focus for understanding different mitigation decisions is risk perception, with the assumption that the issue is lack of recognition of the risk. While risk perception is a factor in the decision process, it is well established that risk perception alone does not explain different decisions. In fact, a better dynamic to consider would be risk interpretation, given that the key issue of concern is how an individual might understand and respond to, or interpret, a risk differently (Eiser et al. 2012).

Beyond how the risk is perceived, risk attitude, self-efficacy (having the knowledge, resources, etc., to undertake the action), behavioural economics (i.e., monetary incentives) and response effectiveness (actions with the potential to make enough of a difference to merit the effort) can also shape an individual's decisions in response to wildfire (McCaffrey et al. 2020; Meldrum et al. 2021). It is important to note that when a risk is being considered, uncertainty is also an inevitable part of the equation, which puts further pressure on how decisions are made. Risk interpretation can also influence how supportive someone might be of different risk mitigation efforts, particularly those that may impact specific social or cultural values.

Overall, risk perception tends to be less influential than the level of knowledge about how and why a practice (e.g., prescribed fire) is being used and the level of confidence in those who are implementing the action. Because of this, effective outreach efforts – that is, ones that are interactive and take local context (including gender differentiated cultural roles) into account – are a critical component of any endeavour aimed at wildfire risk mitigation (McCaffrey and Olsen 2012). A study in North-Eastern Namibia demonstrated that women and men have different roles to play in fire lighting. In the region, most of the fires were lit by women running contrary to earlier assumptions that led to fire management programs that targeted men (Heikkilä et al. 2007).

The concept of fire-adapted communities appeared in the USA in the early 2000s, partly due to the realization that effectively addressing the wildland fire threat required engaging a range of affected stakeholders who should work together in the face of evolving threats (planning and preparing, responding, and recovering) (Wildland Fire Executive Council [WFEC] 2014).

There are several interventions available for mitigating fire risk to the human-built environment, including:

- building regulations (fire-resilient materials and design, smoke infiltration and water supply protection, backup energy, etc.)
- inclusive community engagement programmes (efforts to increase knowledge and capacity among all community members to prevent, prepare for, and mitigate wildfire risk, including identifying and restricting high-risk activities, undertaken by both men and women, that can lead to fire ignitions, making structures more ignition resistant, preparing evacuation plans, and designating safer shelters)
- fuel treatment activities (e.g., managing vegetation around structures, fuel breaks or prescribed burning to reduce the risk in high fire risk areas)
- land-use planning (where to build in the landscape, distance from vegetation, preservation of open space buffers, etc.)
- social networks and local communication mechanisms during emergencies (Castillo et al. 2020).

5.6 Managing fuels to minimise wildfire risk

The technological limitations of wildfire suppression, particularly under severe fire weather conditions, are widely recognised, as described in the previous section. Even large air tankers that can drop 10,000–25,000 litres of suppressant or retardant on a fire have been found to only be effective at intensities up to 3–5 per cent of the peak intensity of an extreme wildfire (Loane and Gould 1985), primarily due to firebrands breaching the containment line. Strategies to address the fuel component of the fire behaviour triangle (see Figure 1.2) include landscape fuel management to reduce the amount of vegetation and vegetative debris available as fuel or to change its arrangement. Such fuel treatment methods include lighting intentional fires under mild conditions (i.e., prescribed fire or hazard reduction burns), managing selected wildfires rather than immediately suppressing them (sometimes referred to as "natural" prescribed burns), mechanically thinning vegetation, the continued watering of low flammability vegetation to create green fuel breaks, particularly strategically around dwellings (Gibbons et al. 2018), and the application of grazing practices (Curran et al. 2018; Cui et al. 2019).

One of the key challenges of any fuel treatment intervention is understanding that its effectiveness is highly dependent on local conditions and context as well as the behaviour and burning conditions of any subsequent wildfire (Barnett et al. 2016). The selection of methods, and how and where they are applied, depends on vegetation type and structure, environmental constraints, likelihood of impact and effect, and socioeconomic factors (Florec et al. 2019). Fuel treatment effectiveness also has a lifespan, which relates to the rate at which fuel re-accumulates. Fuel treatments specific to reducing fire risk have a common goal to decrease fire intensity, reduce spread, increase likelihood of firefighting success, and protect assets (Agee and Skinner 2005; Moghaddas and Craggs 2007) (Figure 5.2). Fuel treatments do not necessarily stop fires and communities must not solely rely on a fuel treatment approach for protection (Boer et al. 2015).

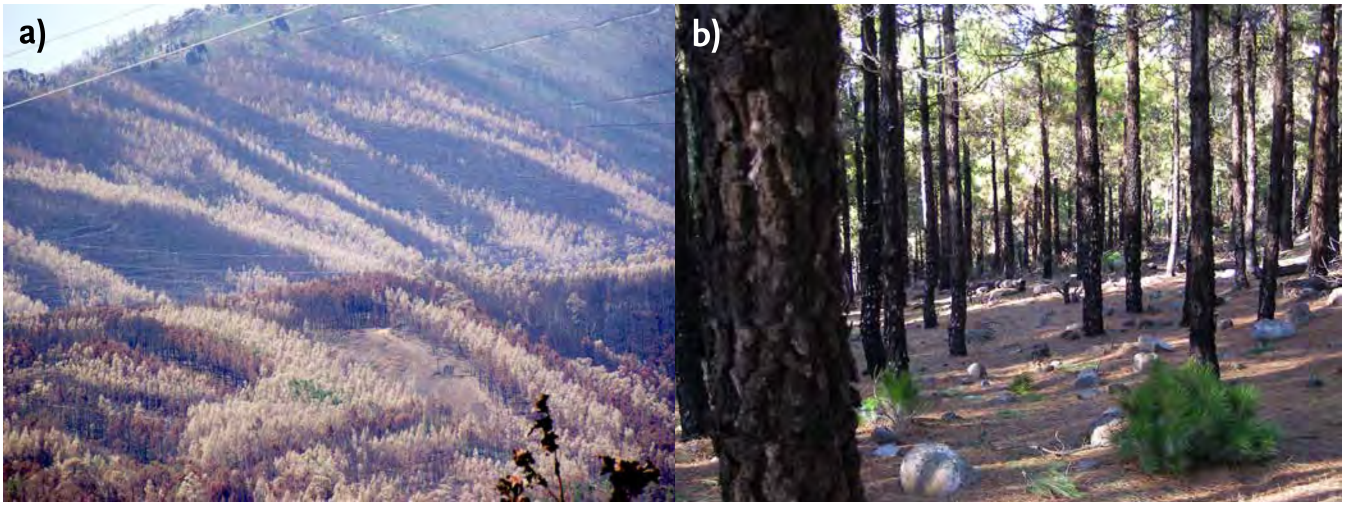

Figure 5.2. Fast-growing forest plantations in Mediterranean-type climates are a significant fire risk in the absence of substantial and regular fuel treatments. (a) The Pedrógão Grande fire in June 2017 in Portugal killed 66 people in a landscape dominated by pine and eucalypt. (b) Tree species with well-developed fire resistance traits such as thick bark – for example, Canary pine (Pinus canariensis) – are the ideal candidates for hazard reduction burning programmes in both natural and planted forests.

Research across the world indicates that, regardless of how they are conducted, fuel treatments are only effective in reducing fire risk if the modified fuel structure (including reduced fuel load) can counteract the influences of weather and drought on fire behaviour (Boer et al. 2009; Kalies and Kent 2016). However, reduced or modified fuels can aid initial-attack fire suppression. This may allow firefighters to take advantage of more moderate night-time weather conditions to work more safely to control fire spread or undertake indirect attack by backburning.

Under conditions of extreme fuel dryness and strong wind (typical of high-intensity wildfire weather conditions), the rate of spread of a fire in forest or shrubland can be predicted reasonably well without reference to fuel structure descriptors (Cruz and Alexander 2019; Cruz et al. 2020). This suggests that under these burning conditions fire spread is not affected by the state of the fuel in a quantifiable way, although fire intensity will still be mitigated as it is directly proportional to fuel consumed. Risk reduction becomes less effective the more extreme the fire weather (Fernandes and Botelho 2003; Fernandes 2015; Hunter and Robles 2020). Therefore, alternative risk management strategies are required to mitigate potential fire impacts arising from the predicted increase in extreme fire weather related to climate change.

5.6.1. Hazard reduction burning

Hazard reduction burning is a type of prescribed burning (which also includes intentional burning to maintain fire-related ecosystem processes and the restoration of fire-adapted/dependent ecosystems). Hazard reduction burning is usually the treatment of choice where landscape-level fuel management is practised, due to its competitive cost-to-benefit ratio at that scale. However, it can involve difficult trade-offs between objectives, such as the protection of people and property, conservation of biodiversity, and air quality during burning (Whittaker and Mercer 2004; Williamson et al. 2016). It can also be a risky operation with sometimes uncertain outcomes (including escaped fire, which itself can become a wildfire) and is therefore often the subject of debate (Altangerel and Kull 2013; Morgan et al. 2020).

Unlike wildfires, smoke impacts from prescribed burns are often more localised and can be managed to decrease exposure and duration. Smoke forecasting methodologies can be used to determine smoke transport pathways, helping managers identify the most appropriate wind conditions for conducting burning (e.g., Odman et al. 2018; State of Victoria,

Case study: Co-designing a wildfire management and risk alert system in Amazonia

In many places, organisations that fight fires are separated from groups or institutions that examine fire mitigation (risks, monitoring, probability, impacts, and communications). In Brazil, a conceptual framework for decreasing the risk and impacts of wildfires has been developed, utilising the expertise of stakeholders. The framework shifts the focus from solely fighting fire threats to mitigating fire risks. It is based on five interconnected pillars: risk knowledge, monitoring and alerts, education and communication, prevention capacity, and response/reaction capacity (Anderson et al. 2019).

Three online wildfire management tools were developed through projects under this framework. The first project, developed in south-western Amazonia, aimed to co-develop an online, open access fire risk and management platform based on the five pillars, to be promoted via education activities in secondary schools. The platform combines a series of meteorological data associated with the increase in fire risk (e.g., number of days without rain, air temperature, rainfall forecast), the fire location given by satellite-based detection, land tenure information (e.g., rural private properties, protected areas) that can potentially be used to attribute legal responsibility for the fire, land cover data to extract "what has burnt", and other complementary data, such as air quality, location of roads, rivers, and schools, and health infrastructure. The platform can be accessed at: http://terrama.cemaden.gov.br/griif/mapfire/monitor/.

The second project concerned the development of seasonal fire probability alerts for South American protected areas. Fire reports are disseminated via an online platform to share data and an alert status. The categorization of risks in terms of alert levels is key to supporting the identification of priority areas, although uncertainties in fire forecasts must be taken into consideration.

The third project focused on building capacity and developing guidelines on how to mitigate fires, based on the capacity and limitations of the institutions involved. It is hoped that sharing common fire risks and impact scenarios that show where fires are more likely to occur and what their impacts could be will facilitate communication among stakeholders, improving the transparency of information and subsidizing data-based decision-making. Moreover, as the tool is online and has downloadable data, it can be used by the public to increase local awareness and preparedness and to conduct research.

Department of Environment, Land, Water and Planning 2019; Hu et al. 2019). However, the conditions that are most suitable for a hazard reduction burn (moderately dry fuels, mild temperatures, and calm winds) can lead to smoke remaining trapped at the surface rather than being dispersed. The advanced notice of planned burning can help decrease potential short-term health impacts by enabling sensitive individuals to take appropriate protective measures, such as remaining indoors with doors and windows closed. Targeted smoke management programmes can minimise the potential for prescribed burn smoke impacts that exceed air quality standards or affect sensitive populations.

5.6.2. Barriers to fuel treatment as risk mitigation

There are a range of obstacles to utilising fuel treatments, such as prescribed fire, as risk mitigation (Schultz et al. 2019). In the aftermath of a severe wildfire season, attention tends to focus on improving responses, particularly fire suppression (Calkin et al. 2015; Robinne et al. eds 2018). This creates competition between emergency responses and land management resources, both among and within agencies (Driscoll et al. 2010; Bowman et al. 2013), which often results in reduced forest management capacity in general, and fuel management in particular (Calkin et al. 2011; North et al. 2015). Governments responding to wildfires by spending more on suppression (usually with the primary objective of protecting human life and property) via technological solutions, such as aircraft, usually means they have less funds for preparation and mitigation strategies, such as fuel management (Ingalsbee and Raja 2015). Risk aversion in relation to conducting hazard reduction burning has also fluctuated due to concerns regarding sustainability, ecological costs, health impacts and the potential for litigation when the burning gets out of control (Morgan et al. 2020). Loss of institutional experience through natural attrition and reduced exposure to operational burning of new staff more generally increases risk-averse decision-making.

As with many mitigation initiatives to reduce fire risk, especially in developing countries, the prohibitive costs mean they are often not sustainable over the long term. For this reason, many projects in fuel management usually fail within one or two years as there are no actions and mechanisms for maintenance, monitoring, or follow-up. This happens when there is little budget or when projects compete with other activities classified as higher priority (Castillo et al. 2017).

Insurance as a mitigation tool

Despite the common belief that insurance may have a role in changing individual behaviours and thereby contribute to wildfire mitigation, it is not as straightforward as many may believe. There is an assumption that homeowners will mitigate risk more if they are concerned about losing their policy or their premiums increasing, but there is little empirical evidence to support this. Insurance is only effective as a mitigation tool if insurance companies have the interest or capacity to assess the risk at the individual property level, which is generally not the case.

Although there are indications that policies are harder to get and more expensive in some fire-prone areas, such as California, it is often supposed that high insurance costs or the inability to obtain a policy will stop people living in these areas. This may be true for individuals who are more financially risk-averse, but there is no evidence available to show that this is the case, Wealthy individuals can self-insure and those less wealthy just take their chances. Anecdotal evidence suggests that for the less wealthy, the lack of insurance may prompt them to mitigate wildfire risks more, but it may also make them more likely to plan to stay to protect their property if a fire occurs, since they may not be able to absorb the potential loss. Depending on the circumstances (e.g., staying to defend property in extreme conditions), there is potential for increased fatalities.

Perhaps more important in the long-term is the question of equity. The logic that raising insurance rates discourages people from living in the wildland-residential interface essentially suggests that only the wealthy can live there. Higher insurance rates or the inability to obtain insurance does not appear to make it more likely that people will move out of fire-prone areas, but it would increase the vulnerability of many already-vulnerable populations.

Insurers rely on catastrophe modelling (CAT models), which combine hazard models with existing asset vulnerabilities to predict losses. For wildfires, CAT modellers have been limited by the lack of reliable wildfire models, while wildfire researchers interested in risk evaluation have limited access to actuarial data. Better collaboration between wildfire scientists and actuaries could enhance CAT modelling and mitigation efforts. Due to a lack of data, accurately determining changes to insurance premiums that would lead to a mitigating effect is not currently feasible, but it is an active area of research for the insurance industry (e.g., Tamm and Klose 2019).

Power line failures and wildfires

Power line fires, although relatively rare on an annual basis, are a major cause of large, fast-moving and damaging fires during periods of high fire hazard. Miller et al. (2017) found a large statistical over-representation of electrical fires in south-eastern Australia when fire weather conditions were high. According to a Royal Commission report into the 2009 Black Saturday fires in Australia (Teague et al. 2009), power lines were the cause of 45 per cent of fires during the 1955 Black Sunday Fires, 56 per cent of the fires that occurred in Victoria in 1977, and half of the Ash Wednesday bushfires in 1983. In southern California, 9 of 20 major fires that occurred during the October 2007 "fire siege", which burnt more than 3,100 structures and killed at least seven people (Keeley 2009), were caused by power line failures (Mitchell 2009). More recently, the Wine Country Fires in northern California, which killed 44 people and caused at least US$14.5 billion in damages, were almost entirely (11 of 12 named fires) caused by power line failures during an extreme autumn windstorm. The Wine Country Fires, and the even more destructive Camp Fire the following year, which was also caused by a wind-driven power line failure, forced California's largest energy provider into bankruptcy. The significant fires in Chile in January and February 2017 were also largely attributed to ignitions caused by power line failures.

Finding the best way to mitigate the risk of power line failures during periods of critical fire danger has been a subject of debate for decades. Power lines are operated under rigorous safety standards in most countries, with standards often updated in response to increasing risk resulting from rising populations and higher voltage lines in high fire-risk areas. However, climate change-driven extreme weather events (especially damaging winds), combined with increases in the length of the fire season and decreases in late-season fuel moisture are resulting in an increased risk of power line failure. Inspection and maintenance often fall to the power line owner, with poorly maintained lines continuing to cause fires. For example, the power lines that caused the 2018 Camp Fire in California's Sierra Nevada (which killed 85 people, burnt nearly 19,000 structures, and caused US$16.7 billion in economic damage) were built in 1919 and had been due for replacement for over 20 years (Cowan 2019).

Although power lines could be buried in known wind corridors, the costs of doing so are exceptionally high (> US$1 million/km). Efforts by utility companies in the USA to pass costs on to customers have been denied in many areas, with regulators citing them unreasonable (Kousky et al. 2018) and instead forcing companies to focus on more stringent monitoring of power line conditions and the weather. When power line problems are detected, or when fire weather conditions become extreme, some utility companies have taken to proactively shutting off power to affected areas to reduce the risk of fire ignition. While this may reduce the companies' exposure to financial loss and litigation, it causes major problems for electricity users, especially when fires do occur. Loss of electricity is of particular concern for health-care facilities and other infrastructure, such as water supplies. In the wake of the 1983 Ash Wednesday wildfires, the South Australian Government implemented aerial bundling of cables to reduce the risk of line clashing in high-risk areas. Following the 2009 Black Saturday bushfires, the State of Victoria invested heavily in upgrading the electricity distribution network across the state to reduce the likelihood of ignition. Measures included the programmed disabling of automatic circuit reclosers during high-risk periods (Roozbahani et al. 2015). The recent review by Arab et al. (2021) revealed that while there is extensive research on power grid disaster risk management, there is a gap in understanding risk from a wildfire perspective. Due to the increasing risk of wildfires, they concluded that a multidisciplinary approach was needed to address the problem, especially power line-related ignition, focusing on ageing infrastructure assets and limited resources.

Case study: Indigenous cultural burning for wildfire prevention, mitigation, and response

Fire has always played a significant role in the lives of many Indigenous peoples around the world. For some peoples, their understanding and use of fire is entrenched in creation stories and practice. Although there is little documentation of Indigenous groups using fire to manage wildfire events, evidence from oral histories and Western science shows that many Indigenous peoples actively engage in landscape burning to achieve diverse and varied cultural, social, and environmental objectives, including a reduction in wildfire occurrences (Roos et al. 2021). In Australia, the use of fire to create mosaic landscapes for hunting and gathering purposes (e.g., Jones 1969; Bowman et al. 2004) also broke up the continuity of fuels and so inhibited the extensive spread of wildfires. In Canada, there is evidence of Indigenous peoples using fire as a way of managing their territory (Miller and Davidson-Hunt 2010). Indigenous peoples from Amazonia in Venezuela, Brazil (including savanna regions), and Guyana, have used fire for subsistence activities and the control of savanna plant fuel levels to prevent the spread of wildfires into adjacent forests (e.g., Bilbao et al. 2019).

Despite colonisation and ongoing colonial legacies that have resulted in the loss of traditional lands/territories and disrupted cultural fire practices (among other things), some Indigenous peoples have retained aspects of their culture, language, and knowledge (including of fire), with others proactively reclaiming and revitalizing their Indigenous heritage. Many Indigenous fire managers, stewards and knowledge-holders draw on place-specific knowledge, cultural protocols, and practices for the safe and appropriate use of fire (e.g., for agriculture, community well-being, the protection of sacred and special sites; see Langer and McGee 2017; Firesticks Alliance Indigenous Corporation 2020).

Government and community support for Indigenous involvement and leadership of on-ground management of and decision-making on wildfires differ across continents and between nations. In Brazil, for example, many Indigenous groups (e.g., the Xavante) are trained in total fire suppression (Welch and Coimbra Jr. 2019), whereas in south-east Venezuela, the Pemón use patch mosaic burning to protect and sustain forests in Canaima National Park, which helps reduce the impacts of wildfires in the region (Bilbao et al. 2010; Bilbao et al. 2020).

Recognition of the limits of suppression as a tool to control wildfires has increased the recognition of the value of Indigenous fuel management methods, as well as the need to learn from and support such practices. For example, in South America the Participatory and Intercultural Fire Management Network (PARUPA), endorsed by Indigenous peoples, academics and civil servants from Brazil, Guyana and Venezuela, promotes the weaving of traditional, adaptive indigenous fire knowledge with scientific knowledge and institutional technical capacities into a fire management plan for Indigenous territories (Bilbao et al. 2019). In Canada, a national team of wildfire management experts has developed the Blueprint for Wildland Fire Science in Canada (2019–2029) (Sankey 2018). One of the six priority research themes of this blueprint is recognizing Indigenous knowledge, which represents the first national recognition of the need to collaborate with Indigenous peoples for better wildfire management.

In the United States of America, several key initiatives such as the Amah Mutsun Land Trust, the Cultural Fire Management Council, the Karuk Tribe's Eco-Cultural Revitalization Branch, and the Nature Conservancy's Indigenous Peoples Burning Network, have developed opportunities to support Indigenous burning activities as a way to engage in wildfire prevention and mitigation at the landscape level. Recent government planning for climate resilience further supports the need for Indigenous burning (e.g., Bedsworth et al. 2018; Goode et al. 2018).

Recommendations to enhance Indigenous leadership in this space, and importantly to recognise that the relationship between Indigenous peoples, landscapes and fire are diverse and varied between and within nations, include:

- programmes to empower Indigenous fire knowledge and management practices (e.g., cultural burning) for land management, including wildfire prevention, migration, and response
- support for building collaborative partnerships across and between sectors to enable Indigenous fire managers, stewards, and knowledge-holders to work with other fire managers across different land tenures
- processes that acknowledge the role of Indigenous cultural values and knowledge to inform the development of strategies to provide protection and involvement with disaster management and risk reduction
- recognition of Indigenous sovereignty and self-determination to build knowledge about and use fire to manage their traditional estates and territories
- support for developing integrated, gender-responsive, coordinated and intersectoral fire management strategies and approaches that include multiple perspectives, knowledge, and actors, including the adaptive practices of Indigenous cultures and communities living in rural territories
- protection for Indigenous fire knowledge systems as an important cultural asset of possible adaptive solutions to climate change, including protected area management.

In some countries, however, some Indigenous leaders remain sceptical about how recognition of cultural fire management will influence centralized decision-making. Nevertheless, there are growing opportunities for Indigenous peoples and their fire knowledge to be recognised within government policies, practices, and programmes, with such opportunities likely resulting in multiple benefits (for example, cultural, spiritual, social, economic, political self-determination and health and well-being) for Indigenous fire managers, their communities, and their land (e.g., Maclean eds. 2018; Christianson et al. 2020).

Meadow burning in Treaty 6 territory, Alberta, Canada.

Australian Indigenous cultural fire management

Aboriginal Australians skilfully use fire to adaptively manage their local environments. Their cultural fire management (cultural burning), which includes numerous slow, cool burns that result in a mosaic patchwork landscape over time, has been used for millennia. Recently, Indigenous leaders have advocated for a central role for Indigenous fire managers in wildfire prevention, mitigation, and response (see Firesticks Alliance 2020; Steffensen 2020). Interest in Indigenous cultural fire management for wildfire prevention and mitigation increased during and after the Black Summer wildfires in 2019–2020. The Independent Bushfire Inquiries (see New South Wales Government 2020; Inspector-General for Emergency Management 2020) and an independent study on climate and disaster resilience commissioned by the Australian Commonwealth Government (see Commonwealth Scientific and Industrial Research Organisation [CSIRO] 2020) each considered the important role that Indigenous leaders' cultural fire management (and knowledge) could play in wildfire prevention, mitigation, and response.

Indigenous partnerships across Australia support learning and application of cultural fire knowledge and practices for biodiversity conservation and, increasingly, wildfire prevention and mitigation via the development of "on-country[21] Indigenous fire enterprises" (i.e., Indigenous ranger projects; Robinson et al. 2016). These are facilitated through the Northern Australian Indigenous Land and Sea Management Alliance (see North Australian Indigenous Land and Sea Management Alliance [NAILSMA] 2020), the Federation of Victorian Traditional Owner Corporations (see Victorian Traditional Owner Cultural Fire Knowledge Group 2020), the Firesticks Alliance Indigenous Corporation (Firesticks Alliance Indigenous Corporation 2020), as well as fire partnerships and activities coordinated through a range of indigenous-led initiatives (see Maclean et al. 2018; Firesticks Alliance Indigenous Corporation 2020). Cultural burning activities are conducted by variety of Indigenous-led organisations. Work is carried out through a series of partnerships (including with government agencies, scientists, non-governmental organisations and private landholders) which are mainly funded via national government programmes (for example, National Landcare, Indigenous "Working on Country", and Indigenous Protected Areas programmes; see Maclean et al. 2018 for details) and State and territory government programmes (see, for example, Neale et al. 2018; Robinson et al. 2020). The partnerships reflect the growing recognition that increasing support for Indigenous cultural fire management, including the capacity of Indigenous and non-Indigenous fire managers to work together, is important for building Australian climate and disaster resilience and capacity for wildfire prevention, mitigation, and response.

Oliver Costello (Bundjalung) Co-founder of Firesticks and a Director of Jagun Alliance Aboriginal Corporation, at a cultural burn on Bundjalung Jagun (also known as Dunoon, New South Wales, Australia).

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[4] Indigenous Australians call their traditional land, sea, and freshwater territory "country" (e.g., country, salt water country, freshwater country).

5.7 Wildfire mitigation in water catchments

Like many landscape managers, water catchment managers must seek the most appropriate level of fire activity to preserve forest health and thus maintain the desired yield and quality of water from the catchment (Nunes et al. 2018). In catchments where forest ecosystems depend on high-severity fires, there will be a trade-off, which is an active area of research (Gannon et al. 2019; Neris et al. 2021; Rakhmatulina et al. 2021). An atypical fire regime might protect forest cover but could also change the ecosystem altogether (Gresswell 1999), thereby impacting long-term water yield and quality. If too little fuel management is carried out and a high-intensity wildfire spreads through a water-supplying catchment, water yield and quality may be adversely affected for several years or even decades.

In any case, restoring and maintaining an appropriate level of fire activity in water catchments helps manage fuel amounts and restrict fire behaviour, which also makes potential wildfires easier to manage (Oliveira et al. 2016; Barros et al. 2018; Gannon et al. 2020). In instances where broadacre hazard reduction burning cannot be used to manage fuels, mechanical options can be used instead. Although mechanical options may not have the same range of positive effects as well-planned and well-executed prescribed fires, their detrimental impact on water resources tends to be limited (Santos et al. 2015; Hahn et al. 2019).

Fuel management – regardless of the method used, including animal rewilding or grazing – may also reduce overall water consumption by live vegetation, thereby allowing for increased water storage and yield (Simonit et al. 2015; Ellison et al. 2017; Boisramé et al. 2019; Jones et al. 2020). Also known as landscape rewetting, targeted reductions in biomass that aim to increase water storage can help buffer against dry periods by preserving soil and vegetation moisture, increasing the density of wetlands and open water bodies. Forest thinning, wetland restoration, and the reintroduction of landscape-engineering species (e.g., beavers) are possible avenues (Fairfax and Whittle 2020) to enhance water storage capacity within forested watersheds, with the beneficial side effect of reducing general sensitivity to fire.

Complementary actions can also be taken downstream of fire-prone catchments. For instance, efforts can be made to retrofit water infrastructures so they can withstand post-fire water quality degradation and other extreme climate events that have detrimental cascading effects (Becker et al. 2018; Robinne et al. 2021). In many areas, building such infrastructure would also likely support other water security aspects (Bhaduri et al. 2016; Becker et al. 2018).

In water-supplying catchments that have experienced extreme fire events, the ability to measure, monitor and mitigate post-fire issues is paramount. Remote sensing is the primary tool used to measure wildfire severity and its physical impact on the vegetation of a catchment. Ideally, this should be complemented by on-site severity measurements and a network of hydrometric sensors upstream and downstream of the burnt perimeter. Combined with predictive models, critical soil erosion zones and downstream areas susceptible to flash floods and debris flows can be identified, and erosion and run-off mitigation mechanisms put in place (Miller et al. 2016). In many situations, it is crucial to implement pre-emptive interventions immediately after a fire to limit possible hillslope erosion from major rainfall events. The use of mulch, for example, has shown positive results and is easily accessible (Robichaud et al. 2013a; Robichaud et al. 2013b). Forest restoration in recently burnt areas critical to water supply also seems a valuable option, though more work is needed on this (Scheper et al. 2021).

Global emergency wildfire cooperation

The global fire management community has long recognised the importance of international cooperation in mitigating the growing risk and threat of wildfire around the world. Greater international collaboration provides enhanced fire prevention capability by sharing critical fuel, weather and fire knowledge for prevention and pre-suppression planning, along with the techniques, methods and processes that underpin risk reduction activities. Suppression capacity may be enhanced through sharing expertise and resources, while suppression effectiveness is increased by sharing advances in training and science, and technology transfer.

Global cooperation in fire management involves a series of formal and informal bilateral and multilateral arrangements, many of which have evolved over a number of decades, between countries and jurisdictional bodies. Several actors are involved in these international arrangements, with many originating from federal and provincial land management agencies with fire management responsibilities who have created cross-border agreements with neighbours and others for mutual benefit, primarily involving resource-sharing and surge capacity uptake. Other arrangements include those of the United Nations Office for the Coordination of Humanitarian Affairs (OCHA) and the United Nations Office for Disaster Risk Reduction (Secretariat for the Sendai Framework). European Union efforts include the European Civil Protection and Humanitarian Aid Operations through the Emergency Response Coordination Centre (ERCC). With the exception of the ERCC, these arrangements do not deal specifically with wildfires, though OCHA has supported coordination around some large wildfire incidents where support was sought.

The United Nations has a broader focus on disaster risk reduction, which includes wildfires, with the FAO Forestry Division in particular having land and fire management as a technical mandate. The World Meteorological Organization provides support through forecasting, warnings, and data standards. The United Nations Environment Programme (UNEP)/OCHA Joint Environment Unit (JEU) is responsible for international coordination on environmental emergencies and has a Post-Conflict and Disaster Management Branch and an Advisory Group on Environmental Emergencies that brings together environmental experts and disaster managers to improve risk reduction, readiness, and response. The United Nations Development Programme has implemented fire activities and UNECE convened a Forest Fire Specialist Team until 2014. Other international bodies react to wildfires with interventions when the scale, impacts or profile of wildfires creates sufficient attention. The World Bank has been involved in readiness, response, and post-disaster recovery and reconstruction. The International Tropical Timber Organization (ITTO) promotes sustainable development through tropical forest management, has developed the ITTO Guidelines on Fire Management in Tropical Forests (1997), and continues to support fire management projects.

The international scientific community has also been active with regular collaboration and connection, including through various bodies such as the International Union of Forestry Research Organizations (IUFRO) Unit 8.03.05 "Forest Fire", the Global Observations of Forest and Land Cover Dynamics – Fire Mapping and Monitoring under the Group on Earth Observations (GEO) and regular international research conferences.

At the regional level, the European Union Civil Protection Mechanism has been used since 2001, the Association of Southeast Asian Nations (ASEAN) has an Agreement on Transboundary Haze Pollution that was adopted and came into force in November 2003, and 14 Member States of the Southern African Development Community (SADC) seek to provide a framework for cooperation on fire management issues.

There has been significant interaction on fire management through training, research, capacity-building, and study tours, among others. The greatest potential for coherent and consistent improvements in fire management is likely to be through interactions and exchanges, joint problem solving and sharing experiences in fire management and research. The International Association of Wildland Fire allows wildfire researchers and practitioners to interact in formal and informal forums.

Case study: The role of communities in wildfire monitoring and management in Indonesia

Indonesia is too familiar with the tragic price of wildfires. In 2015, for example, extensive forest and peatland fires affected the health and livelihoods of millions of people and caused billions of dollars of damage (Glauber et al. 2016). Tanjung Jabung Barat regency, on the east coast of Jambi Province on the island of Sumatra is at high risk because it sits on vast peatlands that cover two fifths of the total land area. The region has experienced widespread deforestation from logging and encroachment by agribusiness and small-scale farmers, losing 76,000 hectares or 34 per cent of its forest cover between 2001 and 2019 (Global Forest Watch 2020). The deforested peatland is prone to fires during the dry season. When the normally high peatland water table is lowered by drainage, often exacerbated by drought, the risk of ignition rises. Smouldering peat fires can persist below the surface for weeks or months and are difficult to extinguish. Most are not fully suppressed until the onset of the wet season and the recovery of the water table.

Although the Indonesian Government has banned the use of fire for land management, the practice continues because of tradition and convenience (Tata et al. 2016). Between 2017 and 2019, intentional burning resulted in a fourfold increase in the annual number of fire outbreaks in Tanjung Jabung Barat (Badan Nasional Penanggulangan Bencana [BNPB] 2020). Efforts to monitor and manage fires are further undermined by indeterminate law enforcement and unclear land tenure, which people take advantage of to use fire to clear and claim land when ownership is uncertain.

Together, these risk factors highlight the need for forest communities to monitor and prevent fires themselves, and to be able to manage fires through integrated fire management (Ganz 2020). Forest communities are the first to detect wildfires and are the first line of defence. They are also the communities most affected by wildfires. In 2003, Indonesia launched the Fire Care Community programme (Masyarakat Peduli Api – MPA), which established a regional fire alert system that engaged companies and civil society organisations in training forest communities and equipping them to monitor and mitigate fires (Budiningsih et al. 2020).

The Regional Community Forestry Training Centre (RECOFTC) worked with Global Forest Watch to provide the Forest Watcher app to communities in Tanjung Jabung Barat in a pilot initiative that ran from October 2019 to August 2020. The app allows users to identify and share information on fire threats. The communities have made several reports of increased fire risk through Forest Watcher, including cases of peatland clearing, forest occupation, illegal logging, declining peat water levels and dried-up wells that should have held water for fire control. In response, authorities have acted to remedy these issues, thereby helping to mitigate the fire risk.

The case of Tanjung Jabung Barat shows that forest communities can prevent and mitigate wildfires when trained and equipped with tools to overcome connectivity barriers. To fulfil this role, they also need their reporting tools to be integrated with regional and national mapping databases that track wildfires and the haze that they cause. In this way, forest communities and local governments can quickly identify threats and make evidence-based decisions to manage them effectively.

Appendix

Future burnt area and fuel load maps

The ConFire Bayesian fire model (Kelley et al. 2019; Kelley et al. 2021) was run using output from four climate models taking part in the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2b) (Frieler et al. 2017).

The ISIMIP2b Protocol was developed in response to the IPCC Special Report on the 1.5°C target for temperature increase above pre-industrial levels (IPCC 2018). RCP2.6 is the lowest emission scenario considered within the Coupled Model Intercomparison Project 5 (CMIP5), in line with the Paris target. RCP2.6 allows for a potential overshoot before returning to below 1.5°C (Rogelj et al. 2015). RCP6.0 represents a no-mitigation scenario. The Shared Socioeconomic Pathway 2 (SSP2) storyline represents middle-of-the-road socioeconomic development concerning population, mitigation, and adaptation challenges (O'Neill et al. 2014).

The Joint UK Land Environment Simulator Earth System (JULES-ES) land surface model (Best et al. 2011; Clark et al. 2011; Harper et al. 2016; Sellar et al. 2019) simulated vegetation cover and soil moisture through daily calls to the dynamic vegetation routines instead of the standard every 10 days (Mathison et al. in preparation). ConFire's Bayesian inference step was run as per Kelley et al. (2021), though only over 10 per cent of randomly sampled parts of the globe as per Kelley et al. (2019). The optimization was made against the fourth version of the Global Fire Emissions Database (GFED4s) using ISMIP2b climate and corresponding JULES-ES land surface output for 1997–2006 (the overlapping period of ISMIP and GFED4s). ConFire expresses outputs as probability distributions. "Significance" is expressed as the percentage dissimilarity between historical and future probability distributions, defined as one less the square-rooted product of both distributions. Climate simulations from ISIMIP2 were bias-corrected to EWEMBI (EartH2Observe, WFDEI and ERA-Interim data Merged and Bias-corrected for ISIMIP; Lange 2019; Frieler et al. 2017), and ConFire burnt areas were bias-corrected to 1997–2002 against GFED4s as per Kelley and Harrison (2014).

Wildfire events were defined by gridcell as the burnt area where there was a 1 per cent likelihood or greater of burnt area between 2010 and 2020 according to ConFire. The change in wildfire events in the future is the change in likelihood that burnt areas exceed this threshold.

References

Recommendations

Agency for Integrated Rural Fire Management (2020). National Plan for Integrated Wildland Fire Management. Lisbon. https://www.agif.pt/app/uploads/2019/05/PNGIFR_ENGLISHVERSION_menor.pdf

Behlert, B., Diekjobst, R., Felgentreff, C., Manandhar, T., Mucke, P., Pries, L. et al. (2020). World Risk Report 2020. Bündnis Entwicklung Hilft and Ruhr University Bochum – Institute for International Law of Peace and Conflict. https://weltrisikobericht.de/wp-content/uploads/2020/12/WRR_2020_online_.pdf

Thomas, D., Butry, D., Gilbert, S., Webb, D. and Fung, J. (2017). The Costs and Losses of Wildfires: A literature review. Special Publication 1215. National Institute of Standards and Technology. U.S. Department of Commerce.

Chapter 1

Bond, W.J., Woodward, F.I. and Midgley, G.F. (2005). The global distribution of ecosystems in a world without fire. New Phytologist, 165(2), 525-538. https://doi.org/10.1111/j.1469-8137.2004.01252.x.

Booysen, P.D.V. and Tainton, N.M. (eds.) (1984). Ecological Effects of Fire in South African Ecosystems. Ecological Studies Series. Berlin: Springer-Verlag.

Bowman, D.M.J.S., Balch, J., Artaxo, P., Bond, W.J., Cochrane, M.A., D'Antonio, C.M. et al. (2011). The human dimension of fire regimes on Earth. Journal of Biogeography, 38(12), 2223-2236. https://doi.org/10.1111/j.1365-2699.2011.02595.x.

Burrows, N.D. (1994). Experimental Development of a Fire Management Model for Jarrah (Eucalyptus Marginata Donn ex Sm.) Forest. Doctoral Thesis. Canberra: Australian National University. https://doi.org/10.25911/5d78d89a1d6c0.

Byram, G.M. (1959). Combustion of forest fuels. In Forest Fire: Control and Use. Davis, K.P. (ed.). New York, NY: McGraw Hill. Chapter 3. 61-89.

Chuvieco, E., Lizundia-Loiola, J., Pettinari, M.L., Ramo, R., Padilla, M., Tansey, K. et al. (2018). Generation and analysis of a new global burned area product based on MODIS 250 m reflectance bands and thermal anomalies. Earth System Science Data, 10(4), 2015-2031. https://doi.org/10.5194/essd-10-2015-2018.

Countryman, C.M. (1966). The concept of the fire environment. Fire Control Notes, 27(4), 8-10.

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., Ayres, M.P., Flannigan, M.D. et al. (2001). Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience, 51(9), 723-734. https://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2.

Grime, J.P. (1979). Plant Strategies, Vegetation Processes. Chichester: John Wiley and Sons.

Hollis, J.J., Matthews, S., Anderson, W.R., Cruz, M.G. and Burrows, N.D. (2011). Behind the flaming zone: predicting woody fuel consumption in eucalypt forest fires in southern Australia. Forest Ecology and Management, 261(11), 2049-2067. https://doi.org/10.1016/j.foreco.2011.02.031.

Intergovernmental Panel on Climate Change (IPCC) (2021). Summary for Policymakers. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, A.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T. Yelekçi, O., Yu, R. and Zhou, B. (eds.). Cambridge: Cambridge University Press. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Full_Report.pdf

James, S.R., Dennell, R.W., Gilbert, A.S., Lewis, H.T., Gowlett, J.A.J., Lynch, T.F. et al. (1989). Hominid use of fire in the Lower and Middle Pleistocene: a review of the evidence. Current Anthropology, 30(1), 1-26. https://www.jstor.org/stable/2743299.

Jones, G.M., Kramer, H.A., Whitmore, S.A., Berigan, W.J., Tempel, D.J., Wood, C.M. et al. (2020). Habitat selection by spotted owls after a megafire reflects their adaptation to historical frequent-fire regimes. Landscape Ecology, 35(5), 1199-1213. https://doi.org/10.1007/s10980-020-01010-y.

Keeley, J.E. (2008). Fire. In Encyclopedia of Ecology. Jørgensen, S.E. and Fath, B.D. (eds.). Amsterdam: Elsevier B.V. 1557-1564. https://doi.org/10.1016/B978-008045405-4.00496-1.

Keeley, J.E. (2009). Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire, 18(1), 116-126. https://doi.org/10.1071/WF07049.

Kelley, D.I., Bistinas, I., Whitley, R., Burton, C., Marthews, T.R. and Dong, N. (2019). How contemporary bioclimatic and human controls change global fire regimes. Nature Climate Change, 9(9), 690-696. https://doi.org/10.1038/s41558-019-0540-7.

Lévi-Strauss, C. (1969). The Raw and the Cooked. New York, NY: Harper and Row.

National Aeronautics and Space Administration (2020). Active Fire Data. https://firms.modaps.eosdis.nasa.gov/active_fire/. Accessed 9 June 2021.

Oakley, K. (1956). Fire as a Palaeolithic tool and weapon. Proceedings of the Prehistoric Society, 21, 36-48. https://doi.org/10.1017/S0079497X00017382.

Pickett, S. and White, P. (eds.) (1985). The Ecology of Natural Disturbance and Patch Dynamics. San Diego, CA: Academic Press.

Roebroeks, W. and Villa, P. (2011). On the earliest evidence for habitual use of fire in Europe. Proceedings of the National Academy of Sciences of the United States of America, 108(13), 5209-5214. https://doi.org/10.1073/pnas.1018116108.

Roy, D.P., Boschetti, L. and Trigg, S.N. (2006). Remote sensing of fire severity: assessing the performance of the normalized burn ratio. IEEE Geoscience and Remote Sensing Letters, 3(1), 112-116. https://doi.org/10.1109/LGRS.2005.858485.

Rykiel, E.J. Jr. (1985). Towards a definition of ecological disturbance. Australian Journal of Ecology, 10(3), 361-365. https://doi.org/10.1111/j.1442-9993.1985.tb00897.x.

Sullivan, A.L. (2017). Inside the inferno: fundamental processes of wildland fire behaviour. Part 1: combustion chemistry and heat release. Current Forestry Reports, 3, 132-149. https://doi.org/10.1007/s40725-017-0057-0.

Sullivan, A.L., Surawski, N.C., Crawford, D., Hurley, R.J., Volkova, L. and Weston, C.J. (2018). Effect of woody debris on the rate of spread of surface fires in forest fuels in a combustion wind tunnel. Forest Ecology and Management, 424, 236-245. https://doi.org/10.1016/j.foreco.2018.04.039.

Surawski, N.C., Sullivan, A.L., Roxburgh, S.H. and Polglase, P.J. (2016). Estimates of greenhouse gas and black carbon emissions from a major Australian wildfire with high spatiotemporal resolution. Journal of Geophysical Research: Atmospheres, 121(16), 9892-9907. https://doi.org/10.1002/2016JD025087.

Van Wagtendonk, J.W., Sugihara, N.G., Stephens, S.L., Thode, A.E., Shaffer, K.E. and Fites-Kaufman, J. (eds.) (2018). Fire in California's Ecosystems. Second edition. Berkeley, CA: University of California Press.

Voosen, P. (2021). Global temperatures in 2020 tied record highs. Science, 371(6527), 334-335. https://doi.org/10.1126/science.371.6527.334.

Wooster, M.J., Roberts, G., Perry, G.L.W. and Kaufman, Y.J. (2005). Retrieval of biomass combustion rates and totals from fire radiative power observations: FRP derivation and calibration relationships between biomass consumption and fire radiative energy release. Journal of Geophysical Research: Atmospheres, 110, D24311. https://doi.org/10.1029/2005JD006318.

Wright, H.E. and Heinselman, M.L. (2014). The ecological role of fire in natural conifer forests of western and northern North America: introduction. Fire Ecology, 10(3), 4-13. https://doi.org/10.1007/BF03400628.

Chapter 2

Abatzoglou, J.T., Battisti, D.S., Williams, A.P., Hansen, W.D., Harvey, B.J. and Kolden, C.A. (2021). Projected increases in western US forest fire despite growing fuel constraints. Communications Earth & Environment, 2(1), pp.1-8. https://doi.org/10.1038/s43247-021-00299-0.

Abram, N.J., Henley, B.J., Gupta, A.S., Lippmann, T.J., Clarke, H., Dowdy, A.J., Sharples, J.J., Nolan, R.H., Zhang, T., Wooster, M.J. and Wurtzel, J.B. (2021). Connections of climate change and variability to large and extreme forest fires in southeast Australia. Communications Earth & Environment, 2(1), pp.1-17. https://doi.org/10.1038/s43247-020-00065-8.

Adams, M.A., Shadmanroodposhti, M. and Neumann, M. (2020). Causes and consequences of Eastern Australia's 2019-20 season of mega-fires: a broader perspective. Global Change Biology, 26(7), 3756-3758. https://doi.org/10.1111/gcb.15125.

Aleman, J.C. and Fayolle, A. (2020). Long-term vegetation change in Central Africa: the need for an integrated management framework for forests and savannas. In Sustainability Challenges in Sub-Saharan Africa I: Continental Perspectives and Insights from Western and Central Africa. Science for Sustainable Societies Series. Gasparatos, A., Ahmed, A., Naidoo, M., Karanja, A., Fukushi, K., Saito, O. et al. (eds.). Singapore: Springer. Chapter 9. 281-315. https://doi.org/10.1007/978-981-15-4458-3_9.

Alvarado, S.T., Andela, N., Silva, T.S.F. and Archibald, S. (2019). Thresholds of fire response to moisture and fuel load differ between tropical savannas and grasslands across continents. Global Ecology and Biogeography, 29(2), 331-344. https://doi.org/10.1111/geb.13034.

Andela, N., Morton, D.C., Giglio, L., Chen, Y., Van der Werf, G.R. and Kasibhatla, P.S. (2017). A human-driven decline in global burned area. Science, 356(6345), 1356-1362. https://doi.org/10.1126/science.aal4108.

Aragão, L.E.O.C., Anderson, L.O., Fonseca, M.G., Rosan, T.M., Vedovato, L.B., Wagner, F.H. et al. (2018). 21st century drought-related fires counteract the decline of Amazon deforestation carbon emissions. Nature Communications, 9(1), 536. https://doi.org/10.1038/s41467-017-02771-y.

Aragão, L.E.O.C., Barbier, Y.M.N., Lima, A., Shimabukuro, Y., Anderson, L. and Saatchi, S. 2008. Interactions between rainfall, deforestation and fires during recent years in the Brazilian Amazonia. Philosophical Transactions of the Royal Society B: Biological Sciences. 363(1498): 1779-1785.

Archibald, S., Lehmann, C.E.R., Belcher, C.M., Bond, W.J., Bradstock, R.A., Daniau, A.L. et al. (2018). Biological and geophysical feedbacks with fire in the Earth system. Environmental Research Letters, 13(3), 033003. https://doi.org/10.1088/1748-9326/aa9ead.

Archibald, S., Lehmann, C.E.R, Gómez-Dans, J.L. and Bradstock, R.A. (2013). Defining pyromes and global syndromes of fire regimes. Proceedings of the National Academy of Sciences of the United States of America, 110(16), 6442-6447. https://doi.org/10.1073/pnas.1211466110.

Archibald, S., Roy, D.P., Van Wilgen, B.W. and Scholes, R.J. (2009). What limits fire? An examination of drivers of burnt area in Southern Africa. Global Change Biology, 15(3), 613-630. https://doi.org/10.1111/j.1365-2486.2008.01754.x.

Armenteras, D., Barreto, J.S., Tabor, K., Molowny-Horas, R. and Retana, J. (2017). Changing patterns of fire occurrence in proximity to forest edges, roads and rivers between NW Amazonian countries. Biogeosciences, 14(11), 2755-2765. https://doi.org/10.5194/bg-14-2755-2017.

Armenteras, D., González, T.M. and Retana, J. (2013). Forest fragmentation and edge influence on fire occurrence and intensity under different management types in Amazon forests. Biological Conservation, 159, 73-79. https://doi.org/10.1016/j.biocon.2012.10.026.

Australia, Geoscience Australia (n.d.). NationalMap. http://nationalmap.gov.au. Accessed 15 October 2021.

Bagley, J.E., Desai, A.R., Harding, K.J., Snyder, P.K. and Foley, J.A. (2014). Drought and deforestation: has land cover change influenced recent precipitation extremes in the Amazon? Journal of Climate, 27(1), 345-361. https://doi.org/10.1175/JCLI-D-12-00369.1.

Balch, J.K., Nepstad, D.C. and Curran, L.M. (2009). Pattern and process: fire-initiated grass invasion at Amazon transitional forest edges. In Tropical Fire Ecology. Cochrane, M.A. (ed.). Berlin, Heidelberg: Springer. 481-502. https://doi.org/10.1007/978-3-540-77381-8_17.

Beckett, H. (2018). Firestorms in a Mesic Savanna-Forest Mosaic. Doctoral Thesis. Cape Town: University of Cape Town, Faculty of Science. https://core.ac.uk/download/pdf/200257801.pdf.

Berenguer, E., Ferreira, J., Gardner, T.A., Aragão, L.E.O.C., De Camargo, P.B., Cerri, C.E. et al. (2014). A large-scale field assessment of carbon stocks in human-modified tropical forests. Global Change Biology, 20(12), 3713-3726. https://doi.org/10.1111/gcb.12627.

Bieniek, P.A., Bhatt, U.S., York, A., Walsh, J.E., Lader, R., Strader, H. et al. (2020). Lightning variability in dynamically downscaled simulations of Alaska's present and future summer climate. Journal of Applied Meteorology and Climatology, 59(6), 1139-1152. https://doi.org/10.1175/JAMC-D-19-0209.1.

Bistinas, I., Harrison, S.P., Prentice, I.C. and Pereira, J.M.C. (2014). Causal relationships versus emergent patterns in the global controls of fire frequency. Biogeosciences, 11(18), 5087-5101. https://doi.org/10.5194/bg-11-5087-2014.

Boehm, H.D.V. and Siegert, F. (2001). Ecological impact of the one million hectare rice project in Central Kalimantan, Indonesia, using remote sensing and GIS. Paper presented at the 22nd Asian Conference on Remote Sensing. Singapore. November.

Booth, T.H., Kirschbaum, M.U.F. and Battaglia, M. (2008). Forestry. In An Overview of Climate Change Adaptation in the Australian Agricultural Sector: Impacts, Options and Priorities. Stokes, C.J. and Howden, S.M. (eds.). Canberra: Commonwealth Scientific and Industrial Research Organisation. Chapter 8. 207-228. http://www.csiro.au/resources/AgricultureAdaptationReport2008.html.

Bowman, D.M.J.S., Balch, J.K., Artaxo, P., Bond, W.J., Carlson, J.M., Cochrane, M.A. et al. (2009). Fire in the Earth system. Science, 324(5926), 481-484. https://doi.org/10.1126/science.1163886.

Bowman, D.M.J.S., Balch, J.K., Artaxo, P., Bond, W.J., Cochrane, M.A., D'Antonio, C.M. et al. (2011). The human dimension of fire regimes on Earth. Journal of Biogeography, 38(12), 2223-2236. https://doi.org/10.1111/j.1365-2699.2011.02595.x.

Bowman, D.M.J.S., Kolden, C.A., Abatzoglou, J.T., Johnston, F.H., Van der Werf, G.R. and Flannigan, M. (2020a). Vegetation fires in the Anthropocene. Nature Reviews Earth and Environment, 1(10), 500-515. https://doi.org/10.1038/s43017-020-0085-3.

Bowman, D.M.J.S., Moreira-Muñoz, A., Kolden, C.A., Chávez, R.O., Muñoz, A.A., Salinas, F. et al. (2019). Human–environmental drivers and impacts of the globally extreme 2017 Chilean fires. Ambio, 48(4), 350-362. https://dx.doi.org/10.1007%2Fs13280-018-1084-1.

Bowman, D.M.J.S., Williamson, G., Yebra, M., Lizundia-Loiola, J., Pettinari, M.L., Shah, S. et al. (2020b). Wildfires: Australia needs national monitoring agency. Nature, 584, 188-191. https://doi.org/10.1038/d41586-020-02306-4.

Bradstock, R.A., Williams, J.E. and Gill, A.M. (eds.) (2002). Flammable Australia: The Fire Regimes and Biodiversity of a Continent. Cambridge: Cambridge University Press.

Brando, P.M., Balch, J.K., Nepstad, D.C., Morton, D.C., Putz, F.E., Coe, M.T. et al. (2014). Abrupt increases in Amazonian tree mortality due to drought-fire interactions. Proceedings of the National Academy of Sciences of the United States of America, 111(17), 6347-6352. https://doi.org/10.1073/pnas.1305499111.

Canadell, J.G., Meyer, C.P., Cook, G.D., Dowdy, A., Briggs, P.R., Knauer, J., Pepler, A. and Haverd, V. (2021). Multi-decadal increase of forest burned area in Australia is linked to climate change. Nature Communications, 12(1), pp.1-11. https://doi.org/10.1038/s41467-021-27225-4.

Castello, L. and Macedo, M.N. (2016). Large-scale degradation of Amazonian freshwater ecosystems. Global Change Biology, 22(3), 990-1007. https://doi.org/10.1111/gcb.13173.

Chagas, T., Galt, H., Lee, D., Neeff, T. and Streck, C. (2020). A Close Look at the

Quality of REDD+ Carbon Credits. https://www.climatefocus.com/publications/close-look-quality-redd-carbon-credits.

Chen, Y., Morton, D.C., Andela, N., Van der Werf, G.R., Giglio, L. and Randerson, J.T. (2017). A pan-tropical cascade of fire driven by El Niño/Southern Oscillation. Nature Climate Change, 7(12), 906-911. https://doi.org/10.1038/s41558-017-0014-8.

Chipman, M.L. and Hu, F.S. (2017). Linkages among climate, fire, and thermoerosion in Alaskan tundra over the past three millennia. Journal of Geophysical Research: Biogeosciences, 122(12), 3362-3377. https://doi.org/10.1002/2017JG004027.

Ciavarella, A., Cotterill, D., Stott, P., Kew, S., Philip, S., Van Oldenborgh, G.J. et al. (2020). Prolonged Siberian

Heat of 2020. World Weather Attribution. https://www.worldweatherattribution.org/wp-content/uploads/WWA-Prolonged-heat-Siberia-2020.pdf.

Cochrane, M.A. and Laurance, W.F. (2008). Synergisms among fire, land use, and climate change in the Amazon. AMBIO: A Journal of the Human Environment, 37(7), 522-527. https://doi.org/10.1579/0044-7447-37.7.522.

Coops, N.C., Hermosilla, T., Wulder, M.A., White, J.C. and Bolton, D.K. (2018). A thirty year, fine-scale, characterization of area burned in Canadian forests shows evidence of regionally increasing trends in the last decade. PLoS One, 13(5), p.e0197218. https://doi.org/10.1371/journal.pone.0197218.

Crippa, P., Castruccio, S., Archer-Nicholls, S., Lebron, G.B., Kuwata, M., Thota, A. et al. (2016). Population exposure to hazardous air quality due to the 2015 fires in Equatorial Asia. Scientific Reports, 6(1), 37074. https://doi.org/10.1038/srep37074.

Crump, J. (ed.) (2017). Smoke on Water: Countering Global Threats from Peatland Loss and Degradation. Nairobi and Arendal: United Nations Environment Programme and GRID-Arendal. https://www.grida.no/publications/355.

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., Ayres, M.P., Flannigan, M.D. et al. (2001). Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience, 51(9), 723-734. https://doi.org/10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2.

Daniau, A.L., Sánchez Goñi, M.F., Martinez, P., Urrego, D.H., Bout-Roumazeilles, V., Desprat, S. et al. (2013). Orbital-scale climate forcing of grassland burning in southern Africa. Proceedings of the National Academy of Sciences of the United States of America, 110(13), 5069-5073. https://doi.org/10.1073/pnas.1214292110.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. et al. (2017). Age, extent and carbon storage of the central Congo Basin peatland complex. Nature, 542(7639), 86-90. https://doi.org/10.1038/nature21048.

Davidson, P.J.A. (ed.) (2006). The Biodiversity of the Tonle Sap Biosphere: 2005 Status Review. Phnom Penh: Wildlife Conservation Society. https://library.wcs.org/doi/ctl/view/mid/33065/pubid/DMX1253100000.aspx.

Duchelle, A.E., Seymour, F., Brockhaus, M., Angelsen, A., Larson, A.M., Moeliono, M. et al. (2018). REDD+: Lessons from National and Subnational Implementation. Ending Tropical Deforestation: A Stock-Take of Progress and Challenges Working Paper Series. Washington, DC: World Resources Institute. https://helda.helsinki.fi//bitstream/handle/10138/299239/ending_tropical_deforestation_redd_lessons_implementation.pdf?sequence=1.

Engelbrecht, F., Adegoke, J., Bopape, M.J., Naidoo, M., Garland, R., Thatcher, M. et al. (2015). Projections of rapidly rising surface temperatures over Africa under low mitigation. Environmental Research Letters, 10(8), 085004. https://doi.org/10.1088/1748-9326/10/8/085004.

Estop-Aragonés, C., Czimczik, C.I., Heffernan, L., Gibson, C., Walker, J.C., Xu, X. et al. (2018). Respiration of aged soil carbon during fall in permafrost peatlands enhanced by active layer deepening following wildfire but limited following thermokarst. Environmental Research Letters, 13(8), 85002. https://doi.org/10.1088/1748-9326/aad5f0.

Fasullo, J.T., Otto-Bliesner, B.L. and Stevenson, S. (2018). ENSO's changing influence on temperature, precipitation, and wildfire in a warming climate. Geophysical Research Letters, 45(17), 9216-9225. https://doi.org/10.1029/2018GL079022.

Ferreira, G.B., Collen, B., Newbold, T., Oliveira, M.J.R., Pinheiro, M.S., de Pinho, F.F. et al. (2020). Strict protected areas are essential for the conservation of larger and threatened mammals in a priority region of the Brazilian Cerrado. Biological Conservation, 251, 108762. https://doi.org/10.1016/j.biocon.2020.108762.

Field, R.D., Van der Werf, G.R., Fanin, T., Fetzer, E.J., Fuller, R., Jethva, H. et al. (2016). Indonesian fire activity and smoke pollution in 2015 show persistent nonlinear sensitivity to El Niño-induced drought. Proceedings of the National Academy of Sciences of the United States of America, 113(33), 9204-9209. https://doi.org/10.1073/pnas.1524888113.

Filkov, A.I., Ngo, T., Matthews, S., Telfer, S. and Penman, T.D. (2020). Impact of Australia's catastrophic 2019/20 bushfire season on communities and environment. Retrospective analysis and current trends. Journal of Safety Science and Resilience, 1(1), 44-56. https://doi.org/10.1016/j.jnlssr.2020.06.009.

Flannigan, M.D., Wotton, B.M., Marshall, G.A., de Groot, W.J., Johnston, J., Jurko, N. et al. (2016). Fuel moisture sensitivity to temperature and precipitation: climate change implications. Climatic Change, 134(1-2), 59-71. https://doi.org/10.1007/s10584-015-1521-0.

Forest Carbon Partnership Facility (2021). 2020 Annual Report. Washington, DC: World Bank. https://www.forestcarbonpartnership.org/system/files/documents/FCPF%202020%20Annual%20Report_Web_update.pdf.

Forest Trends' Ecosystem Marketplace (2021). A Green Growth Spurt: State of Forest Carbon Finance 2021. Washington, DC: Forests Trends Association.

Forkel, M., Dorigo, W., Lasslop, G., Chuvieco, E., Hantson, S., Heil, A. et al. (2019). Recent global and regional trends in burned area and their compensating environmental controls. Environmental Research Communications 1(5), 051005. https://doi.org/10.1088/2515-7620/ab25d2.

French, N.H.F., Jenkins, L.K., Loboda, T.V., Flannigan, M., Jandt, R., Bourgeau-Chavez L.L. et al. (2015). Fire in Arctic tundra of Alaska: past fire activity, future fire potential, and significance for land management and ecology. International Journal of Wildland Fire, 24(8), 1045-1061. https://doi.org/10.1071/WF14167.

Ganz, D. (2020). Coronavirus, climate change, forest fires: integrated fire management and people-centred policymaking, 30 April. www.recoftc.org/stories/coronavirus-climate-change-forest-fires-integrated-fire-management-and-people-centred. Accessed 5 October 2021.

Goldstein, J. (2016). Carbon bomb: Indonesia's failed mega rice project. Arcadia: Explorations in Environmental History, 6. https://doi.org/10.5282/rcc/7474.

Goldstein, A., Turner, W.R., Spawn, S.A., Anderson-Teixeira, K.J., Cook-Patton, S., Fargione, J. et al. (2020). Protecting irrecoverable carbon in Earth's ecosystems. Nature Climate Change, 10(4), 287-295. https://doi.org/10.1038/s41558-020-0738-8.

Geirinhas, J.L., Trigo, R.M., Libonati, R., Coelho, C.A. and Palmeira, A.C. (2018). Climatic and synoptic characterization of heat waves in Brazil. International Journal of Climatology, 38(4), 1760-1776. https://doi.org/10.1002/joc.5294.

Gibson, C.M., Chasmer, L.E., Thompson, D.K., Quinton, W.L., Flannigan, M.D. and Olefeldt, D. (2018). Wildfire as a major driver of recent permafrost thaw in boreal peatlands. Nature Communications, 9, 3041. https://doi.org/10.1038/s41467-018-05457-1.

Gill, A.M. (1975). Fire and the Australian flora: a review. Australian Forestry, 38(1), 4-25. https://doi.org/10.1080/00049158.1975.10675618.

Glauber, A.J., Moyer, S., Adriani, M. and Iwan, G. (2016). The Cost of Fire: An Economic Analysis of Indonesia's 2015 Fire Crisis. Indonesia Sustainable Landscapes Knowledge Note No. 1. Jakarta: World Bank. https://openknowledge.worldbank.org/handle/10986/23840.

Goss, M., Swain, D.L., Abatzoglou, J.T., Sarhadi, A., Kolden, C.A., Williams, A.P. et al. (2020). Climate change is increasing the likelihood of extreme autumn wildfire conditions across California. Environmental Research Letters, 15(9), 094016. https://doi.org/10.1088/1748-9326/ab83a7.

Hanes, C.C., Wang, X., Jain, P., Parisien, M-A., Little, J.M. and Flannigan, M.D. (2019). Fire-regime changes in Canada over the last half century. Canadian Journal of Forest Research, 49(3), 256-269. https://doi.org/10.1139/cjfr-2018-0293.

Hantson, S., Arneth, A., Harrison, S.P., Kelley, D.I., Prentice, I.C., Rabin, S.S. et al. (2016). The status and challenge of global fire modelling. Biogeosciences,13(11), 3359-75. https://doi.org/10.5194/bg-13-3359-2016

Hantson, S., Kelley, D.I., Arneth, A., Harrison, S.P., Archibald, S., Bachelet, D. et al. (2020). Quantitative assessment of fire and vegetation properties in simulations with fire-enabled vegetation models from the Fire Model Intercomparison Project. Geoscientific Model Development 13(7), 3299-318. https://doi.org/10.5194/gmd-13-3299-2020

Higgins, S.I. and Scheiter, S. (2012). Atmospheric CO2 forces abrupt vegetation shifts locally, but not globally. Nature, 488(7410), 209-212. https://doi.org/10.1038/nature11238.

Higuera, P.E., Brubaker, L.B., Anderson, P.M., Hu, F.S. and Brown, T.A. (2009). Vegetation mediated the impacts of postglacial climate change on fire regimes in the south-central Brooks Range, Alaska. Ecological Monographs, 79(2), 201-219. https://doi.org/10.1890/07-2019.1.

Higuera, P.E., Chipman, M.L., Barnes, J.L., Urban, M.A. and Hu, F.S. (2011). Variability of tundra fire regimes in Arctic Alaska: millennial-scale patterns and ecological implications. Ecological Applications, 21(8), 3211-3226. https://www.jstor.org/stable/41417122.

Hoffman, M.T., Rohde, R.F. and Gillson, L. (2019). Rethinking catastrophe? Historical trajectories and modelled future vegetation change in southern Africa. Anthropocene, 25, 100189. https://doi.org/10.1016/j.ancene.2018.12.003.

Hoffman, W.A., Schroeder, W. and Jackson, R.B. (2003). Regional feedbacks among fire, climate, and tropical deforestation. Journal of Geophysical Research: Atmospheres, 108(D23), 4721. https://doi.org/10.1029/2003JD003494.

Holloway, J.E., Lewkowicz, A.G., Douglas, T.A., Li, X., Turetsky, M.R., Baltzer, J.L. et al. (2020). Impact of wildfire on permafrost landscapes: a review of recent advances and future prospects. Permafrost and Periglacial Processes, 31(3), 371-382. https://doi.org/10.1002/ppp.2048.

Hsiang, S., Kopp, R., Jina, A., Rising, J., Delgado, M., Mohan, S., Rasmussen, D.J., Muir-Wood, R., Wilson, P., Oppenheimer, M. and Larsen, K. (2017). Estimating economic damage from climate change in the United States. Science, 356(6345), pp.1362-1369. https://doi.org/10.1126/science.aal4369.

Hu, F.S., Higuera, P.E., Duffy, P., Chipman, M.L., Rocha, A.V., Young, A.M. et al. (2015). Arctic tundra fires: natural variability and responses to climate change. Frontiers in Ecology and the Environment, 13(7), 369-377. https://doi.org/10.1890/150063.

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R.B., Jones, M., MacDonald, G. et al. (2020). Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. Proceedings of the National Academy of Sciences of the United States of America, 17(34), 20438-20446. https://doi.org/10.1073/pnas.1916387117.

Humphrey, G.J., Gillson, L. and Ziervogel, G. (2020). How changing fire management policies affect fire seasonality and livelihoods. Ambio, 50(2), 475-491. https://doi.org/10.1007/s13280-020-01351-7.

Intergovernmental Panel on Climate Change. (2013). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M.M.B., Allen, S.K., Boschung, J., Nauels, A, Xia, Y., Bex, V. and Midgley, P.M. (eds.) Cambridge: Cambridge University Press. https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_all_final.pdf.

Intergovernmental Panel on Climate Change. (2018). Global Warming of 1.5°C: An IPCC Special Report on the Impacts of Global Warming of 1.5°C Above Pre-Industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty. Masson-Delmotte, V., Zhai, P., Pörtner, H.-O., Roberts, D., Skea, J., Shukla, P.R., Pirani, A., Moufouma-Okia, W., Péan, C., Pidcock, R., Connors, S., Matthews, J.B.R., Chen, Y., Zhou, X., Gomis, M.I., Lonnoy, E., Maycock, T., Tignor, M. and Waterfield, T. (eds.) https://www.ipcc.ch/site/assets/uploads/sites/2/2019/06/SR15_Full_Report_High_Res.pdf.

Intergovernmental Panel on Climate Change (IPCC) (2021). Summary for Policymakers. In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, A.L., Péan, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J.B.R., Maycock, T.K., Waterfield, T. Yelekçi, O., Yu, R. and Zhou, B. (eds.). Cambridge: Cambridge University Press. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Full_Report.pdf

Jimenez, J.C., Libonati, R. and Peres, L.F. (2018). Droughts over Amazonia in 2005, 2010, and 2015: a cloud cover perspective. Frontiers in Earth Science, 6. https://doi.org/10.3389/feart.2018.00227.

Jeffery, K.J., Korte, L., Palla, F., Walters, G.M., White, L.J.T. and Abernethy, K.A. (2014). Fire management in a changing landscape: a case study from Lopé National Park, Gabon. PARKS: The International Journal of Protected Areas and Conservation, 20(1), 39-52. https://doi.org/10.2305/IUCN.CH.2014.PARKS-20-1.KJJ.en.

Johnston, F.H., Borchers-Arriagada, N., Morgan, G.G., Jalaudin, B., Palmer, A.J., Williamson, G.J. et al. (2021). Unprecedented health costs of smoke-related PM2.5 from the 2019–20 Australian megafires. Nature Sustainability, 4(1), 42-47. https://doi.org/10.1038/s41893-020-00610-5.

Jones, B.M., Grosse, G., Arp, C.D., Miller, E., Liu, L., Hayes, D.J. et al. (2015). Recent Arctic tundra fire initiates widespread thermokarst development. Scientific Reports, 5, 15865. https://doi.org/10.1038/srep15865.

Joosten, H. (2015). Peatlands, Climate Change Mitigation and Biodiversity Conservation: An Issue Brief on the Importance of Peatlands for Carbon and Biodiversity Conservation and the Role of Drained Peatlands as Greenhouse Gas Emission Hotspots. Copenhagen: Nordic Council of Ministers. http://dx.doi.org/10.6027/ANP2015-727.

Kauppi, P., Hanewinkel, M., Lundmark, T., Nabuurs, G.J., Peltola, H., Trasobares, A. et al. (2018). Climate smart forestry in Europe. European Forest Institute. https://efi.int/sites/default/files/files/publication-bank/2018/Climate_Smart_Forestry_in_Europe.pdf.

Keegan, K.M., Albert, M.R., McConnell, J.R. and Baker, I. (2014). Climate change and forest fires synergistically drive widespread melt events of the Greenland Ice Sheet. Proceedings of the National Academy of Sciences of the United States of America, 111(22), 7964-7967. https://doi.org/10.1073/pnas.1405397111.

Kelley, D.I., Bistinas, I., Whitley, R., Burton, C., Marthews, T.R. and Dong, N. (2019). How contemporary bioclimatic and human controls change global fire regimes. Nature Climate Change, 9(9), 690-696. https://doi.org/10.1038/s41558-019-0540-7.

Kelley, D.I., Burton, C., Huntingford, C., Brown, M.A., Whitley, R. and Dong, N. (2021). Low meteorological influence found in 2019 Amazonia fires. Biogeosciences, 18(3), 787-804. https://doi.org/10.5194/bg-18-787-2021.

Kelley, D.I., Mathison, C., Burton, C., Brown, M., Sullivan, A., Baker, E. and Kurvits, T. (2021) Likely future(s) of global wildfires. UK Earth System Modelling News. December 2021. https://ukesm.ac.uk/portfolio-item/likely-futures-of-global-wildfires/

Kelly, R., Chipman, M.L., Higuera, P.E., Stefanova, I., Brubaker, L.B. and Hu, F.S. (2013). Recent burning of boreal forests exceeds fire regime limits of the past 10,000 years. Proceedings of the National Academy of Sciences of the United States of America, 110(32), 13055-13060. https://dx.doi.org/10.1073%2Fpnas.1305069110.

Kharuk, V.I., Dvinskaya, M.L., Petrov, I.A., Im, S.T. and Ranson, K.J. (2016). Larch forests of Middle Siberia: long-term trends in fire return intervals. Regional Environmental Change, 16(8), 2389-2397. https://dx.doi.org/10.1007%2Fs10113-016-0964-9.

Kharuk, V.I., Ponomarev, E.I., Ivanova, G.A., Dvinskaya, M.L., Coogan, S.C. and Flannigan, M.D. (2021). Wildfires in the Siberian taiga. Ambio, pp.1-22. https://doi.org/10.1007/s13280-020-01490-x.

King, A.D., Pitman, A.J., Henley, B.J., Ukkola, A.M. and Brown, J.R. (2020). The role of climate variability in Australian drought. Nature Climate Change, 10(3), 177-179. https://doi.org/10.1038/s41558-020-0718-z.

Kirchmeier-Young, M.C., Zwiers, F.W., Gillett, N.P. and Cannon, A.J. (2017). Attributing extreme fire risk in Western Canada to human emissions. Climatic Change, 144(2), 365-379. https://doi.org/10.1007/s10584-017-2030-0.

Kloster, S. and Lasslop, G. (2017) Historical and future fire occurrence (1850 to 2100) simulated in CMIP5 Earth System Models. Global and Planetary Change. 150, 58-69. https://doi.org/10.1016/j.gloplacha.2016.12.017

Knorr, W., Kaminski, T., Arneth A. and Weber, U. (2014). Impact of human population density on fire frequency at the global scale. Biogeosciences, 11(4), 1085-1102. https://doi.org/10.5194/bg-11-1085-2014.

Koplitz, S.N., Mickley, L.J., Marlier, M.E., Buonocore, J.J., Kim, P.S., Liu, T. et al. (2016). Public health impacts of the severe haze in Equatorial Asia in September–October 2015: demonstration of a new framework for informing fire management strategies to reduce downwind smoke exposure. Environmental. Research Letters, 11(9), 094023. https://doi.org/10.1088/1748-9326/11/9/094023.

Kraaij, T., Baard, J.A., Arndt, J., Vhengani, L. and Van Wilgen, B.W. (2018). An assessment of climate, weather, and fuel factors influencing a large, destructive wildfire in the Knysna region, South Africa. Fire Ecology, 14(2), 4. https://doi.org/10.1186/s42408-018-0001-0.

Krawchuk, M.A. and Moritz, M.A. (2011). Constraints on global fire activity vary across a resource gradient. Ecology, 92(1), 121-132. https://doi.org/10.1890/09-1843.1.

Langenbrunner, B., Neelin, J.D., Lintner, B.R. and Anderson, B.T. (2015). Patterns of precipitation change and climatological uncertainty among CMIP5 models, with a focus on the midlatitude Pacific storm track. Journal of Climate, 28(19), 7857-7872. https://doi.org/10.1175/JCLI-D-14-00800.1.

Laris, P. (2002). Burning the seasonal mosaic: preventative burning strategies in the wooded savanna of southern Mali. Human Ecology, 30(2), 155-186. https://doi.org/10.1023/A:1015685529180.

Laris, P., Caillault, S., Dadashi, S. and Jo, A. (2015). The human ecology and geography of burning in an unstable savanna environment. Journal of Ethnobiology, 35(1), 111-139. https://doi.org/10.2993/0278-0771-35.1.111.

Lasslop, G., Hantson, S., Harrison, S.P., Bachelet, D., Burton, C., Forkel, M. et al. (2020). Global ecosystems and fire: multi-model assessment of fire-induced tree-cover and carbon storage reduction. Global Change Biology, 26(9), 5027-5041. https://doi.org/10.1111/gcb.15160.

Libonati, R., DaCamara, C. C., Setzer, A. W., Morelli, F. and Melchiori, A. E. (2015). An algorithm for burned area detection in the Brazilian Cerrado using 4 μm MODIS imagery. Remote sensing. 7(11), 15782-15803. https://doi.org/10.3390/rs71115782

Libonati, R., Pereira, J.M.C., DaCamara, C.C., Peres, L.F., Oom, D., Rodrigues, J.A. et al. (2021). Twenty-first century droughts have not increasingly exacerbated fire season severity in the Brazilian Amazon. Scientific Reports, 11, 4400. https://doi.org/10.1038/s41598-021-82158-8.

Littell, J.S., Peterson, D.L., Riley, K.L., Liu, Y. and Luce, C.H. (2016). A review of the relationships between drought and forest fire in the United States. Global Change Biology, 22(7), 2353-2369. https://doi.org/10.1111/gcb.13275.

Liu, J.C., Wilson, A., Mickley, L.J., Dominici, F., Ebisu, K., Wang, Y., Sulprizio, M.P., Peng, R.D., Yue, X., Son, J.Y. and Anderson, G.B. (2017). Wildfire-specific fine particulate matter and risk of hospital admissions in urban and rural counties. Epidemiology (Cambridge, Mass.), 28(1), p.77. https://doi.org/10.1097/EDE.0000000000000556.

Lohberger, S., Stängel, M., Atwood, E.C. and Siegert, F. (2018). Spatial evaluation of Indonesia's 2015 fire-affected area and estimated carbon emissions using Sentinel-1. Global Change Biology, 24(2), 644-654. https://doi.org/10.1111/gcb.13841.

Lowering Emissions by Accelerating Forest Finance Commission (2021). Homepage. https://leafcoalition.org/. Accessed 22 September 2021.

Luke, R.H. and McArthur, A.G. (1978). Bushfires in Australia. Canberra: Australian Government Publishing Service.

Machado-Silva, F., Peres, L.F., Gouveia, C.M., Enrich-Prast, A., Peixoto, R.B., José M. C. et al. (2011). Drought Resilience Debt Drives NPP Decline in the Amazon Forest. Global Biogeochemical Cycles. 35 (9), e2021GB007004. https://doi.org/10.1029/2021GB007004

Mack, M.C., Bret-Harte, M.S., Hollingsworth, T.N., Jandt, R.R., Schuur, E.A.G., Shaver, G.R., et al. (2011). Carbon loss from an unprecedented Arctic tundra wildfire. Nature, 475, 489-492. https://doi.org/10.1038/nature10283.

Mahood, S.P., Poole, C.M., Watson, J.E.M., MacKenzie, R.A., Sharma, S. and Garnett, S.T. (2020). Agricultural intensification is causing rapid habitat change in the Tonle Sap Floodplain, Cambodia. Wetlands Ecology and Management, 28(5), 713-726. https://doi.org/10.1007/s11273-020-09740-1.

Marlon, J.R., Bartlein, P.J., Carcaillet, C., Gavin, D.G., Harrison, S.P., Higuera, P.E. et al. (2008). Climate and human influences on global biomass burning over the past two millennia. Nature Geoscience, 1(10), 697-702. https://doi.org/10.1038/ngeo313.

Masrur, A., Petrov, A.N. and DeGroote, J. (2018). Circumpolar spatio-temporal patterns and contributing climatic factors of wildfire activity in the Arctic tundra from 2001–2015. Environmental Research Letters, 13(1), 014019. https://doi.org/10.1088/1748-9326/aa9a76.

Matthews, S., Sullivan, A.L., Watson, P. and Williams, R.J. (2012). Climate change, fuel, and fire behaviour in a eucalypt forest. Global Change Biology 18(10), 3212-3223. https://doi.org/10.1111/j.1365-2486.2012.02768.x.

McCarty, J.L., Smith, T.E.L. and Turetsky, M.R. (2020). Arctic fires re-emerging. Nature Geoscience, 13(10), 658-660. https://doi.org/10.1038/s41561-020-00645-5.

McKenzie, D. and Littell, J.S. (2017). Climate change and the eco-hydrology of fire: will area burned increase in a warming western USA? Ecological Applications, 27(1), 26-36. https://doi.org/10.1002/eap.1420.

Moncrieff, G.R., Scheiter, S., Slingsby, J.A. and Higgins, S.I. (2015). Understanding global change impacts on South African biomes using Dynamic Vegetation Models. South African Journal of Botany, 101, 16-23. https://doi.org/10.1016/j.sajb.2015.02.004.

Moreira, F., Ascoli, D., Safford, H.D., Adams, M.A., Moreno, J.M., Pereira, J.M. et al. (2020). Wildfire management in Mediterranean-type regions: paradigm change needed. Environmental Research Letters, 15(1), 011001. https://doi.org/10.1088/1748-9326/ab541e.

Nepstad, D., Carvalho, G., Barros, A.C., Alencar, A., Capobianco, J.P., Bishop, J. et al. (2001). Road paving, fire regime feedbacks, and the future of Amazon forests. Forest Ecology and Management, 154(3), 395-407. https://doi.org/10.1016/S0378-1127(01)00511-4.

Nikonovas, T., Spessa, A., Doerr, S.H., Clay, G.D. and Mezbahuddin, S. (2020). Near-complete loss of fire-resistant primary tropical forest cover in Sumatra and Kalimantan. Communications Earth and Environment, 1, 65. https://doi.org/10.1038/s43247-020-00069-4.

Notohadirprawiro, T. (1998). September. Conflict between problem-solving and optimizing approach to land resource development: the case of Central Kalimantan wetlands. In Proceedings of the International Peat Symposium "The Spirit of Peatlands". Sopo, R. (ed.). Jyvaskyla: International Peat Society. 7-9.

O'Hara, K.C., Ranches, J., Roche, L.M., Schohr, T.K., Busch, R.C. and Maier, G.U. (2021). Impacts from Wildfires on Livestock Health and Production: Producer Perspectives. Animals, 11(11), p.3230. https://doi.org/10.3390/ani11113230.

Osterkamp, T.E. (2005). The recent warming of permafrost in Alaska. Global and Planetary Change, 49(3-4), 187-202. https://doi.org/10.1016/j.gloplacha.2005.09.001.

Owens, D. and O'Kane, M. (2020). Final Report of the NSW Bushfire Inquiry. Sydney: New South Wales Government. https://www.dpc.nsw.gov.au/assets/dpc-nsw-gov-au/publications/NSW-Bushfire-Inquiry-1630/Final-Report-of-the-NSW-Bushfire-Inquiry.pdf.

Page, S.E., Rieley, J.O. and Banks, C.J. (2011). Global and regional importance of the tropical peatland carbon pool. Global Change Biology, 17(2), 798-818. https://doi.org/10.1111/j.1365-2486.2010.02279.x.

Page, S.E., Siegert, F., Rieley, J.O., Boehm, H.D.V., Jaya, A. and Limin, S. (2002). The amount of carbon released from peat and forest fires in Indonesia during 1997. Nature, 420, 61-65. https://doi.org/10.1038/nature01131.

Pan, Y., Chen, J.M., Birdsey, R., McCullough, K., He, L. and Deng, F. (2011). Age structure and disturbance legacy of North American forests. Biogeosciences, 8(3), 715-732. https://doi.org/10.5194/bg-8-715-2011.

Panisset, J.S., Libonati, R., Gouveia, C.M.P., Machado-Silva, F., França, D.A., França, J.R.A. et al. (2018). Contrasting patterns of the extreme drought episodes of 2005, 2010 and 2015 in the Amazon Basin. International Journal of Climatology, 38(2), 1096-1104. https://doi.org/10.1002/joc.5224.

Partain, J.L., Alden, S., Strader, H., Bhatt, U.S., Bieniek, P.A., Brettschneider, B.R. et al. (2016). An assessment of the role of anthropogenic climate change in the Alaska fire season of 2015. Bulletin of the American Meteorological Society, 97(12), S14-S18. https://doi.org/10.1175/BAMS-D-16-0149.1.

Pausas, J.G. and Ribeiro, E. (2013). The global fire–productivity relationship. Global Ecology and Biogeography, 22(6), 728-736. https://doi.org/10.1111/geb.12043.

Pivello, V.R. (2011). The use of fire in the Cerrado and Amazonian rainforests of Brazil: past and present. Fire Ecology, 7, 24-39. https://doi.org/10.4996/fireecology.0701024.

Ponomarev, E., Masyagina, O., Litvintsev, K., Ponomareva, T., Shvetsov, E. and Finnikov, K. (2020). The effect of post-fire disturbances on a seasonally thawed layer in the permafrost larch forests of central Siberia. Forests, 11(8), 790. https://doi.org/10.3390/F11080790.

Prentice, I.C., Kelley, D.I., Foster, P.N., Friedlingstein, P., Harrison, S.P. and Bartlein, P.J. (2011). Modeling fire and the terrestrial carbon balance. Global Biogeochemical Cycles, 25(3). https://doi.org/10.1029/2010GB003906.

Restaino, C.M. and Safford, H.D. (2018). Fire and climate change. In Fire in California's Ecosystems. Van Wagtendonk, J.W., Sugihara, N.G., Stephens, S.L., Thode, A.E., Shaffer, K.E. and Fites-Kaufman, J. (eds.). Second edition. Berkeley, CA: University of California Press. Chapter 26, 493-506.

Richardson, J. and Lehmann, A. (2018). Forest Resilience Finance: Opportunities and Challenges. Helping Forests Adapt to Climate Change. London: Parhelion Underwriting. https://www.parhelionunderwriting.com/wp-content/uploads/2018/12/ForestAdaptationFinance_FOEN_Parhelion_ForPrint_FINAL.pdf.

Schmidt, I.B. and Eloy, L. (2020). Fire regime in the Brazilian Savanna: recent changes, policy and management. Flora, 268, 151613. https://doi.org/10.1016/j.flora.2020.151613.

Scholze, M., Knorr, W., Arnell, N.W. and Prentice, I.C. (2006). A climate-change risk analysis for world ecosystems. Proceedings of the National Academy of Sciences of the United States of America, 103(35), 13116-13120. https://doi.org/10.1073/pnas.0601816103.

Seymour, F. (2020). INSIDER: 4 reasons why a jurisdictional approach for REDD+ crediting is superior to a project-based approach, 5 May. https://www.wri.org/insights/insider-4-reasons-why-jurisdictional-approach-redd-crediting-superior-project-based. Accessed 6 October 2021.

Silva, C.V.J., Aragão, L.E.O.C., Barlow, J., Espirito-Santo, F., Young, P.J., Anderson, L.O. et al. (2018). Drought-induced Amazonian wildfires instigate a decadal-scale disruption of forest carbon dynamics. Philosophical Transactions of the Royal Society B: Biological Sciences, 373(1760), 20180043. https://doi.org/10.1098/rstb.2018.0043.

Silva, C.V.J., Aragão, L.E.O.C., Young, P.J., Espirito-Santo, F., Berenguer, E., Anderson, L.O. et al. (2020). Estimating the multi-decadal carbon deficit of burned Amazonian forests. Environmental Research Letters, 15(11), 114023. https://doi.org/10.1088/1748-9326/abb62c.

Silva, P.S., Bastos, A., DaCamara, C.C. and Libonati, R. (2016). Future projections of fire occurrence in Brazil using EC-Earth climate model. Revista Brasileira de Meteorologia, 31(3), 288-297. https://doi.org/10.1590/0102-778631320150142.

Silva, P.S., Bastos, A., Libonati, R., Rodrigues, J.A. and DaCamara, C.C. (2019). Impacts of the 1.5°C global warming target on future burned area in the Brazilian Cerrado. Forest Ecology and Management, 446, 193-203. https://doi.org/10.1016/j.foreco.2019.05.047.

Silva, P.S., Nogueira, J., Rodrigues, J.A., Santos, F.L.M., Pereira, J.M.C., DaCamara, C.C. et al. (2021). Putting fire on the map of Brazilian savanna ecoregions. Journal of

Environmental Management. 296, 113098. https://doi.org/10.1016/j.jenvman.2021.113098.

Silva Junior, C.H.L., Aragão, L.E.O.C., Fonseca, M.G., Almeida, C.T., Vedovato, L.B. and Anderson, L.O. (2018). Deforestation-induced fragmentation increases forest fire occurrence in central Brazilian Amazonia. Forests, 9(6), 305. https://doi.org/10.3390/f9060305.

Sizov, O., Ezhova, E., Tsymbarovich, P., Soromotin, A., Prihod'ko, N., Petäjä, T. et al. (2021). Fire and vegetation dynamics in northwest Siberia during the last 60 years based on high-resolution remote sensing. Biogeosciences, 18(1), 207-228. https://doi.org/10.5194/bg-18-207-2021.

Smit, I.P.J., Asner, G.P., Govender, N., Vaughn, N.R. and Van Wilgen, B.W. (2016). An examination of the potential efficacy of high-intensity fires for reversing woody encroachment in savannas. Journal of Applied Ecology, 53(5), 1623-1633. https://doi.org/10.1111/1365-2664.12738.

Smith, A.J.P., Jones, M.W., Abatzoglou, J.T., Canadell, J.G. and Betts, R.A. (2020). Climate Change Increases the Risk of Wildfires, September 2020. ScienceBrief Review. https://doi.org/10.5281/zenodo.4570195.

Staver, A.C., Archibald, S. and Levin, S.A. (2011). The global extent and determinants of savanna and forest as alternative biome states. Science, 334(6053), 230-232. https://doi.org/10.1126/science.1210465.

Stevens, N., Lehmann C.E.R., Murphy, B.P. and Durigan, G. (2017). Savanna woody encroachment is widespread across three continents. Global Change Biology, 23(1), 235-244. https://doi.org/10.1111/gcb.13409.

Strassburg, B.B.N., Brooks, T., Feltran-Barbieri, R., Iribarrem, A., Crouzeilles, R., Loyola, R. et al. (2017). Moment of truth for the Cerrado hotspot. Nature Ecology and Evolution, 1(4), 0099. https://doi.org/10.1038/s41559-017-0099.

Sullivan, A.L. (2017). Inside the inferno: fundamental processes of wildland fire behaviour. Part 1: combustion chemistry and heat release. Current Forestry Reports, 3, 132-149. https://doi.org/10.1007/s40725-017-0057-0.

Sullivan, A.L., Surawski, N.C., Crawford, D., Hurley, R.J., Volkova, L. and Weston, C.J. (2018). Effect of woody debris on the rate of spread of surface fires in forest fuels in a combustion wind tunnel. Forest Ecology and Management, 424, 236-245. https://doi.org/10.1016/j.foreco.2018.04.039

Surawski, N.C., Sullivan, A.L., Roxburgh, S.H. and Polglase, P.J. (2016). Estimates of greenhouse gas and black carbon emissions from a major Australian wildfire with high spatiotemporal resolution. Journal of Geophysical Research: Atmospheres, 121(16), 9892-9907. https://doi.org/10.1002/2016JD025087.

Swanson, D.K., Sousanes, P.J. and Hill, K. (2021). Increased mean annual temperatures in 2014–2019 indicate permafrost thaw in Alaskan national parks. Arctic, Antarctic, and Alpine Research, 53(1), 1-19. https://doi.org/10.1080/15230430.2020.1859435.

Taskforce on Scaling Voluntary Carbon Markets (2021). Final Report, January 2021. https://www.iif.com/Portals/1/Files/TSVCM_Report.pdf.

Touma, D., Stevenson, S., Lehner, F. and Coats, S. (2021). Human-driven greenhouse gas and aerosol emissions cause distinct regional impacts on extreme fire weather. Nature Communications, 12, 212. https://doi.org/10.1038/s41467-020-20570-w.

Turetsky, M.R., Benscoter, B., Page, S., Rein, G., Van der Werf, G.R. and Watts, A. (2015). Global vulnerability of peatlands to fire and carbon loss. Nature Geoscience, 8(1), 11-14. https://doi.org/10.1038/ngeo2325.

United Nations Framework Convention on Climate Change (2017). Time Series, Annex I Database: GHG Total with LULUCF. http://di.unfccc.int/time_series. Accessed 7 October 2021.

Urák I., Hartel T., Gallé R. and Balog A. (2017). Worldwide peatland degradations and the related carbon dioxide emissions: the importance of policy regulations. Environmental Science and Policy, 69, 57-64. https://doi.org/10.1016/j.envsci.2016.12.012.

Van Oldenborgh, G.J., Krikken, F., Lewis, S., Leach, N.J., Lehner, F., Saunders, K.R. et al. (2021). Attribution of the Australian bushfire risk to anthropogenic climate change. Natural Hazards and Earth System Sciences, 21(3), 941-960. https://doi.org/10.5194/nhess-21-941-2021.

Van Wilgen, B.W., Govender, N., Biggs, H.C., Ntsala, D. and Funda, X.N. (2004). Response of savanna fire regimes to changing fire-management policies in a large African national park. Conservation Biology, 18(6), 1533-1540. https://doi.org/10.1111/j.1523-1739.2004.00362.x.

Veraverbeke, S., Rogers, B.M., Goulden, M.L., Jandt, R.R., Miller, C.E., Wiggins, E.B. et al. (2017). Lightning as a major driver of recent large fire years in North American boreal forests. Nature Climate Change, 7(7), 529-534. https://doi.org/10.1038/nclimate3329.

Walker, X.J., Baltzer, J.L., Cumming, S.G., Day, N.J., Ebert, C., Goetz, S. et al. (2019). Increasing wildfires threaten historic carbon sink of boreal forest soils. Nature, 572(7770), 520-523. https://doi.org/10.1038/s41586-019-1474-y.

Walsh, J.E., Ballinger, T.J., Euskirchen, E.S., Hanna, E., Mård, J., Overland, J.E. et al. (2020). Extreme weather and climate events in northern areas: a review. Earth-Science Reviews, 209, 103324. https://doi.org/10.1016/j.earscirev.2020.103324.

Wich, S.A., Gaveau, D., Abram, N., Ancrenaz, M., Baccini, A., Brend, S. et al. (2012). Understanding the impacts of land-use policies on a threatened species: is there a future for the Bornean orang-utan? PloS One, 7(11), e49142. https://doi.org/10.1371/journal.pone.0049142.

Williams, A.P., Abatzoglou, J.T., Gershunov, A., Guzman-Morales, J., Bishop, D.A., Balch, J.K. et al. (2019). Observed impacts of anthropogenic climate change on wildfire in California. Earth's Future, 7(8), 892-910. https://doi.org/10.1029/2019EF001210.

Williams, R.J., Bradstock, R.A., Matthews, S., Price, O.F., Sullivan, A.F. and Watson, P. (2011). Climate Change, Fire Regimes and Risk in Australian Landscapes: Lessons for

Adaptation – Report to the Department of Climate Change and Energy Efficiency. Darwin: Commonwealth Scientific and Industrial Research Organisation, Ecosystem Sciences.

Witze, A. (2020). The Arctic is burning like never before: and that's bad news for climate change. Nature, 585(7825), 336-337. https://doi.org/10.1038/d41586-020-02568-y.

Woo, S.H.L., Liu, J.C., Yue, X., Mickley, L.J. and Bell, M.L. (2020). Air pollution from wildfires and human health vulnerability in Alaskan communities under climate change. Environmental Research Letters, 15(9), 94019. https://doi.org/10.1088/1748-9326/ab9270.

World Bank (2015). Indonesia Economic Quarterly: Reforming Amid Uncertainty, December 2015. Jakarta. https://www.worldbank.org/en/news/feature/2015/12/15/indonesia-economic-quarterly-december-2015.

York, A.D., Bhatt, U.S., Thoman, R.L. and Ziel, R. (2018). Wildland fire in boreal and arctic North America. In State of the Climate in 2017, Special Supplement to the Bulletin of the American Meteorological Society, 99(8), S167-S169. https://akfireconsortium.files.wordpress.com/2018/06/york-et-al_wildlandfire_ch05_arctic.pdf.

Young, A.M., Higuera, P.E., Duffy, P.A. and Hu, F.S. (2017). Climatic thresholds shape northern high-latitude fire regimes and imply vulnerability to future climate change. Ecography, 40(5), 606-617. https://doi.org/10.1111/ecog.02205.

Zhou, X., Josey, K., Kamareddine, L., Caine, M.C., Liu, T., Cooper, M. and Dominici, F. (2021). Excess of COVID-19 cases and deaths due to fine particulate matter exposure during the 2020 wildfires in the United States. Science Advances. 7(33). https://doi.org/10.1126/sciadv.abi8789.

Chapter 3

Adetona, O., Reinhardt, T.E., Domitrovich, J., Broyles, G., Adetona, A.M., Kleinman, M.T. et al. (2016). Review of the health effects of wildland fire smoke on wildland firefighters and the public. Inhalation Toxicology, 28(3), 95-139. https://doi.org/10.3109/08958378.2016.1145771.

Amazon Environmental Research Institute, Institute for Health Policy Studies, and Human Rights Watch (2020). Technical Report: "The Air is Unbearable". Health Impacts of Deforestation-Related Fires in the Brazilian Amazon. https://www.hrw.org/sites/default/files/media_2020/08/brazil0820_web.pdf.

Arriagada, N.B., Palmer, A.J., Bowman, D.M.J.S, Morgan, G.G., Jalaludin, B.B. and Johnston, F.H. (2020). Unprecedented smoke-related health burden associated with the 2019–20 bushfires in eastern Australia. The Medical Journal of Australia, 213(6), 282-283. https://doi.org/10.5694/mja2.50545.

Augusto, S., Ratola, N., Tarín-Carrasco, P., Jiménez-Guerrero, P., Turco, M., Schuhmacher, M. et al. (2020). Population exposure to particulate-matter and related mortality due to the Portuguese wildfires in October 2017 driven by storm Ophelia. Environment International, 144, 106056. https://doi.org/10.1016/j.envint.2020.106056.

Axelrad, D.A., Bellinger, D.C., Ryan, L.M. and Woodruff, T.J., 2007. Dose–response relationship of prenatal mercury exposure and IQ: an integrative analysis of epidemiologic data. Environmental health perspectives, 115(4), pp.609-615.
https://doi.org/10.1289/ehp.9303.

Blanchi, R., Leonard, J., Haynes, K., Opie, K., James, M. and De Oliveira, F.D. (2014). Environmental circumstances surrounding bushfire fatalities in Australia 1901-2011. Environmental Science and Policy, 37, 192-203. https://doi.org/10.1016/j.envsci.2013.09.013.

Blanchi, R., Leonard, J., Haynes, K., Opie, K., James, M., Kilinc, M. et al. (2012). Life and House Loss Database Description and Analysis: Final Report. Commonwealth Scientific and Industrial Research Organisation. https://doi.org/10.4225/08/584af3d322e91.

Bowman, D.M.J.S. (2018). Wildfire science is at a loss for comprehensive data. Nature, 560(7), 7-8. https://doi.org/10.1038/d41586-018-05840-4.

Bowman, D.M.J.S., Williamson, G.J., Abatzoglou, J.T., Kolden, C.A., Cochrane, M.A. and Smith, A.M. (2017). Human exposure and sensitivity to globally extreme wildfire events. Nature Ecology and Evolution, 1(3), 0058. https://doi.org/10.1038/s41559-016-0058.

Brown, F., Santos, G., Pires, F. and da Costa, C. (2011). Brazil: Drought and Fire Response in the Amazon: World Resources Report Case Study. Washington, DC: World Resources Report.

Brown, I.F., Schroeder, W., Setzer, A., de Los Rios Maldonado, M., Pantoja, N., Duarte, A. et al. (2006). Monitoring fires in southwestern Amazonia rain forests. Eos, 87(26), 253-255. https://doi.org/10.1029/2006EO260001.

Burkhardt, J., Bayham, J., Wilson, A., Berman, J.D., O'Dell, K., Ford, B. et al. (2020). The relationship between monthly air pollution and violent crime across the United States. Journal of Environmental Economics and Policy, 9(2), 188-205. https://doi.org/10.1080/21606544.2019.1630014.

Butt, E.W., Conibear, L., Reddington, C.L., Darbyshire, E., Morgan, W.T., Coe, H. et al. (2020). Large air quality and human health impacts due to Amazon forest and vegetation fires. Environmental Research Communications, 2(9), 095001. https://doi.org/10.1088/2515-7620/abbodb.

Campanharo, W.A., Lopes, A.P., Anderson, L.O., da Silva, T.F.M.R. and Aragão, L.E.O.C. (2019). Translating fire impacts in southwestern Amazonia into economic costs. Remote Sensing, 11(7), 764. https://doi.org/10.3390/rs11070764.

Cascio, W.E. (2018). Wildland fire smoke and human health. Science of the Total Environment, 624, 586-595. https://doi.org/10.1016/j.scitotenv.2017.12.086.

Castellnou, M., Guiomar, N., Rego, F. and Fernandes, P.M. (2018). Fire growth patterns in the 2017 mega fire episode of October 15, central Portugal. In Advances in Forest Fire Research 2018. Viegas, D.X. (ed.). Coimbra: Imprensa da Universidade de Coimbra. 447-453. https://doi.org/10.14195/978-989-26-16-506_48.

Centro de Previsão de Tempo e Estudos Climáticos/Instituto Nacional De Pesquisas Espaciai (2021). Monitoramento dos Focos Ativos por Países [Monitoring of Active Fires by Country] Database. https://queimadas.dgi.inpe.br/queimadas/portal-static/estatisticas_paises/. Accessed 7 October 2021.

Chile (2020). President Piñera implements the National Forest Fire Prevention and Firefighting Plan for 2020-2021: "We have prepared, responsibly and thoroughly", 9 October. https://www.gob.cl/en/news/president-pinera-implements-national-forest-fire-prevention-and-firefighting-plan-2020-2021-we-have-prepared-responsibly-and-thoroughly/. Accessed 7 October 2021.

Da Motta, R.S., Cardoso Mendonça, M.J., Nepstad, D., Diaz, M.C.V., Alencar, A., Gomes, J.C. et al. (2002). O Custo Econômico Do Fogo Na Amazônia. Rio de Janeiro: Instituto de Pesquisa Econômica Aplicada. https://www.ipea.gov.br/portal/images/stories/PDFs/TDs/td_0912.pdf.

Dennekamp, M., Straney, L.D., Erbas, B., Abramson, M.J., Keywood, M., Smith, K. et al. (2015). Forest fire smoke exposures and out-of-hospital cardiac arrests in Melbourne, Australia: a case-crossover study. Environmental Health Perspectives, 123(10), 959-964. https://doi.org/10.1289%2Fehp.1408436.

Do Carmo, C.N., Alves, M.B. and De Souza Hacon, S. (2013). Impact of biomass burning and weather conditions on children's health in a city of Western Amazon region. Air Quality, Atmosphere and Health, 6(2), 517-525. https://doi.org/10.1007/s11869-012-0191-6.

Dodd, W., Scott, P., Howard, C., Scott, C., Rose, C., Cunsolo, A. et al. (2018). Lived experience of a record wildfire season in the Northwest Territories, Canada. Canadian Journal of Public Health, 109(3), 327-337. https://doi.org/10.17269/s41997-018-0070-5.

Doerr, S.H. and Santín, C. (2016). Global trends in wildfire and its impacts: perceptions versus realities in a changing world. Philosophical Transactions of the Royal Society B: Biological Sciences, 371(1696), 20150345. https://doi.org/10.1098/rstb.2015.0345.

Eisenman, D., McCaffrey, S., Donatello, I. and Marshal, G. (2015). An ecosystems and vulnerable populations perspective on solastalgia and psychological distress after a wildfire. Ecohealth, 12(4), 602-610. https://doi.org/10.1007/s10393-015-1052-1.

Fellows, M., Paye, V., Alencar, A., Nicácio, M., Castrol, I., Coelho, M.E. et al. (2020). They Are not Numbers. They Are Lives! COVID-19 Threatens Indigenous Peoples in the Brazilian Amazon. Amazon Environmental Research Institute and Coordination of the Indigenous Organizations of the Brazilian Amazon. https://ipam.org.br/wp-content/uploads/2020/06/NT_COVID-english.pdf.

Frankenberg, E., McKee, D. and Thomas, D. (2005). Health consequences of forest fires in Indonesia. Demography, 42(1), 109-129. https://doi.org/10.1353/dem.2005.0004.

Franzi, L.M., Bratt, J.M., Williams, K.M. and Last, J.A. (2011). Why is particulate matter produced by wildfires toxic to lung macrophages? Toxicology and Applied Pharmacology, 257(2), 182-188. https://doi.org/10.1016%2Fj.taap.2011.09.003.

Glauber, A.J., Moyer, S., Adriani, M. and Iwan, G. (2016). The Cost of Fire: An Economic Analysis of Indonesia's 2015 Fire Crisis. Indonesia Sustainable Landscapes Knowledge Note No. 1. Jakarta: World Bank. https://openknowledge.worldbank.org/handle/10986/23840.

Groot, E., Caturay, A., Khan, Y. and Copes, R. (2019). A systematic review of the health impacts of occupational exposure to wildland fires. International Journal of Occupational Medicine and Environmental Health, 32(2), 121-140. https://doi.org/10.13075/ijomeh.1896.01326.

Guerreiro, J., Fonseca, C., Salgueiro, A., Fernandes, P., Lopez, E., de Neufville, R. et al. (2017). Análise e Apuramento dos Factos Relativos aos Incêndios que Ocorreram em Pedrógão Grande, Castanheira de Pêra, Ansião, Alvaiázere, Figueiró dos Vinhos, Arganil, Góis, Penela, Pampilhosa da Serra, Oleiros e Sertã entre 17 e 24 de junho de 2017. Lisbon: Comissão Técnica Independente. https://www.parlamento.pt/Documents/2017/Outubro/RelatórioCTI_VF%20.pdf.

Haikerwal, A., Akram, M., Del Monaco, A., Smith, K., Sim, M.R., Meyer, M. et al. (2015). Impact of fine particulate matter (PM2.5) exposure during wildfires on cardiovascular health outcomes. Journal of the American Heart Association, 4(7), e001653. https://dx.doi.org/10.1161%2FJAHA.114.001653.

Hallema, D.W., Sun, G., Caldwell, P., Robinne, F.-N., Bladon, K.D., Norman, S. et al. (2019). Wildland Fire Impacts on Water Yield Across the Contiguous United States. General Technical Report SRS-238. Asheville, NC: United States Department of Agriculture, Forest Service. https://doi.org/10.2737/SRS-GTR-238.

Harrison, M.E., Wijedasa, L.S., Cole, L.E.S., Cheyne, S.M., Choiruzzad, S.A.B., Chua, L. et al. (2020). Tropical peatlands and their conservation are important in the context of COVID-19 and potential future (zoonotic) disease pandemics. PeerJ, 8, e10283. https://doi.org/10.7717/peerj.10283.

Haynes, K., Handmer, J., McAneney, J., Tibbits, A. and Coates, L. (2010). Australian bushfire fatalities 1900–2008: exploring trends in relation to the 'Prepare, stay and defend or leave early' policy. Environmental Science and Policy, 13(3), 185-194. https://doi.org/10.1016/j.envsci.2010.03.002.

Heft-Neal, S., Driscoll, A., Yang, W., Shaw, G. and Burke, M., 2022. Associations between wildfire smoke exposure during pregnancy and risk of preterm birth in California. Environmental Research, 203, p.111872.

https://doi.org/10.1016/j.envres.2021.111872.

Henderson, S.B., Brauer, M., Macnab, Y.C. and Kennedy, S.M. (2011). Three measures of forest fire smoke exposure and their associations with respiratory and cardiovascular health outcomes in a population-based cohort. Environmental Health Perspectives, 119(9), 1266-1271. https://doi.org/10.1289/ehp.1002288.

Ho, A.F.W., Wah, W., Earnest, A., Ng, Y.Y., Xie, Z., Shahidah, N. et al. (2018). Health impacts of the Southeast Asian haze problem: a time-stratified case crossover study of the relationship between ambient air pollution and sudden cardiac deaths in Singapore. International Journal of Cardiology, 271, 352-358. https://doi.org/10.1016/j.ijcard.2018.04.070.

Hope, E.S., McKenney, D.W., Pedlar, J.H., Stocks, B.J. and Gauthier, S. (2016). Wildfire suppression costs for Canada under a changing climate. PloS One, 11(8). e0157425. https://doi.org/10.1371/journal.pone.0157425

Huttunen, K., Siponen, T., Salonen, I., Yli-Tuomi, T., Aurela, M., Dufva, H. et al. (2012). Low-level exposure to ambient particulate matter is associated with systemic inflammation in ischemic heart disease patients. Environmental Research, 116, 44-51. https://doi.org/10.1016/j.envres.2012.04.004.

Ignotti, E., Valente, J.G., Longo, K.M., Freitas, S.R., Hacon, S.D.S. and Netto, P.A. (2010). Impact on human health of particulate matter emitted from burnings in the Brazilian Amazon region. Revista de Saúde Pública, 44(1), 121-130. https://doi.org/10.1590/s0034-89102010000100013.

Johnston, F.H., Borchers-Arriagada, N., Morgan, G.G., Jalaudin, B., Palmer, A.J., Williamson, G.J. et al. (2021). Unprecedented health costs of smoke-related PM2.5 from the 2019–20 Australian megafires. Nature Sustainability, 4(1), 42-47. https://doi.org/10.1038/s41893-020-00610-5.

Jones, C.G., Rappold, A.G., Vargo, J., Cascio, W.E., Kharrazi, M., McNally, B. et al. (2020). Out-of-hospital cardiac arrests and wildfire-related particulate matter during 2015–2017 California wildfires. Journal of the American Heart Association, 9(8), e014125. https://doi.org/10.1161/jaha.119.014125.

Karthikeyan, S., Balasubramanian, R. and Iouri, K. (2006). Particulate air pollution from bushfires: human exposure and possible health effects. Journal of Toxicology and Environmental Health A, 69(21), 1895-908. https://doi.org/10.1080/15287390600751264.

Kondo, M.C., De Roos, A.J., White, L.S., Heilman, W.E., Mockrin, M.H., Gross-Davis, C.A. et al. (2019). Meta-analysis of heterogeneity in the effects of wildfire smoke exposure on respiratory health in North America. International Journal of Environmental Research and Public Health, 16(6), 960. https://doi.org/10.3390/ijerph16060960.

Liu, J.C., Pereira, G., Uhl, S.A., Bravo, M.A. and Bell, M.L. (2015). A systematic review of the physical health impacts from non-occupational exposure to wildfire smoke. Environmental Research, 136, 120-132. https://doi.org/10.1016/j.envres.2014.10.015.

Machado-Silva, F., Libonati, R., de Lima, T.F.M., Peixoto, R.B., de Almeida França, J.R., Magalhães, M.D.A.F.M. et al. (2020). Drought and fires influence the respiratory diseases hospitalizations in the Amazon. Ecological Indicators, 109, 105817. https://doi.org/10.1016/j.ecolind.2019.105817.

Martin, D.A. (2019). Linking fire and the United Nations Sustainable Development Goals. Science of the Total Environment, 662, 547-558. https://doi.org/10.1016/j.scitotenv.2018.12.393.

Milton, L.A. and White, A.R. (2020). The potential impact of bushfire smoke on brain health. Neurochemistry International, 139, 104796. https://doi.org/10.1016/j.neuint.2020.104796.

Nakayama Wong, L.S., Aung, H.H., Lamé, M.W., Wegesser, T.C. and Wilson, D.W. (2011). Fine particulate matter from urban ambient and wildfire sources from California's San Joaquin Valley initiate differential inflammatory, oxidative stress, and xenobiotic responses in human bronchial epithelial cells. Toxicology In Vitro, 25(8), 1895-1905. https://doi.org/10.1016/j.tiv.2011.06.001.

National Interagency Fire Center (2021). Suppression Costs: Federal Firefighting Costs (Suppression Only). https://www.nifc.gov/fire-information/statistics/suppression-costs.

Pinto, M.M., Hurduc, A., Trigo, R.M., Trigo, I.F. and DaCamara, C.C. (2018). The extreme weather conditions behind the destructive fires of June and October 2017 in Portugal. In Advances in Forest Fire Research 2018. Viegas, D.X. (ed.). Coimbra: Imprensa da Universidade de Coimbra. 138-145. https://doi.org/10.14195/978-989-26-16-506_13.

Plucinski, M.P. (2019). Contain and control: wildfire suppression effectiveness at incidents and across landscapes. Current Forestry Reports, 5(1), 20-40. https://doi.org/10.1007/s40725-019-00085-4.

Reddington, C.L., Butt, E.W., Ridley, D.A., Artaxo, P., Morgan, W.T., Coe, H. et al. (2015). Air quality and human health improvements from reductions in deforestation-related fire in Brazil. Nature Geoscience, 8(1), 768-771. https://doi.org/10.1038/ngeo2535.

Reid, C.E., Brauer, M., Johnston, F.H., Jerrett, M., Balmes, J.R. and Elliott, C.T. (2016). Critical review of health impacts of wildfire smoke exposure. Environmental Health Perspectives, 124(9), 1334-1343. http://dx.doi.org/10.1289/ehp.1409277.

Rifkin, D.I., Long, M.W. and Perry, M.J. (2018). Climate change and sleep: a systematic review of the literature and conceptual framework. Sleep Medicine Reviews, 42, 3-9. https://doi.org/10.1016/j.smrv.2018.07.007.

Sánchez-Benítez, A., García-Herrera, R., Barriopedro, D., Sousa, P.M. and Trigo, R.M. (2018). June 2017: the earliest European summer mega-heatwave of reanalysis period. Geophysical Research Letters, 45(4), 1955-1962. https://doi.org/10.1002/2018GL077253.

Soares-Filho, B.S., Rajao, R., Costa, W.L.S, Lima, L.S., Oliveira, A.R., Rodrigues, H.O. et al. (2017). Impacto de políticas públicas voltadas à implementação do novo código florestal. Relatório de Projeto, Centro do Sensoriamento Remoto. Doctoral Thesis. Universidade Federal de Minas Gerais.

Stocks, B.J. and Martell, D.L. (2016). Forest fire management expenditures in Canada: 1970–2013. The Forestry Chronicle, 92(3), 298-306. https://doi.org/10.5558/tfc2016-056.

Tacconi, L. (2003). Fires in Indonesia: Causes, Costs and Policy Implications. Bogor: Center for International Forestry Research. https://doi.org/10.17528/cifor/001130.

Tan, W.C., Qui, D., Liam, B.L., Ng, T.P., Lee, S.H., Van Eeden, S.F. et al. (2000). The human bone marrow response to acute air pollution caused by forest fires. American Journal of Respiratory and Critical Care Medicine, 161(4), 1213-1217. https://doi.org/10.1164/ajrccm.161.4.9904084.

Turco, M., Jerez, S., Augusto, S., Tarín-Carrasco, P., Ratola, N., Jiménez-Guerrero, P. et al. (2019). Climate drivers of the 2017 devastating fires in Portugal. Scientific Reports, 9, 13886. https://doi.org/10.1038/s41598-019-50281-2.

Werth, P.A., Potter, B.E., Alexander, M.E., Clements, C.B., Cruz, M.G., Finney, M.A. et al. (2016). Synthesis of Knowledge of Extreme Fire Behavior: Volume 2 for Fire Behavior Specialists, Researchers, and Meteorologists. General Technical Report PNW-GTR-891. Portland, OR: United States Department of Agriculture, Forest Service. https://doi.org/10.2737/PNW-GTR-891.

World Bank (2020). World Bank Policy Note: Managing Wildfire in a Changing Climate, March 2020. Washington, DC. https://www.profor.info/sites/profor.info/files/PROFOR_ManagingWildfires_2020_final.pdf.

Chapter 4

Abaimov A.P. (2010). Geographical Distribution and Genetics of Siberian Larch Species. In: Osawa A., Zyryanova O., Matsuura Y., Kajimoto T. and Wein R. (eds.) Permafrost Ecosystems. Ecological Studies (Analysis and Synthesis). vol 209. Dordrecht: Springer. 41–58. https://doi.org/10.1007/978-1-4020-9693-8_3

Abaimov, A.P., Zyryanova, O.A. and Prokushkin, S.G. (2002). Long-term investigations of larch forests in cryolithic zone of Siberia: brief history, recent results and possible changes under global warming. Eurasian Journal of Forest Research, 5(2), 95-106. http://hdl.handle.net/2115/22155.

Abell, R., Asquith, N., Boccaletti, G., Bremer, L., Chapin. E., Erickson-Quiroz, A. et al. (2017). Beyond the Source: The Environmental, Economic and Community Benefits of Source Water Protection. Arlington, VA: The Nature Conservancy. https://conservationtools.org/library_items/1424-Beyond-the-Source-The-environmental-economic-and-community-benefits-of-source-water-protection.

Alexander, H.D. and Mack, M.C. (2016). A canopy shift in interior Alaskan boreal forests: consequences for above- and belowground carbon and nitrogen pools during post-fire succession. Ecosystems. 19: 98–114. https://doi.org/10.1007/s10021-015-9920-7

Alexander, H.D., Mack, M.C., Goetz, S., Loranty, M.M., Beck, P.S.A., Earl, K. et al. (2012). Carbon accumulation patterns during post-fire succession in Cajander larch (Larix cajanderi) forests of Siberia. Ecosystems, 15(7), 1065-1082. https://doi.org/10.1007/s10021-012-9567-6.

Alexander, H.D., Natali, S.M., Loranty, M.M., Ludwig, S.M., Spektor, V.V., Davydov, S. et al. (2018). Impacts of increased soil burn severity on larch forest regeneration on permafrost soils of far northeastern Siberia. Forest Ecology and Management, 417, 144-153. https://doi.org/10.1016/j.foreco.2018.03.008.

Aragão, L.E.O.C., Anderson, L.O., Fonseca, M.G., Rosan, T.M., Vedovato, L.B., Wagner, F.H. et al. (2018). 21st century drought-related fires counteract the decline of Amazon deforestation carbon emissions. Nature Communications, 9(1), 536. https://doi.org/10.1038/s41467-017-02771-y.

Armenteras, D., González, T.M. and Retana, J. (2013). Forest fragmentation and edge influence on fire occurrence and intensity under different management types in Amazon forests. Biological Conservation, 159, 73-79. https://doi.org/10.1016/j.biocon.2012.10.026.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Mac Nally, R. et al. (2016). Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature, 535, 144-147. https://doi.org/10.1038/nature18326.

Bell, D.T., Plummer, J.A. and Taylor, S.K. (1993). Seed germination ecology in southwestern Western Australia. The Botanical Review, 59(1), 24-73. https://doi.org/10.1007/BF02856612.

Berenguer, E., Ferreira, J., Gardner, T.A., Aragão, L.E.O.C., De Camargo, P.B., Cerri, C.E. et al. (2014). A large-scale field assessment of carbon stocks in human-modified tropical forests. Global Change Biology, 20(12), 3713-3726. https://doi.org/10.1111/gcb.12627.

Berenguer, E., Malhi, Y., Brando, P., Cordeiro, A.C.N., Ferreira, J., França, F. et al. (2018). Tree growth and stem carbon accumulation in human-modified Amazonian forests following drought and fire. Philosophical Transactions of the Royal Society B: Biological Sciences, 373(1760), 20170308. https://doi.org/10.1098/rstb.2017.0308.

Bixby, R.J., Cooper, S.D., Gresswell, R.E., Brown, L.E., Dahm, C.N. and Dwire, K.A. (2015). Fire effects on aquatic ecosystems: an assessment of the current state of the science. Freshwater Science, 34(4), 1340-1350. https://doi.org/10.1086/684073.

Bladon, K.D., Emelko, M.B., Silins, U. and Stone M. (2014). Wildfire and the future of water supply. Environmental Science and Technology, 48(16), 8936-8943. https://doi.org/10.1021/es500130g.

Boer, M. M., Price, O. F. and Bradstock, R. A. (2015). Wildfires: Weigh policy effectiveness. Science, 350(6263), 920-920.

Bodí, M.B., Martin, D.A., Balfour, V.N., Santín, C., Doerr, S.H., Pereira, P. et al. (2014). Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth-Science Reviews, 130, 103-127. https://doi.org/10.1016/j.earscirev.2013.12.007.

Bond, W.J. and van Wilgen, B.W. (1996). Fire and Plants. London: Chapman & Hall.

Bond, W.J., Woodward, F.I. and Midgley, G.F. (2005). The global distribution of ecosystems in a world without fire.

New Phytologist, 165(2), 525-538. https://doi.org/10.1111/j.1469-8137.2004.01252.x.

Bowman D.M.J.S., Murphy B.P., Neyland, D.L.J., Williamson, G.J. and Prior L.D. (2014). Abrupt fire regime change may cause landscape-wide loss of mature obligate seeder forests. Global Change Biology, 20(3), 1008-1015. https://doi.org/10.1111/gcb.12433.

Bradstock, R.A., Bedward, M., Kenny, B.J. and Scott, J. (1998). Spatially-explicit simulation of the effect of prescribed burning on fire regimes and plant extinctions in shrublands typical of south-eastern Australia. Biological Conservation, 86(1), 83-95. https://doi.org/10.1016/S0006-3207(97)00170-5.

Bradstock, R.A., Bedward, M., Scott, J. and Keith, D.A. (1996). Simulation of the effect of spatial and temporal variation in fire regimes on the population viability of a Banksia species. Conservation Biology, 10(3), 776-784. https://doi.org/10.1046/j.1523-1739.1996.10030776.x.

Bradshaw, S.D., Dixon, K.W., Lambers, H., Cross, A.T., Bailey, J. and Hopper, S.D. (2018). Understanding the long-term impact of prescribed burning in Mediterranean-climate biodiversity hotspots, with a focus on south-western Australia. International Journal of Wildland Fire, 27(10), 643-657. https://doi.org/10.1071/WF18067.

Brando, P.M., Balch, J.K., Nepstad, D.C., Morton, D.C., Putz, F.E., Coe, M.T. et al. (2014). Abrupt increases in Amazonian tree mortality due to drought-fire interactions. Proceedings of the National Academy of Sciences of the United States of America, 111(17), 6347-6352. https://doi.org/10.1073/pnas.1305499111.

Brandt, J.P., Flannigan, M.D., Maynard, D.G., Thompson, I.D. and Volney, W.J.A. (2013). An introduction to Canada's boreal zone: ecosystem processes, health, sustainability, and environmental issues. Environmental Reviews, 21(4), 207-226. https://doi.org/10.1139/er-2013-0040.

Brauman K.A. (2015). Hydrologic ecosystem services: linking ecohydrologic processes to human well-being in water research and watershed management. Wiley Interdisciplinary Reviews: Water, 2(4), 345-358. https://doi.org/10.1002/wat2.1081.

Brotons, L., Herrando, S. and Pons, P. (2008). Wildfires and the expansion of threatened farmland birds: the ortolan bunting Emberiza hortulana in Mediterranean landscapes. Journal of Applied Ecology, 45(4), 1059-1066. http://doi.org/10.1111/j.1365-2664.2008.01467.x.

Burke, M.P., Hogue, T.S., Kinoshita, A.M., Barco, J., Wessel, C. and Stein, E.D. (2013). Pre-and post-fire pollutant loads in an urban fringe watershed in Southern California. Environmental Monitoring and Assessment, 185(12), 10131-10145. https://doi.org/10.1007/s10661-013-3318-9.

Burton, C., Betts, R.A., Jones, C.D., Feldpausch, T.R., Cardoso, M. and Anderson, L.O. (2020). El Niño driven changes in global fire 2015/16. Frontiers in Earth Science, 8. https://doi.org/10.3389/feart.2020.00199.

Cai, W., Yang, J., Liu, Z., Hu, Y. and Weisberg, P.J. (2013). Post-fire tree recruitment of a boreal larch forest in northeast China. Forest Ecology and Management, 307, 20-29. https://doi.org/10.1016/j.foreco.2013.06.056.

Campioli, M., Samson, R., Michelsen, A., Jonasson, S., Baxter, R. and Lemeur, R. (2009). Nonvascular contribution to ecosystem NPP in a subarctic heath during early and late growing season. Plant Ecology, 202(1), 41-53. http://dx.doi.org/10.1007/s11258-008-9527-6.

Certini, G. (2005). Effects of fire on properties of forest soils: a review. Oecologia, 143(1), 1-10. https://doi.org/10.1007/s00442-004-1788-8.

Chia, E.K., Bassett, M., Leonard, S.W.J., Holland, G.J., Ritchie, E.G., Clarke, M.F. et al. (2016). Effects of the fire regime on mammal occurrence after wildfire: site effects vs landscape context in fire-prone forests. Forest Ecology and Management, 363, 130-139. https://doi.org/10.1016/j.foreco.2015.12.008.

Chia, E.K., Bassett, M., Nimmo, D.G., Leonard, S.W.J., Ritchie, E.G., Clarke, M.F. et al. (2015). Fire severity and fire-induced landscape heterogeneity affect arboreal mammals in fire-prone forests. Ecosphere, 6(10), 1-14. https://doi.org/10.1890/ES15-00327.1.

Chuvieco, E., Pettinari, M.L., Lizundia-Loiola, J., Storm, T. and Padilla Parellada, M. (2018). ESA Fire Climate Change Initiative (Fire_cci): MODIS Fire_cci Burned Area Pixel Product, Version 5.1 Dataset. http://dx.doi.org/10.5285/58f00d8814064b79a0c49662ad3af537. Accessed 7 October 2021.

Cochrane, M.A. (2003). Fire science for rainforests. Nature, 421(6926), 913-918. https://doi.org/10.1038/nature01437.

Connell, J.H. (1978). Diversity in tropical rain forests and coral reefs: high diversity of trees and corals is maintained only in a nonequilibrium state. Science, 199(4335), 1302-1310. https://doi.org/10.1126/science.199.4335.1302.

Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M.D. et al. (2020). Wildfire-driven forest conversion in western North American landscapes. BioScience, 70(8), 659-673. https://doi.org/10.1093/biosci/biaa061.

Creed, I.F., Jones, J.A., Archer, E., Claassen, M., Ellison, D., McNulty, S.G. et al. (2019). Managing forests for both downstream and downwind water. Frontiers in Forests and Global Change, 2, 1-8. https://doi.org/10.3389/ffgc.2019.00064.

Dahm, C.N., Candelaria-Ley, R.I., Reale, C.S., Reale, J.K. and Van Horn, D.J. (2015). Extreme water quality degradation following a catastrophic forest fire. Freshwater Biology, 60(12), 2584-2599. https://doi.org/10.1111/fwb.12548.

Dalrymple, S.E. and Safford, H.D. (2019). Ants, wind, and low litter deposition contribute to the maintenance of fire-protective clearings around Jeffrey pine trees (Pinus jeffreyi). Forest Ecology and Management, 438(1), 44-50. http://doi.org/10.1016/j.foreco.2019.01.043.

D'Antonio, C.M. and Vitousek, P.M. (1992). Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and

Systematics, 23(1), 63-87. https://doi.org/10.1146/annurev.es.23.110192.000431.

Davies, K.W. (2011). Plant community diversity and native plant abundance decline with increasing abundance of an exotic annual grass. Oecologia, 167(2), 481-491. https://doi.org/10.1007/s00442-011-1992-2.

DeBano, L.F., Neary, D.G. and Ffolliott, P.F. (1998). Fire's Effects on Ecosystems. New York, NY: John Wiley and Sons.

Didham, R.K., Tylianakis, J.M., Gemmell, N.J., Rand, T.A. and Ewers, R.M. (2007). Interactive effects of habitat modification and species invasion on native species decline. Trends in Ecology and Evolution, 22(9), 489-496. https://doi.org/10.1016/j.tree.2007.07.001.

Driscoll, D.A., Smith, A.L., Blight, S. and Sellar, I. (2020). Interactions among body size, trophic level, and dispersal traits predict beetle detectability and occurrence responses to fire. Ecological Entomology, 45(2), 300-310. https://doi.org/10.1111/een.12798.

Dunham, J.B., Young, M.K., Gresswell, R.E. and Rieman, B.E. (2003). Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management, 178(1-2), 183-196. https://doi.org/10.1016/S0378-1127(03)00061-6.

Ellison, D., Futter, M.N. and Bishop, K. (2012). On the forest cover-water yield debate: from demand- to supply-side thinking. Global Change Biology, 18(3), 806-820. https://doi.org/10.1111/j.1365-2486.2011.02589.x.

Ellison, D., Morris, C.E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarso, D. et al. (2017). Trees, forests and water: cool insights for a hot world. Global Environmental Change, 43, 51-61. https://doi.org/10.1016/j.gloenvcha.2017.01.002.

Feikema, P.M., Sherwin, C.B. and Lane, P.N.J. (2013). Influence of climate, fire severity and forest mortality on predictions of long-term streamflow: potential effect of the 2009 wildfire on Melbourne's water supply catchments. Journal of Hydrology, 488, 1-16. https://doi.org/10.1016/j.jhydrol.2013.02.001.

Feurdean, A., Florescu, G., Tan–u, I., Vannière, B., Diaconu, A.C., Pfeiffer, M. et al. (2020). Recent fire regime in the southern boreal forests of western Siberia is unprecedented in the last five millennia. Quaternary Science Reviews, 244, 106495. https://doi.org/10.1016/j.quascirev.2020.106495.

Fontaine, J.B. and Kennedy, P.L. (2012). Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in US fire-prone forests. Ecological Applications, 22(5), 1547-1561. https://doi.org/10.1890/12-0009.1.

Foster, C.N., Sato, C.F., Lindenmayer, D.B. and Barton, P.S. (2016). Integrating theory into disturbance interaction experiments to better inform ecosystem management. Global Change Biology, 22(4), 1325-1335. https://doi.org/10.1111/gcb.13155.

Furyaev, V.V., Vaganov, E.A., Tchebakova, N.M. and Valendik, E.N. (2001). Effects of fire and climate on successions and structural changes in the Siberian boreal forest. Eurasian Journal of Forest Research, 2, 1-15. http://hdl.handle.net/2115/22125.

Garcia, L.C., Szabo, J.K., de Oliveira Roque, F., Pereira, A.M.M, Nunes da Cunha, C., Damasceno-Júnior, G.A. et al. (2021). Record-breaking wildfires in the world's largest continuous tropical wetland: integrative fire management is urgently needed for both biodiversity and humans. Journal of Environmental Management, 293, 112870. https://doi.org/10.1016/j.jenvman.2021.112870.

Gerwing, J.J. and Uhl, C. (2002). Pre-logging liana cutting reduces liana regeneration in logging gaps in the eastern Brazilian Amazon. Ecological Applications, 12(6), 1642-1651. https://doi.org/10.1890/1051-0761(2002)012[1642:PLLCRL]2.0.CO;2.

González, T. M., González-Trujillo, J.D., Palmer, J.R.B., Pino, J., and Armenteras, D. (2017) Movement behavior of a tropical mammal: The case of Tapirus terrestris. Ecological Modelling. 360: 223–229. https://doi.org/10.1016/j.ecolmodel.2017.07.006

Greene, D.F., and Johnson, E.A. (1995). Long-distance wind dispersal of tree seeds. Canadian Journal of Botany, 73(7), 1036-1045. https://doi.org/10.1139/b95-113.

Hallema, D.W., Robinne F-N. and Bladon, K.D. (2018). Reframing the challenge of global wildfire threats to water supplies. Earth's Future, 6(6), 772-776. https://doi.org/10.1029/2018EF000867.

Haslem A., Kelly L.T., Nimmo D.G., Watson, S.J., Kenny, S.A., Taylor, R.S. et al. (2011). Habitat or fuel? Implications of long-term, post-fire dynamics for the development of key resources for fauna and fire. Journal of Applied Ecology, 48(1), 247-256. https://doi.org/10.1111/j.1365-2664.2010.01906.x.

Hoekstra, A.Y., Buurman, J. and Van Ginkel, K.C.H. (2018). Urban water security: a review. Environmental Research Letters, 13(5), 053002. https://doi.org/10.1088/1748-9326/aaba52.

Hohner, A.K., Rhoades, C.C., Wilkerson, P. and Rosario-Ortiz, F.L. (2019). Wildfires alter forest watersheds and threaten drinking water quality. Accounts of Chemical Research, 52(5), 1234-1244. https://doi.org/10.1021/acs.accounts.8b00670.

Hohner, A.K., Terry, L.G., Townsend, E.B., Summers, R.S. and Rosario-Ortiz, F.L. (2017). Water treatment process evaluation of wildfire-affected sediment leachates. Environmental Science: Water Research and Technology, 3(2), 352-365. https://doi.org/10.1039/c6ew00247a.

Hradsky, B.A., Mildwaters, C., Ritchie, E.G., Christie, F. and Di Stefano, J. (2017). Responses of invasive predators and native prey to a prescribed forest fire. Journal of Mammalogy, 98(3), 835-847, https://doi.org/10.1093/jmammal/gyx010.

Hughes, F., Vitousek, P.M. and Tunison, T. (1991). Alien grass invasion and fire in the seasonal submontane zone of Hawai'i. Ecology, 72(2), 743-747. https://doi.org/10.2307/2937215.

Huston, M.A. (1994). Biological Diversity: The Coexistence of

Species on Changing Landscapes. Cambridge: Cambridge University Press.

Huston, M.A. (2003). Understanding the effects of fire and other mortality-causing disturbances on species diversity. In Fire in South-Western Australian forests: Impacts and Management. Burrows, N.C. and Abbott, I. (eds.). Leiden: Backhuys Publishers. 51-84.

Intergovernmental Panel on Climate Change. (2013). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M.M.B., Allen, S.K., Boschung, J., Nauels, A, Xia, Y., Bex, V. and Midgley, P.M. (eds.) Cambridge: Cambridge University Press. https://www.ipcc.ch/site/assets/uploads/2018/02/WG1AR5_all_final.pdf.

Jackson, W.D. (1968). Fire, air, water and earth: an elemental ecology of Tasmania. Proceedings of the Ecological Society of Australia, 3, 9-16.

Jenkins, M. and Schaap, B. (2018). Untapped Potential: Forest Ecosystem Services for Achieving SDG 15 UNFF13 Background Analytical Study. United Nations Forum on Forests. https://www.un.org/esa/forests/wp-content/uploads/2018/05/UNFF13_BkgdStudy_ForestsEcoServices.pdf.

Johnstone, J.F., Allen, C.D., Franklin, J.F., Frelich, L.E., Harvey, B.J., Higuera, P.E. et al. (2016). Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment, 14(7), 369-378. https://doi.org/10.1002/fee.1311.

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

Jones, G.M., Kramer, H.A., Whitmore, S.A., Berigan, W.J., Tempel, D.J., Wood, C.M. et al. (2020). Habitat selection by spotted owls after a megafire reflects their adaptation to historical frequent-fire regimes. Landscape Ecology, 35(5), 1199-1213. https://doi.org/10.1007/s10980-020-01010-y.

Kasischke, E.S. and Turetsky M.R. (2006). Recent changes in the fire regime across the North American boreal region: spatial and temporal patterns of burning across Canada and Alaska. Geophysical Research Letters, 33(9), L09703. https://doi.org/10.1029/2006GL025677.

Keeley, J.E. (1987). Role of fire in seed germination of woody taxa in California chaparral. Ecology, 68(2), 434-443. http://dx.doi.org/10.2307/1939275.

Keeley, J.E. and Bond, W.J. (1997). Convergent seed germination in South African fynbos and Californian chaparral. Plant Ecology, 133(2), 153-167. https://doi.org/10.1023/A:1009748603202.

Keeley, J.E., Bond, W.J., Bradstock, R.A., Pausas, J.G. and Rundel, P.W. (2012). Fire in Mediterranean Ecosystems: Ecology, Evolution and Management. Cambridge: Cambridge University Press.

Keeley, J.E., Fotheringham, C.J. and Morais M. (1999). Reexamining fire suppression impacts on brushland fire regimes. Science, 284(5421), 1829-1832. https://doi.org/10.1126/science.284.5421.1829.

Keeley, J.E. and Safford H.D. (2016). Fire as an ecosystem process. In Ecosystems of California. Mooney, H. and Zavaleta, E. (eds.). Berkeley, CA: University of California Press. Chapter 3. 27-45.

Keith, H., Mackey, B.G. and Lindenmayer, D.B. (2009). Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. Proceedings of the National Academy of Sciences of the United States of America, 106(28), 11635-11640. https://doi.org/10.1073/pnas.0901970106.

Kelly, L.T., Nimmo, D.G., Spence-Bailey, L.M., Taylor, R.S., Watson, S.J., Clarke, M.F. et al. (2012). Managing fire mosaics for small mammal conservation: a landscape perspective. Journal of Applied Ecology, 49(2), 412-421. https://doi.org/10.1111/j.1365-2664.2012.02124.x.

Kharuk, V.I., Ranson, K.J., Dvinskaya, M.L. and Im, S.T. (2011). Wildfires in northern Siberian larch dominated communities. Environmental Research Letters, 6(4), 045208. https://doi.org/10.1088/1748-9326/6/4/045208.

Krylov, A., McCarty, J.L., Potapov, P., Loboda, T., Tyukavina, A., Turubanova, S. et al. (2014). Remote sensing estimates of stand-replacement fires in Russia, 2002–2011. Environmental Research Letters, 9(10), 105007. https://doi.org/10.1088/1748-9326/9/10/105007.

Kumar, P., Chen, H.Y.H., Thomas, S.C. and Shahi, C. (2018). Linking resource availability and heterogeneity to understorey species diversity through succession in boreal forest of Canada. Journal of Ecology, 106(3), 1266-1276. https://doi.org/10.1111/1365-2745.12861.

Kuusela, K. (1992). The boreal forests: an overview. Unasylva, 43(179), 3-13.

La Roi, G.H. (1967). Ecological studies in the boreal spruce-fir forests of the North American taiga. I. Analysis of the vascular flora. Ecological Monographs. 37(3), 229-253.

Laboratório de Aplicações de Satélites Ambientais (2020). Sistema ALARMES. https://lasa.ufrj.br/alarmes/.

Landres, P.B., Morgan, P. and Swanson, F.J. (1999). Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications, 9(4), 1179-1188. https://doi.org/10.1890/1051-0761(1999)009[1179:OOTUON]2.0.CO;2.

Langlands, P.R., Brennan, K.E.C., Framenau, V.W. and Main, B.Y. (2011). Predicting the post-fire responses of animal assemblages: testing a trait-based approach using spiders. Journal of Animal Ecology, 80(3), 558-568. https://doi.org/10.1111/j.1365-2656.2010.01795.x.

Libonati, R., Belém, L.B.C., Rodrigues, J.A., Santos, F.L.M., Sena, C.A.P., Pinto, M.M. et al. (2020a). Sistema ALARMES – Alerta de área queimada Pantanal, situação atual – primeira semana de outubro de 2020. Rio de Janeiro: Laboratório de Aplicações de Satélites Ambientais. https://doi.org/10.13140/RG.2.2.34755.63526.

Libonati, R., Belém, L.B.C., Rodrigues, J.A., Santos, F.L.M., Sena, C.A.P., Pinto, M.M. et al. (2021). Sistema ALARMES – Alerta de área queimada Pantanal, situação final de 2020. Nota Técnica 01/2021. Rio de Janeiro: Laboratório de Aplicações de Satélites Ambientais. https://doi.org/10.13140/RG.2.2.11629.51687.

Libonati, R., DaCamara, C.C., Peres, L.F., de Carvalho, L.A.S. and Garcia, C.L. (2020b). Rescue Brazil's burning Pantanal wetlands. Nature, 588, 217-219. https://doi.org/10.1038/d41586-020-03464-1.

Litt, A.R. and Steidl, R.J. (2011). Interactive effects of fire and nonnative plants on small mammals in grasslands. Wildlife Monographs, 176(1), 1-31. https://doi.org/10.1002/wmon.2.

Liu, Z., Yang, J., Chang, Y., Weisberg, P.J. and He, H.S. (2012). Spatial patterns and drivers of fire occurrence and its future trend under climate change in a boreal forest of Northeast China. Global Change Biology, 18(6), 2041-2056. https://doi.org/10.1111/j.1365-2486.2012.02649.x.

Loranty, M.M., Berner, L.T., Goetz, S.J., Jin, Y. and Randerson, J.T. (2014). Vegetation controls on northern high latitude snow-albedo feedback: observations and CMIP 5 model simulations. Global Change Biology, 20(2), 594-606. https://doi.org/10.1111/gcb.12391.

Ma, J., Bu, R., Liu, M., Chang, Y., Han, F., Qin, Q. et al. (2016). Recovery of understory vegetation biomass and biodiversity in burned larch boreal forests in Northeastern China. Scandinavian Journal of Forest Research, 31(4), 382-393. https://doi.org/10.1080/02827581.2015.1072238.

Marengo, J.A., Cunha, A.P., Cuartas, L.A., Deusdará Leal, K.R., Broedel, E., Seluchi, M.E. et al. (2021). Extreme Drought in the Brazilian Pantanal in 2019–2020: characterization, causes, and impacts. Frontiers in Water, 3, 639204. https://doi.org/10.3389/frwa.2021.639204.

Matveeva T.A. and Matveev A.M. (2008). Role of forest density during the post-fire reforestation. Bulletin of the Altai State Agricultural University. 10(48), 24–28. [In Russian.]

Mega, E.R. (2020). 'Apocalyptic' fires are ravaging the world's largest tropical wetland. Nature, 586, 20-21. https://doi.org/10.1038/d41586-020-02716-4.

Miller, J.E.D., Root, H.T. and Safford H.D. (2018). Altered fire regimes cause long-term lichen diversity losses. Global Change Biology, 24(10), 4909-4918. https://doi.org/10.1111/gcb.14393.

Miller, J.E.D. and Safford, H.D. (2020). Are plant community responses to wildfire contingent upon historical disturbance regimes? Global Ecology and Biogeography, 29(10), 1621-1633. https://doi.org/10.1111/geb.13115.

Mollicone, D., Achard, F., Marchesini, L.B., Federici, S., Wirth, C., Leipold, M. et al. (2002). A remote sensing based approach to determine forest fire cycle: case study of the Yenisei Ridge dark taiga. Tellus B: Chemical and Physical Meteorology, 54(5), 688-695. https://doi.org/10.3402/tellusb.v54i5.16714.

Moody, J.A., Ebel, B.A., Nyman, P., Martin, D.A., Stoof, C. and McKinley, R. (2015). Relations between soil hydraulic properties and burn severity. International Journal of Wildland Fire, 25(3), 279-293. https://doi.org/10.1071/WF14062.

Mowat, E.J., Webb, J.K. and Crowther, M.S. (2015). Fire-mediated niche-separation between two sympatric small mammal species. Austral Ecology, 40(1), 50-59. https://doi.org/10.1111/aec.12166.

Murphy, S.F., McCleskey, R.B., Martin, D.A., Holloway, J.M. and Writer, J.H. (2020). Wildfire-driven changes in hydrology mobilize arsenic and metals from legacy mine waste. Science of the Total Environment, 743, 140635. https://doi.org/10.1016/j.scitotenv.2020.140635.

Navarro, K.M., Cisneros, R., Noth, E.M., Balmes, J.R. and Hammond, S.K. (2017). Occupational exposure to polycyclic aromatic hydrocarbon of wildland firefighters at prescribed and wildland fires. Environmental Science and Technology, 51(11), 6461-6469. https://doi.org/10.1021/acs.est.7b00950.

Navarro, K.M., Kleinman, M.T., Mackay, C.E., Reinhardt, T.E., Balmes, J.R., Broyles, G.A. et al. (2019). Wildland firefighter smoke exposure and risk of lung cancer and cardiovascular disease mortality. Environmental Research, 173, 462-468. https://doi.org/10.1016/j.envres.2019.03.060.

Niemeyer, R.J., Bladon, K.D. and Woodsmith, R.D. (2020). Long-term hydrologic recovery after wildfire and post-fire forest management in the interior Pacific Northwest. Hydrological Processes, 34(5), 1182-1197. https://doi.org/10.1002/hyp.13665.

Nilsson, M.C. and Wardle, D.A. (2005). Understory vegetation as a forest ecosystem driver: evidence from the northern Swedish boreal forest. Frontiers in Ecology and the Environment, 3(8), 421-428. https://doi.org/10.1890/1540-9295(2005)003[0421:UVAAFE]2.0.CO;2.

Nimmo, D.G., Avitabile, S., Banks, S.C., Bliege Bird, R., Callister, K., Clarke, M.F. et al. (2019). Animal movements in fire-prone landscapes. Biological Reviews, 94(3), 981-998. https://doi.org/10.1111/brv.12486.

Nolan, R.H., Boer, M.M., Collins, L., Resco de Dios, V., Clarke, H., Jenkins, M. et al. (2020). Causes and consequences of eastern Australia's 2019–20 season of mega-fires. Global Change Biology, 26(3), 1039-1041. https://doi.org/10.1111/gcb.14987.

Oliver, A.A. Bogan, M.T., Herbst, D.B. and Dahlgren, R.A. (2012). Short-term changes in-stream macroinvertebrate communities following a severe fire in the Lake Tahoe basin, California. Hydrobiologia, 694(1), 117-130. https://doi.org/10.1007/s10750-012-1136-7.

Pausas, J.G. (2019). Generalized fire response strategies in plants and animals. Oikos, 128(2), 147-153. https://doi.org/10.1111/oik.05907.

Pausas, J.G. and Keeley, J.E. (2014). Abrupt climate-independent fire regime changes. Ecosystems, 17(6), 1109-1120. https://doi.org/10.1007/s10021-014-9773-5.

Penman, T.D. and Towerton, A.L. (2008). Soil temperatures during autumn prescribed burning: implications for the germination of fire responsive species? International Journal of Wildland Fire, 17(5), 572-578. https://doi.org/10.1071/WF07092.

Peres, C.A. (1999). Ground fires as agents of mortality in a Central Amazonian forest. Journal of Tropical Ecology, 15(4), 535-541. http://doi.org/10.1017/S0266467499000991.

Peterson, D.W. and Reich, P.B. (2001). Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. Ecological Applications, 11(3), 914-927. http://doi.org/10.2307/3061125.

Pickett, S. and White, P. (eds.) (1985). The Ecology of Natural Disturbance and Patch Dynamics. San Diego, CA: Academic Press.

Pingree, M.R.A. and Kobziar, L.N. (2019). The myth of the biological threshold: a review of biological responses to soil heating associated with wildland fire. Forest Ecology and Management, 432, 1022-1029. https://doi.org/10.1016/j.foreco.2018.10.032.

Ponomarev, E.I., Kharuk, V.I. and Ranson, K.J. (2016). Wildfire dynamics in Siberian larch forests. Forests, 7(6), 125. https://doi.org/10.3390/f7060125.

Proctor, C.R., Lee, J., Yu, D., Shah, A.D. and Whelton, A.J. (2020). Wildfire caused widespread drinking water distribution network contamination. AWWA Water Science, 2(4), e1183.
https://doi.org/10.1002/aws2.1183.

Richter, C., Rejmánek, M., Miller, J.E., Welch, K.R., Weeks, J. and Safford, H.D. (2019). The species diversity× fire severity relationship is hump-shaped in semiarid yellow pine and mixed conifer forests. Ecosphere, 10(10), e02882. https://doi.org/10.1002/ecs2.2882.

Robichaud, P.R., Wagenbrenner, J.W., Pierson, F.B., Spaeth, K.E., Ashmun, L.E. and Moffet, C.A. (2016). Infiltration and interrill erosion rates after a wildfire in western Montana, USA. Catena, 142, 77-88. https://doi.org/10.1016/j.catena.2016.01.027.

Robinne, F.-N., Bladon, K.D., Miller, C., Parisien, M.A., Mathieu, J. and Flannigan, M.D. (2018). A spatial evaluation of global wildfire-water risks to human and natural systems. Science of the Total Environment, 610-611, 1193-1206. https://doi.org/10.1016/j.scitotenv.2017.08.112.

Robinne, F.-N., Hallema, D.W., Bladon, K.D. and Buttle, J.M. (2020). Wildfire impacts on hydrologic ecosystem services in North American high-latitude forests: a scoping review. Journal of Hydrology, 581, 124360. https://doi.org/10.1016/j.jhydrol.2019.124360.

Robinne, F.-N., Hallema, D.W., Bladon, K.D., Flannigan, M.D., Boisramé, G., Bréthaut, C.M. et al. (2021). Scientists' warning on extreme wildfire risks to water supply. Hydrological Processes, 35(5), e14086. https://doi.org/10.1002/hyp.14086.

Robinson, N.M., Leonard, S.W.J., Ritchie, E.G., Bassett, M., Chia, E.K., Buckingham, S. et al. (2013). Refuges for fauna in fire-prone landscapes: their ecological function and importance. Journal of Applied Ecology, 50(6), 1321-1329. https://doi.org/10.1111/1365-2664.12153.

Roznik, E.A. and Reichling, S.B. (2021). Survival, movements and habitat use of captive-bred and reintroduced dusky gopher frogs. Animal Conservation, 24(1), 51-63. https://doi.org/10.1111/acv.12599.

Safford, H.D. and Stevens, J.T. (2017). Natural Range of Variation (NRV) for Yellow Pine and Mixed Conifer Forests in the Sierra Nevada, Southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report PSW-GTR-256. Albany, CA: United States Department of Agriculture, Forest Service. https://doi.org/10.2737/PSW-GTR-256.

Safford, H.D. and Vallejo, V.R. (2019). Ecosystem management and ecological restoration in the Anthropocene: integrating global change, soils, and disturbance in boreal and Mediterranean forests. In Global Change and Forest soils: Conservation of a Finite Natural Resource. Busse, M., Giardina, C.P., Morris, D.M. and Page-Dumroese, D.S. (eds.). Cambridge, MA: Elsevier. Chapter 12. 259-308. https://doi.org/10.1016/B978-0-444-63998-1.00012-4.

Safford, H.D. and K.M. Van de Water. (2011). Using Fire Return Interval Departure (FRID) Analysis to Map Spatial and Temporal Changes in Fire Frequency on National Forest Lands in California. Research Paper PSW-RP-266, USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Sankey, J.B., Kreitler, J.R., Hawbaker, T.J., McVay, J.L., Miller, M.E., Mueller, E.R. et al. (2017). Climate, wildfire, and erosion ensemble foretells more sediment in western USA watersheds. Geophysical Research Letters, 44(17), 8884-8892. https://doi.org/10.1002/2017GL073979.

Santín, C. and Doerr, S.H. (2016). Fire effects on soils: the human dimension. Philosophical Transactions of the Royal Society B: Biological Sciences, 371(1696), 20150171. https://doi.org/10.1098/rstb.2015.0171.

Scheffer, M., Hirota, M., Holmgren, M., Van Nes, E.H. and Chapin III, F.S. (2012). Thresholds for boreal biome transitions. Proceedings of the National Academy of Sciences of the United States of America, 109(52), 21384-21389. https://doi.org/10.1073/pnas.1219844110.

Schulze, E.-D., Wirth, C., Mollicone, D. and Ziegler, W. (2005). Succession after stand replacing disturbances by fire, wind throw, and insects in the dark Taiga of Central Siberia. Oecologia, 146, 77-88. https://doi.org/10.1007/s00442-005-0173-6.

Selwood, K.E. and Zimmer, H.C. (2020). Refuges for biodiversity conservation: a review of the evidence. Biological Conservation, 245, 108502. https://doi.org/10.1016/j.biocon.2020.108502.

Shakesby, R.A. (2011). Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth Science Reviews, 105(3-4), 71-100. https://doi.org/10.1016/j.earscirev.2011.01.001.

Shakesby, R.A. and Doerr, S.H. (2006). Wildfire as a hydrological and geomorphological agent. Earth Science Reviews, 74(3-4), 269-307. https://doi.org/10.1016/j.earscirev.2005.10.006.

Shlisky, A., Alencar, A.A.C., Nolasco, M.M. and Curran, L.M. (2009). Overview: global fire regime conditions, threats, and opportunities for fire management in the tropics. In Tropical Fire Ecology. Cochrane, M.A. (ed.). Berlin: Springer. 65-83.

Shorohova, E., Kuuluvainen, T., Kangur, A. and Jõgiste, K. (2009). Natural stand structures, disturbance regimes and successional dynamics in the Eurasian boreal forests: a review with special reference to Russian studies. Annals of Forest Science, 66(2), 1-20. https://doi.org/10.1051/forest/2008083.

Shvidenko, A.Z. and Schepaschenko, D.G. (2013). Climate change and wildfires in Russia. Contemporary Problems of Ecology, 6, 683-692. https://doi.org/10.1134/S199542551307010X.

Silva, C.V.J., Aragão, L.E.O.C., Young, P.J., Espirito-Santo, F., Berenguer, E., Anderson, L.O. et al. (2020a). Estimating the multi-decadal carbon deficit of burned Amazonian forests. Environmental Research Letters, 15(11), 114023. https://doi.org/10.1088/1748-9326/abb62c.

Silva, L.G.M., Doyle, K.E., Duffy, D., Humphries, P., Horta, A. and Baumgartner, L.J. (2020b). Mortality events resulting from Australia's catastrophic fires threaten aquatic biota. Global Change Biology, 26(10), 5345-5350. https://doi.org/10.1111/gcb.15282.

Silva Junior, C.H.L., Aragão, L.E.O.C., Fonseca, M.G., Almeida, C.T., Vedovato, L.B. and Anderson, L.O. (2018). Deforestation-induced fragmentation increases forest fire occurrence in central Brazilian Amazonia. Forests, 9(6), 305. https://doi.org/10.3390/f9060305.

Silveira, M.V., Petri, C.A., Broggio, I.S., Chagas, G.O., Macul, M.S., Leite, C.C.S.S. et al. (2020). Drivers of fire anomalies in the Brazilian Amazon: lessons learned from the 2019 fire crisis. Land, 9(12), 516. https://doi.org/10.3390/land9120516.

Silvério, D.V., Brando, P.M., Balch, J.K., Putz, F.E., Nepstad, D.C., Oliveira-Santos, C. et al. (2013). Testing the Amazon savannization hypothesis: fire effects on invasion of a neotropical forest by native cerrado and exotic pasture grasses. Philosophical Transactions of the Royal Society B: Biological Sciences, 368(1619), 20120427. https://doi.org/10.1098/rstb.2012.0427.

Sofronov, M.A. and Volokitina, A.V. (2010). Wildfire ecology in continuous permafrost zone. In Permafrost Ecosystems: Siberian Larch Forests. Osawa, A., Zyryanova, O.A., Matsuura, Y., Kajimoto, T. and Wein, R.W. (eds.). Dordrecht: Springer. 59-82.

Springgay, E., Casallas Ramirez, S., Janzen, S. and Brito. V.V. (2019). The Forest–Water Nexus: an international perspective. Forests, 10(10), 915. https://doi.org/10.3390/f10100915.

Steel, Z.L., Campos, B., Frick, W.F., Burnett, R. and Safford, H.D. (2019). The effects of wildfire severity and pyrodiversity on bat occupancy and diversity in fire-suppressed forests. Scientific Reports, 9(1), 16300. https://doi.org/10.1038/s41598-019-52875-2.

Steel, Z.L., Fogg, A.M., Burnett, R., Roberts, L.J. and Safford, H.D. (2021). When bigger isn't better: implications of large high-severity wildfire patches for avian diversity and community composition. Diversity and Distributions, Early View. https://doi.org/10.1111/ddi.13281.

Steel, Z.L., Safford, H.D. and Viers, J.H. (2015). The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere, 6(1), 1-23. https://doi.org/10.1890/ES14-00224.1.

Stillman, A.N., Lorenz, T.J., Fischer, P.C., Siegel, R.B., Wilkerson, R.L., Johnson, M. et al. (2021). Juvenile survival of a burned forest specialist in response to variation in fire characteristics. Journal of Animal Ecology, 90(5), 1317-1327. https://doi.org/10.1111/1365-2656.13456.

Swain, D.L., Singh, D., Touma, D. and Diffenbaugh, N.S. (2020). Attributing extreme events to climate change: a new frontier in a warming world. One Earth, 2(6), 522-527. https://doi.org/10.1016/j.oneear.2020.05.011.

Swanson, F.J. (1981). Fire and geomorphic processes. In Fire Regime and Ecosystem Properties. General Technical Report. Mooney, H.A., Bonnicksen, T.M., Christensen, N.L., Lotan, J.E. and Reiners, W.A. (eds.). Washington. DC: United States Department of Agriculture, Forest Service. 401-444. http://www.fsl.orst.edu/rna/Documents/publications/Fire%20and%20geomorphic%20processes%201981.pdf.

Tautenhahn, S., Lichstein, J.W., Jung, M., Kattge, J., Bohlman, S.A., Heilmeier, H. et al. (2016). Dispersal limitation drives successional pathways in Central Siberian forests under current and intensified fire regimes. Global Change Biology, 22(6), 2178-2197. https://doi.org/10.1111/gcb.13181.

Throop, H.L., Salem, M.A. and Whitford, W.G. (2017). Fire enhances litter decomposition and reduces vegetation cover influences on decomposition in a dry woodland. Plant Ecology, 218(7), 799-811. https://doi.org/10.1007/s11258-017-0730-1.

Tulloch, A.I., Pichancourt, J.B., Gosper, C.R., Sanders, A. and Chadès, I. (2016). Fire management strategies to maintain species population processes in a fragmented landscape of fire-interval extremes. Ecological Applications, 26(7), 2175-2189. https://doi.org/10.1002/eap.1362.

Underwood, E.C., Safford, H.D., Molinari, N.A. and Keeley, J.E. (eds.) (2018). Valuing Chaparral: Ecological, Socio-Economic, and Management Perspectives. Springer Series on Environmental Management. Cham: Springer International Publishing.

United Nations Economic Commission for Europe and Food and Agriculture Organization of the United Nations (2018). Forests and Water: Valuation and Payments for Forest Ecosystem Services. Geneva Timber and Forest Study Papers Series. Geneva: United Nations. https://doi.org/10.18356/bf32044a-en.

Van Eeden, L., Nimmo, D., Mahony, M., Herman, K., Ehmke, G., Driessen, J. et al. (2020). Australia's 2019-2020 Bushfires: The Wildlife Toll. Sydney: World Wildlife Fund. https://www.wwf.org.au/what-we-do/bushfire-recovery/in-depth/resources/australia-s-2019-2020-bushfires-the-wildlife-toll#gs.d60m12.

Van Langevelde, F., Van De Vijver, C.A.D.M., Kumar, L., Van De Koppel, J., De Ridder, N., Van Andel, J. et al. (2003). Effects of fire and herbivory on the stability of savanna ecosystems. Ecology, 84(2), 337-350. https://doi.org/10.1890/0012-9658(2003)084[0337:EOFAHO]2.0.CO;2.

Van Wagtendonk, J.W., Sugihara, N.G., Stephens, S.L., Thode, A.E., Shaffer, K.E. and Fites-Kaufman, J. (eds.) (2018). Fire in California's Ecosystems. Second edition. Berkeley, CA: University of California Press.

Volokitina, A.V. (2015). Methodical aspects for describing post-fire forest site (in Russian). Vestnik Tomskogo gosudarstvennogo universiteta. Biologiya [Tomsk State University Journal of Biology], 3, 84-98.

Ward, M., Tulloch, A.I., Radford, J.Q., Williams, B.A., Reside, A.E., Macdonald, S.L. et al. (2020). Impact of 2019–2020 mega-fires on Australian fauna habitat. Nature Ecology and Evolution, 4(10), 1321-1326. https://doi.org/10.1038/s41559-020-1251-1.

Welch, K.R., Safford, H.D. and Young, T.P. (2016). Predicting conifer establishment post wildfire in mixed conifer forests of the North American Mediterranean-climate zone. Ecosphere, 7(12), e01609. https://doi.org/10.1002/ecs2.1609.

White, A.M., Manley, P.N., Tarbill, G.L., Richardson, T.W., Russell, R.E., Safford, H.D. et al. (2016). Avian community responses to post-fire forest structure: implications for fire management in mixed conifer forests. Animal Conservation, 19(3), 256-264. https://doi.org/10.1111/acv.12237.

Whitman, E., Parisien, M.A., Thompson, D.K. and Flannigan, M.D. (2019). Short-interval wildfire and drought overwhelm boreal forest resilience. Scientific Reports, 9(1), 18796.

https://doi.org/10.1038/s41598-019-55036-7.

Wintle, B.A., Legge, S. and Woinarski, J.C.Z. (2020). After the Megafires: What Next for Australian Wildlife? Trends in Ecology & Evolution. 35, 753-757. https://doi.org/10.1016/j.tree.2020.06.009

Wittenberg, L. (2012). Post-fire soil ecology: properties and erosion dynamics. Israel Journal of Ecology and Evolution, 58(2-3), 151-164. https://doi.org/10.1560/IJEE.58.2-3.151.

Zedler, P.H., Gautier, C.R. and McMaster, G.S. (1983). Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. Ecology, 64(4), 809-818. https://doi.org/10.2307/1937204.

Zhang, Y., Chen, H.Y.H. and Taylor, A.R. (2017). Positive species diversity and above-ground biomass relationships are ubiquitous across forest strata despite interference from overstorey trees. Functional Ecology, 31(2), 419-426. https://doi.org/10.1111/1365-2435.12699.

Zituni, R., Wittenberg, L. and Malkinson, D. (2019). The effects of post-fire forest management on soil erosion rates 3 and 4 years after a wildfire, demonstrated on the 2010 Mount Carmel fire. International Journal of Wildland Fire, 28(5), 377-385. https://doi.org/10.1071/WF18116.

Zyryanova, O.A., Yaborov, V.T., Tchikhacheva, T.L., Koike, T., Kobayashi, M., Matsuura, Y. et al. (2007). The structure and biodiversity after fire disturbance in Larix gmelinii (Rupr.) Rupr. forest, northeastern Asia. Eurasian Journal of Forest Research, 10(1), 19-29. http://hdl.handle.net/2115/24482.

Chapter 5

Abt, K.L., Butry, D.T., Prestemon, J.P. and Scranton, S. (2015). Effect of fire prevention programs on accidental and incendiary wildfires on tribal lands in the United States. International Journal of Wildland Fire, 24(6), 749-762. https://doi.org/10.1071/WF14168.

Agee, J.K. and Skinner, C.N. (2005). Basic principles of forest fuel reduction treatments. Forest Ecology and Management, 211(1-2), 83-96. https://doi.org/10.1016/j.foreco.2005.01.034.

Altangerel, K. and Kull, C.A. (2013). The prescribed burning debate in Australia: conflicts and compatibilities. Journal of Environmental Planning and Management, 56(1), 103-120. https://doi.org/10.1080/09640568.2011.652831.

Anderson, L.O., Marchezini, V., Morello, T.F. and Cunningham, C.A. (2019). Conceptual model of disaster risk management and warning system associated with wildfires and public policy challenges in Brazil. Territorium, 26(I), 43-61. https://doi.org/10.14195/1647-7723_26-1_4.

Arab, A., Khodaei, A., Eskandarpour, R., Thompson, M.P. and Wei, Y. (2021). Three lines of defence for wildfire risk management in electric power grids: a review. IEEE Access, 9, 61577-61593. https://doi.org/10.1109/ACCESS.2021.3074477.

Archibald, S., Staver, A.C. and Levin, S.A. (2012). Evolution of human-driven fire regimes in Africa. Proceedings of the National Academy of Sciences of the United States of America, 109(3), 847-852. https://doi.org/10.1073/pnas.1118648109.

Badan Nasional Penanggulangan Bencana (2020). Statistik Bencana di Tanjung Jabung Barat [Disaster Statistics in Tanjung Jabung Barat]. Tungkal City.

Barnett, K., Parks, S.A., Miller, C. and Naughton, H.T. (2016). Beyond fuel treatment effectiveness: characterizing interactions between fire and treatments in the US. Forests, 7(10), 237. https://doi.org/10.3390/f7100237.

Barros, A.M.G., Ager, A.A., Day, M.A., Krawchuk, M.A. and Spies, T.A. (2018). Wildfires managed for restoration enhance ecological resilience. Ecosphere, 9(3), e02161. https://doi.org/10.1002/ecs2.2161.

Becker, W.C., Hohner, A., Rosario-Ortiz, F. and DeWolfe, J. (2018). Preparing for wildfires and extreme weather: plant design and operation recommendations. Journal of the American Water Works Association, 110(7), 32-40. https://doi.org/10.1002/awwa.1113.

Bedsworth, L., Cayan, D., Franco, G., Fisher, L. and Ziaja, S. (2018). Statewide Summary Report. California's Fourth Climate Change Assessment. Publication number: SUMCCCA4-2018-013. California Governor's

Office of Planning and Research, Scripps Institution of Oceanography, California Energy Commission, California Public Utilities Commission.

Bhaduri, A., Bogardi, J., Siddiqi, A., Voigt, H., Vörösmarty, C., Pahl-Wostl, C. et al. (2016). Achieving Sustainable Development Goals from a water perspective. Frontiers in Environmental Science, 4, 64. https://doi.org/10.3389/fenvs.2016.00064.

Bilbao, B.A., Leal, A.V. and Méndez, C.L. (2010). Indigenous use of fire and forest loss in Canaima National Park, Venezuela: assessment of and tools for alternative strategies of fire management in Pemón indigenous lands. Human Ecology, 38(5), 663-673. https://doi.org/10.1007/s10745-010-9344-0.

Bilbao, B.A., Mistry, J., Millán, A. and Berardi, A. (2019). Sharing multiple perspectives on burning: towards a participatory and intercultural fire management policy in Venezuela, Brazil, and Guyana. Fire, 2(3), 39. https://doi.org/10.3390/fire2030039.

Bilbao, B.A., Steil, L., Urbieta, I.R., Anderson, L., Pinto, C., González, M.E. et al. (2020). Wildfires. In Adaptation to Climate Change Risks in Ibero-American Countries: RIOCCADAPT Report. Moreno, J.M., Laguna-Defior, C., Barros, V., Calvo Buendía, E., Marengo, J.A. and Oswald Spring, U. (eds.). Madrid: McGraw Hill. Chapter 12. 435-496.

Blanchi, R., Leonard, J., Haynes, K., Opie, K., James, M. and de Oliveira, F.D. (2014). Environmental circumstances surrounding bushfire fatalities in Australia 1901-2011. Environmental Science and Policy, 37, 192-203. https://doi.org/10.1016/j.envsci.2013.09.013.

Boer, M.M., Sadler, R.J., Wittkuhn, R.S., McCaw, L. and Grierson, P.F. (2009). Long-term impacts of prescribed burning on regional extent and incidence of wildfires: evidence from 50 years of active fire management in SW Australian forests. Forest Ecology and Management, 259(1), 132-142. https://doi.org/10.1016/j.foreco.2009.10.005.

Boisramé, G.F.C., Thompson, S.E., Tague, C.N. and Stephens, S.L. (2019). Restoring a natural fire regime alters the water balance of a Sierra Nevada catchment. Water Resources Research, 55(7), 5751-5769. https://doi.org/10.1029/2018WR024098.

Bowman, D.M.J.S., Murphy, B.P., Boer, M.M., Bradstock, R.A., Cary, G.J., Cochrane, M.A. et al. (2013). Forest fire management, climate change, and the risk of catastrophic carbon losses. Frontiers in Ecology and the Environment, 11(2), 66-67. https://doi.org/10.1890/13.WB.005.

Bowman, D.M.J.S., Walsh, A. and Prior, L.D. (2004). Landscape analysis of Aboriginal fire management in Central Arnhem Land, north Australia. Journal of Biogeography, 31(2), 207-223. https://doi.org/10.1046/j.0305-0270.2003.00997.x.

Budiningsih, K., Setiabudi, I.M. and Septiani, A.D. (2020). Fire care community development in Batanghari District and Tanjung Jabung Timur District in Jambi Province: an overview. IOP Conference Series: Earth and Environmental Science, 487(1), 012020. https://doi.org/10.1088/1755-1315/487/1/012020.

Burkhardt, J., Bayham, J., Wilson, A., Berman, J.D., O'Dell, K., Ford, B. et al. (2020). The relationship between monthly air pollution and violent crime across the United States. Journal of Environmental Economics and Policy, 9(2), 188-205. https://doi.org/10.1080/21606544.2019.1630014.

Burton, I., Kates, R.W. and White, G.F. (1993). The Environment as Hazard. Second Edition. New York: Guilford Press.

Calkin, D.C., Finney, M.A., Ager, A.A., Thompson, M.P. and Gebert, K.M. (2011). Progress towards and barriers to implementation of a risk framework for US federal wildland fire policy and decision making. Forest Policy and Economics, 13(5), 378-389. https://doi.org/10.1016/j.forpol.2011.02.007.

Calkin, D.E., Thompson, M.P. and Finney, M.A. (2015). Negative consequences of positive feedbacks in US wildfire management. Forest Ecosystems, 2(1), 9. https://doi.org/10.1186/s40663-015-0033-8.

Castillo, M.E., Molina, J.R., Rodríguez y Silva, F., García-Chevesich, P. and Garfias, R. (2017). A system to evaluate fire impacts from simulated fire behavior in Mediterranean areas of Central Chile. Science of the Total Environment, 579, 1410-1418. https://doi.org/10.1016/j.scitotenv.2016.11.139

Castillo Soto, M., Garay Moena, R.M., Tapia Zarricueta, R., Garfias, R. and Orell Arenas, M. (2020). Metodología de evaluación de infraestructuras críticas en zonas de riesgo de incendios forestales [Evaluation Methodology for Critical Infrastructure in Areas at Risk of Forest Fire]. Santiago de Chile: Universidad de Chile, Facultad de Ciencias Forestales y Conservación de la Naturaleza. http://repositorio.uchile.cl/handle/2250/173421.

Christianson, A.C., Caverley, N., Diabo, D., Ellsworth, K., Highway, B., Joe, J. et al. (2020). Blazing the Trail: Celebrating Indigenous Fire Stewardship. Edmonton: FireSmart Canada.

Cohen, J.D. (2000). Preventing disaster: home ignitability in the wildland-urban interface. Journal of Forestry, 98(3), 15-21. https://doi.org/10.1093/jof/98.3.15.

Collins, K.M., Price, O.F. and Penman, T.D. (2015). Spatial patterns of wildfire ignitions in south-eastern Australia. International Journal of Wildland Fire, 24(8), 1098-1108. https://doi.org/10.1071/WF15054.

Commonwealth Scientific and Industrial Research Organisation (2020). Climate and Disaster: Technical Report, July 2020. https://www.csiro.au/en/Research/Environment/Extreme-Events/Bushfire/frontline-support/report-climate-disaste-resilience.

Cowan, J. (2019). What one old tower says about PG&E's lax safety culture. New York Times, 20 March. https://www.nytimes.com/2019/03/20/us/tower-pge-safety-culture-camp-fire.html. Accessed 7 October 2021.

Crippa, P., Castruccio, S., Archer-Nicholls, S., Lebron, G.B., Kuwata, M., Thota, A. et al. (2016). Population exposure to hazardous air quality due to the 2015 fires in Equatorial Asia. Scientific Reports, 6(1), 37074. https://doi.org/10.1038/srep37074.

Cruz, M.G. and Alexander, M.E. (2019). The 10% wind speed rule of thumb for estimating a wildfire's forward rate of spread in forests and shrublands. Annals of Forest Science, 76(2), 44. https://doi.org/10.1007/s13595-019-0829-8.

Cruz, M.G., Alexander, M.E., Fernandes, P.M., Kilinc, M. and Sil, Â. (2020). Evaluating the 10% wind speed rule of thumb for estimating a wildfire's forward rate of spread against an extensive independent set of observations. Environmental Modelling and Software, 133, 104818. https://doi.org/10.1016/j.envsoft.2020.104818.

Cui, X., Alam, M.A., Perry, G.L.W., Paterson, A.M., Wyse, S.V. and Curran, T.J. (2019). Green firebreaks as a management tool for wildfires: lessons from China. Journal of Environmental Management, 233, 329-336. https://doi.org/10.1016/j.jenvman.2018.12.043.

Curran, T.J., Perry, G.L.W., Wyse, S.V. and Alam, M.A. (2018). Managing fire and biodiversity in the wildland-urban interface: a role for green firebreaks. Fire, 1(1), 3. https://doi.org/10.3390/fire1010003.

Driscoll, D.A., Lindenmayer, D., Bennett, A.F., Bode, M., Bradstock, R.A., Cary, G.J. et al. (2010). Resolving conflicts in fire management using decision theory: asset-protection versus biodiversity conservation. Conservation Letters, 3(4), 215-223. https://doi.org/10.1111/j.1755-263X.2010.00115.x.

Eiser, J.R., Bostrom, A., Burton, I., Johnston, D.M., McClure, J., Paton, D. et al. (2012). Risk interpretation and action: a conceptual framework for responses to natural hazards. International Journal of Disaster Risk Reduction, 1, 5-16. https://doi.org/10.1016/j.ijdrr.2012.05.002.

Ellison, D., Morris, C.E., Locatelli, B., Sheil, D., Cohen, J., Murdiyarso, D. et al. (2017). Trees, forests and water: cool insights for a hot world. Global Environmental Change, 43, 51-61. https://doi.org/10.1016/j.gloenvcha.2017.01.002.

Eriksen, C. (2013). Gender and Wildfire: Landscapes of Uncertainty. New York, NY: Routledge.

Fairfax, E. and Whittle, A. (2020). Smokey the beaver: beaver-dammed riparian corridors stay green during wildfire throughout the western United States. Ecological Applications, 30(8), e02225. https://doi.org/10.1002/eap.2225.

Fernandes, P.M. (2015). Empirical support for the use of prescribed burning as a fuel treatment. Current Forestry Reports, 1(2), 118-127. https://doi.org/10.1007/s40725-015-0010-z.

Fernandes, P.M. and Botelho, H.S. (2003). A review of prescribed burning effectiveness in fire hazard reduction. International Journal of Wildland Fire, 12(2), 117-128. https://doi.org/10.1071/WF02042.

Finney, M., Grenfell, I.C. and McHugh, C.W. (2009). Modeling containment of large wildfires using generalized linear mixed-model analysis. Forest Science, 55(3), 249-255.

Firesticks Alliance Indigenous Corporation (2020). Homepage. https://www.firesticks.org.au/.

Florec, V., Burton, M., Pannell, D., Kelso, J. and Milne, G. (2019). Where to prescribe burn: the costs and benefits of prescribed burning close to houses. International Journal of Wildland Fire, 29(5), 440-458. https://doi.org/10.1071/WF18192.

Gannon, B.M., Wei, Y., MacDonald, L.H., Kampf, S.K., Jones, K.W., Cannon, J.B. et al. (2019). Prioritising fuels reduction for water supply protection. International Journal of Wildland Fire, 28(10), 785-803. https://doi.org/10.1071/WF18182.

Gannon, B.M., Wei, Y. and Thompson, M.P. (2020). Mitigating source water risks with improved wildfire containment. Fire, 3(3), 45. https://doi.org/10.3390/fire3030045.

Ganz, D. (2020). Coronavirus, climate change, forest fires: integrated fire management and people-centred policymaking, 30 April. www.recoftc.org/stories/coronavirus-climate-change-forest-fires-integrated-fire-management-and-people-centred. Accessed 5 October 2021.

Gibbons, P., Gill, A.M., Shore, N., Moritz, M.A., Dovers, S. and Cary, G.J. (2018). Options for reducing house-losses during wildfires without clearing trees and shrubs. Landscape and Urban Planning, 174, 10-17. https://doi.org/10.1016/j.landurbplan.2018.02.010.

Glauber, A.J., Moyer, S., Adriani, M. and Iwan, G. (2016). The Cost of Fire: An Economic Analysis of Indonesia's 2015 Fire Crisis. Indonesia Sustainable Landscapes Knowledge Note No. 1. Jakarta: World Bank. https://openknowledge.worldbank.org/handle/10986/23840.

Global Forest Watch (2020). Forest Cover in Tanjung Jabung Barat Regent, Jambi Province. Global Forest Watch and World Resources Institute.

Goode, R., Gaughen, S., Fierro, M., Hankins, D., Johnson-Reyes, K., Middleton, B.R. et al. (2018). Summary Report from Tribal and Indigenous Communities Within California. SUM-CCCA4-2018-010. Sacramento, CA: California Governer's Office of Planning and Research, California Natural Resources Agency, and the California Energy Commission. https://www.energy.ca.gov/sites/default/files/2019-11/Statewide_Reports-SUM-CCCA4-2018-010_TribalCommunitySummary_ADA.pdf.

Gresswell, R.E. (1999). Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society, 128(2), 193-221. https://doi.org/10.1577/1548-8659(1999)128<0193:faaeif>2.0.co;2.

Hahn, G.E., Coates, T.A., Latham, R.E. and Majidzadeh, H. (2019). Prescribed fire effects on water quality and freshwater ecosystems in moist-temperate eastern North America. Natural Areas Journal, 39(1), 46-57. https://doi.org/10.3375/043.039.0104.

Haynes, K., Handmer, J., McAneney, J., Tibbits, A. and Coates, L. (2010). Australian bushfire fatalities 1900–2008: exploring trends in relation to the 'Prepare, stay and defend or leave early' policy. Environmental Science and Policy, 13(3), 185-194. https://doi.org/10.1016/j.envsci.2010.03.002.

Haynes, K., Short, K., Xanthopoulos, G., Viegas, D., Ribeiro, L.M. and Blanchi, R. (2019). Wildfires and WUI fire fatalities. In Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires. Manzello, S.L. (ed.). Cham: Springer International Publishing. 1-16. https://doi.org/10.1007/978-3-319-51727-8_92-1.

Heikkilä, T.V., Grönqvist, R. and Jurvélius, M., 2007. Wildland fire management. Handbook for trainers. Ministry for Foreign Affairs of Finland, Development Policy Information Unit.

Hirsch, K.G. and Martell, D.L. (1996). A review of initial attack fire crew productivity and effectiveness. International Journal of Wildland Fire, 6(4), 199-215. https://doi.org/10.1071/WF9960199.

Hu, Y., Ai, H.H., Odman, M.T., Vaidyanathan, A. and Russell, A.G. (2019). Development of a WebGIS-based analysis tool for human health protection from the impacts of prescribed fire smoke in southeastern USA. International Journal of Environmental Research and Public Health, 16(11), 1981. https://doi.org/10.3390/ijerph16111981.

Huffman, M.R. (2013). The many elements of traditional fire knowledge: synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society, 18(4), 3. http://dx.doi.org/10.5751/ES-05843-180403.

Hunter, M.E. and Robles, M.D. (2020). Tamm review: the effects of prescribed fire on wildfire regimes and impacts: a framework for comparison. Forest Ecology and Management, 475, 118435. https://doi.org/10.1016/j.foreco.2020.118435.

Ingalsbee, T. and Raja, U. (2015). The rising costs of wildfire suppression and the case for ecological fire use. In The Ecological Importance of Mixed-Severity Fires. DellaSala, D.A. and Hanson, C.T. (eds.). Elsevier. Chapter 12. 348-371.

Inspector-General for Emergency Management (2020). Inquiry into the 2019-20 Victorian Fire Season: Phase 1 Report. Melbourne: State of Victoria. https://www.igem.vic.gov.au/fire-season-inquiry/inquiry-reports/inquiry-into-the-2019-20-victorian-fire-season-phase-1-report.

International Tropical Timber Organization (1997). ITTO Guidelines on Fire Management in Tropical Forests. ITTO Policy Development Series No 6. Yokohama. https://gfmc.online/wp-content/uploads/ITTO-Guidelines-Fire-Management-Tropical-Forests-1997-EN.pdf.

Jones, R. (1969). Firestick farming. Australian Natural History, 16, 224-231.

Jones, J.A., Wei, X., Archer, E., Bishop, K., Blanco, J.A., Ellison, D. et al. (2020). Forest-water interactions under global change. In Forest-Water Interactions. Ecological Studies Series, vol. 240. Levia, D.F., Carlyle-Moses, D.E., Iida, S., Michalzik, B., Nanko, K. and Tischer, A. (eds.). Cham: Springer International Publishing. 589-624. https://doi.org/10.1007/978-3-030-26086-6_24.

Kalies, E.L. and Kent, L.L.Y. (2016). Tamm review: are fuel treatments effective at achieving ecological and social objectives? A systematic review. Forest Ecology and Management, 375, 84-95. https://doi.org/10.1016/j.foreco.2016.05.021.

Keeley, J.E., Safford, H.D., Fotheringham, C.J., Franklin, J. and Moritz, M. (2009). The 2007 southern California wildfires: lessons in complexity. Journal of Forestry, 107(6), 287-296. https://doi.org/10.1093/jof/107.6.287.

Kousky, C., Greig, K., Lingle, B. and Kunreuther, K. (2018). Wildfire Costs in California: The Role of Electric Utilities. Philadelphia, PA: Wharton Risk Management and Decision Processes Center, University of Pennsylvania. https://riskcenter.wharton.upenn.edu/wp-content/uploads/2018/08/Wildfire-Cost-in-CA-Role-of-Utilities-1.pdf.

Langer, E.R.L. and McGee, T.K. (2017). Wildfire risk awareness and prevention by predominantly M-ori rural residents, Karikari Peninsula, Aotearoa New Zealand. International Journal of Wildland Fire, 26(9), 820-828. https://doi.org/10.1071/WF16133.

Loane, I.T. and Gould, J.S. (1985). Aerial Suppression of Bushfires: Cost/Benefit Study for Victoria. Canberra: National Bushfire Research Unit, Commonwealth Scientific and Industrial Research Organisation, Division of Forest Research. https://doi.org/10.25919/5c002929cc095.

Maclean, K., Robinson, C. and Costello, O. (eds.) (2018). A National Framework to Report on the Benefits of Indigenous Cultural Fire Management. Commonwealth Scientific and Industrial Research Organisation. https://publications.csiro.au/rpr/download?pid=csiro:EP188803&dsid=DS1.

McCaffrey, S.M., McGee, T.K., Coughlan, M. and Tedim, F. (2020). Understanding wildfire mitigation and preparedness in the context of extreme wildfires and disasters: social science contributions to understanding human response to wildfire. In Extreme Wildfire Events and Disasters: Root Causes and New Management Strategies. Tedim, F., Leone, V. and McGee, T.K. (eds.). Elsevier. Chapter 8. 155-174

McCaffrey, S.M. and Olsen, C. (2012). Research Perspectives on the Public and Fire Management: A Synthesis of Current Social Science on 8 Essential Questions – Final Report to Joint Fire Science Program and Partners in Fire Education. General Technical Report NRS-104. Newtown Square, PA: United States Department of Agriculture, Forest Service. https://www.firescience.gov/projects/06-4-1-26/project/06-4-1-26_PIFE_JFSP_Final_Report.pdf.

McCaffrey, S.M. and Rhodes, A. (2009). Public response to wildfire: is the Australian 'Stay and defend or leave early' approach an option for wildfire management in the United States? Journal of Forestry, 107(1), 9-15. https://doi.org/10.1093/jof/107.1.9.

Meldrum, J.R., Brenkert-Smith, H., Champ, P.A., Gomez, J., Byerly, H., Falk, L. et al. (2021). Would you like to know more? The effect of personalized wildfire risk information and social comparisons on information-seeking behavior in the wildland–urban interface. Natural Hazards, 106(3), 2139-2161. https://doi.org/10.1007/s11069-021-04534-x.

Miller, A.M. and Davidson-Hunt, I.J. (2010). Fire, agency and scale in the creation of Aboriginal cultural landscapes.

Human Ecology, 38(3), 401-414. https://doi.org/10.1007/s10745-010-9325-3.

Miller, M.E., Elliot, W.J., Billmire, M., Robichaud, P.R. and Endsley, K.A. (2016). Rapid-response tools and datasets for post-fire remediation: linking remote sensing and process-based hydrological models. International Journal of Wildland Fire, 25(10), 1061-1073.

https://doi.org/10.1071/WF15162.

Miller, C., Plucinski, M.P., Sullivan, A., Stephenson, A., Huston, C., Charman, K. et al. (2017). Electrically caused wildfires in Victoria, Australia are over-represented when fire danger is elevated. Landscape and Urban Planning, 167, 267-274. https://doi.org/10.1016/j.landurbplan.2017.06.016.

Mitchell, J.W. (2009). Power lines and catastrophic wildland fire in southern California. Proceedings of the 11th International Conference on Fire and Materials. San Francisco, 26-28 January 2009. 225-238.

Moghaddas, J.J. and Craggs, L. (2007). A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. International Journal of Wildland Fire, 16(6), 673-678. https://doi.org/10.1071/WF06066.

Molina-Terrén, D.M., Xanthopoulos, G., Diakakis, M., Ribeiro, L., Caballero, D., Delogu, G.M. et al. (2019). Analysis of forest fire fatalities in Southern Europe: Spain, Portugal, Greece and Sardinia (Italy). International Journal of Wildland Fire, 28(2), 85. https://doi.org/10.1071/WF18004.

Moreira, F., Ascoli, D., Safford, H.D., Adams, M.A., Moreno, J.M., Pereira, J.M. et al. (2020). Wildfire management in Mediterranean-type regions: paradigm change needed. Environmental Research Letters, 15(1), 011001. https://doi.org/10.1088/1748-9326/ab541e.

Moreira, F., Viedma, O., Arianoutsou, M., Curt, T., Koutsias, N., Rigolot, E. et al. (2011). Landscape: wildfire interactions in southern Europe – implications for landscape management.

Journal of Environmental Management, 92(10), 2389-2402. https://doi.org/10.1016/j.jenvman.2011.06.028.

Morgan, G.W., Tolhurst, K.G., Poynter, M.W., Cooper, N., McGuffog, T., Ryan, R. et al. (2020). Prescribed burning in south-eastern Australia: history and future directions. Australian Forestry, 83(1), 4-28. https://doi.org/10.1080/00049158.2020.1739883.

Multihazard Mitigation Council (2018). Natural Hazard Mitigation Saves: Utilities and Transportation Infrastructure. Washington, DC: National Institute of Building Sciences.

Neale, T., Carter, R., Nelson, T. and Bourke, M. (2019). Walking together: a decolonising experiment in bushfire management on Dja Dja Wurrung country. Cultural Geographies, 26(3), 341-359. https://doi.org/10.1177%2F1474474018821419.

Neris, J., Santin, C., Lew, R., Robichaud, P.R., Elliot, W.J., Lewis, S.A. et al. (2021). Designing tools to predict and mitigate impacts on water quality following the Australian 2019/2020 wildfires: insights from Sydney's largest water supply catchment. Integrated Environmental Assessment and Management. https://doi.org/10.1002/ieam.4406.

New South Wales Government. (2020). Final Report of the NSW Bushfire Inquiry. 31 July 2020. https://www.dpc.nsw.gov.au/assets/dpc-nsw-gov-au/publications/NSW-Bushfire-Inquiry-1630/Final-Report-of-the-NSW-Bushfire-Inquiry.pdf

North, M.P., Collins, B.M. and Stephens, S.L. (2012). Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry, 110(7), 392-401. http://dx.doi.org/10.5849/jof.12-021.

North, M.P., Stephens, S.L., Collins, B.M., Agee, J.K., Aplet, G., Franklin, J.F. et al. (2015). Reform forest fire management. Science, 349(6254), 1280-1281. https://doi.org/10.1126/science.aab2356.

North Australian Indigenous Land and Sea Management Alliance (2020). About NAILSMA. https://nailsma.org.au/about-us.

Nunes, J.P., Doerr, S.H., Sheridan, G., Neris, J., Santín, C., Emelko, M.B. et al. (2018). Assessing water contamination risk from vegetation fires: challenges, opportunities and a framework for progress. Hydrological Processes, 32(5), 687-694. https://doi.org/10.1002/hyp.11434.

Odman, M.T., Huang, R., Pophale, A.A., Sakhpara, R.D., Hu, Y., Russell, A.G. et al. (2018). Forecasting the impacts of prescribed fires for dynamic air quality management. Atmosphere, 9(6), 220. https://doi.org/10.3390/atmos9060220.

Oliveira, T.M., Barros, A.M.G., Ager, A.A. and Fernandes, P.M. (2016). Assessing the effect of a fuel break network to reduce burnt area and wildfire risk transmission. International Journal of Wildland Fire, 25(6), 619-632. https://doi.org/10.1071/WF15146.

Otero, I. and Nielsen, J.Ø. (2017). Coexisting with wildfire? Achievements and challenges for a radical social-ecological transformation in Catalonia (Spain). Geoforum, 85, 234-246. https://doi.org/10.1016/j.geoforum.2017.07.020.

Owens, D. and O'Kane, M. (2020). Final report of the NSW bushfire inquiry. Sydney: New South Wales Government. https://www.dpc.nsw.gov.au/assets/dpc-nsw-gov-au/publications/NSW-Bushfire-Inquiry-1630/Final-Report-of-the-NSW-Bushfire-Inquiry.pdf.

Plucinski, M.P. (2019a). Contain and control: wildfire suppression effectiveness at incidents and across landscapes. Current Forestry Reports, 5(1), 20-40. https://doi.org/10.1007/s40725-019-00085-4.

Plucinski, M.P. (2019b). Fighting flames and forging firelines: wildfire suppression effectiveness at the fire edge. Current Forestry Reports, 5(1), 1-19. https://doi.org/10.1007/s40725-019-00084-5.

Radeloff, V.C., Helmers, D.P., Kramer, H.A., Mockrin, M.H., Alexandre, P.M., Bar-Massada, A. et al. (2018). Rapid growth of the US wildland-urban interface raises wildfire risk. Proceedings of the National Academy of Sciences of the United States of America, 115(13), 3314-3319. https://doi.org/10.1073/pnas.1718850115.

Rakhmatulina, E., Boisramé, G., Stephens, S.L. and Thompson, S. (2021). Hydrological benefits of restoring wildfire regimes in the Sierra Nevada persist in a warming climate. Journal of Hydrology, 593, 125808. https://doi.org/10.1016/j.jhydrol.2020.125808.

Robichaud, P.R., Lewis, S.A., Wagenbrenner, J.W., Ashmun, L.E. and Brown, R.E. (2013a). Post-fire mulching for runoff and erosion mitigation. Part I: effectiveness at reducing hillslope erosion rates. Catena, 105, 75-92. https://doi.org/10.1016/j.catena.2012.11.015.

Robichaud, P.R., Wagenbrenner, J.W., Lewis, S.A., Ashmun, L.E., Brown, R.E. and Wohlgemuth, P.M. (2013b). Post-fire mulching for runoff and erosion mitigation. Part II: effectiveness in reducing runoff and sediment yields from small catchments. Catena, 105, 93-111. https://doi.org/10.1016/j.catena.2012.11.016.

Robinne F.-N., Burns J., Kant P., Flannigan M.D., Kleine M., de Groot, B. et al. (eds.) (2018). Global Fire Challenges in a Warming World. Occasional Paper No. 32. Vienna: International Union of Forest Research Organizations.

Robinne, F.-N., Hallema, D.W., Bladon, K.D., Flannigan, M.D., Boisramé, G., Bréthaut, C.M. et al. (2021). Scientists' warning on extreme wildfire risks to water supply. Hydrological Processes, 35(5), e14086. https://doi.org/10.1002/hyp.14086.

Robinson, C., Maclean, K., Costello, O. and Pert, P. (2020). Empowering Indigenous leadership in cultural burning and natural disaster recovery and resilience matters. In Climate and Disaster: Technical Report, July 2020. Commonwealth Scientific and Industrial Research Organisation. Chapter 4. 81-106. https://www.csiro.au/en/Research/Environment/Extreme-Events/Bushfire/frontline-support/report-climate-disaste-resilience.

Robinson, C., McKaige, B., Barber, M., Hill, R., Gerrard, E., Towney, L. et al. (2016). Report on the National Indigenous Fire Knowledge and Fire Management Forum: Building Protocols from Practical Experiences, Darwin, Northern Territory, 9-10 February 2016. Darwin: Commonwealth Scientific and Industrial Research Organisation and the National Environmental Science Programme's Northern Australia Environmental Resources Hub. https://www.aidr.org.au/media/4915/national-indigenous-fire-knowledge-forum-report-2016_final.pdf.

Roos, C.I., Swetnam, T.W., Ferguson, T.J., Liebmann, M.J., Loehman, R.A., Welch, J.R. et al. (2021). Native American fire management at an ancient wildland–urban interface in the Southwest United States. Proceedings of the National Academy of Sciences of the United States of America, 118(4), e2018733118. https://doi.org/10.1073/pnas.2018733118.

Roozbahani, R., Huston, C., Dunstall, S., Abbasi, B., Ernst, A. and Schreider, S. (2015). Minimizing bushfire risk through optimal powerline assets replacement and improvement. MODSIM2015: 21st International Congress on Modelling and Simulation. Weber, T., McPhee, M.J. and Anderssen, R.S. (eds.). Queensland, 29 November-4 December 2015. Modelling and Simulation Society of Australia and New Zealand. 1834-1840. https://doi.org/10.36334/MODSIM.2015.J9.roozbahani.

Sankey, S. (2018). Blueprint for Wildland Fire Science in Canada (2019–2029). Edmonton: Natural Resources Canada, Canadian Forest Service.

Santos, R.M.B., Sanches Fernandes, L.F., Pereira, M.G., Cortes, R.M.V. and Pacheco, F.A.L. (2015). Water resources planning for a river basin with recurrent wildfires. Science of the Total Environment, 526, 1-13. https://doi.org/10.1016/j.scitotenv.2015.04.058.

Scheper, A.C., Verweij, P.A. and Van Kuijk, M. (2021). Post-fire forest restoration in the humid tropics: a synthesis of available strategies and knowledge gaps for effective restoration. Science of the Total Environment, 771, 144647. https://doi.org/10.1016/j.scitotenv.2020.144647.

Schultz, C.A., McCaffrey, S.M. and Huber-Stearns, H.R. (2019). Policy barriers and opportunities for prescribed fire application in the western United States. International Journal of Wildland Fire, 28(11), 874-884. https://doi.org/10.1071/WF19040.

Simonit, S., Connors, J.P., Yoo, J., Kinzig, A. and Perrings, C. (2015). The impact of forest thinning on the reliability of water supply in central Arizona. PLOS One, 10(4), e0121596. https://doi.org/10.1371/journal.pone.0121596.

Simpson, H., Bradstock, R. and Price, O. (2019). A temporal framework of large wildfire suppression in practice, a qualitative descriptive study. Forests, 10(10), 884. https://doi.org/10.3390/f10100884.

Stasiewicz, A.M. and Paveglio, T.B. (2021). Preparing for wildfire evacuation and alternatives: exploring influences on residents' intended evacuation behaviors and mitigations. International Journal of Disaster Risk Reduction, 58, 102177. https://doi.org/10.1016/j.ijdrr.2021.102177.

State of Victoria, Department of Environment, Land Water and Planning (2019). Smoke Emission and Transport Modelling: Final Report. Melbourne. https://nla.gov.au/nla.obj-1645793140/view.

Steel, Z.L., Safford, H.D. and Viers, J.H. (2015). The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere, 6(1), 1-23. https://doi.org/10.1890/ES14-00224.1.

Steffensen, V. (2020). Fire Country: How Indigenous Fire Management Could Help Save Australia. Melbourne: Hardie Grant Books.

Stocks, B.J., Mason, J.A., Todd, J.B., Bosch, E.M., Wotton, B.M., Amiro, B.D. et al. (2002). Large forest fires in Canada, 1959–1997. Journal of Geophysical Research: Atmospheres, 107(D1), FFR 5-1-FFR 5-12. https://doi.org/10.1029/2001JD000484.

Tamm, K. and Klose, C. (2019). Wildfire in Canada: Fostering Resilience Through Advances in Modelling. Zurich: Swiss Re Institute. https://www.swissre.com/dam/jcr:29e29c5f-9a04-44fa-91ba-465d52837b61/Expertise_Publication_Wildfire_in_Canada_Fostering_resilience.pdf.

Tata, H.L., Sakuntaladewi, N., Wibowo, L.R. and Tampubolon, B.A.P. (2016). Managing peat fire risks in Tanjung Jabung Barat district, Jambi, Indonesia. 15th International Peat

Congress. Kuching, Sarawak, 15-19 August 2016. Abstract No. A-453. 381-384.

Teague, B., Pascoe, S. and McLeod, R. (2009). The 2009 Victorian Bushfires Royal Commission Final Report: Summary. Melbourne: Victorian Bushfires Royal Commission. http://royalcommission.vic.gov.au/finaldocuments/summary/PF/VBRC_Summary_PF.pdf.

Tedim, F., Leone, V., Amraoui, M., Bouillon, C., Coughlan, M.R., Delogu, G.M. et al. (2018). Defining extreme wildfire events: difficulties, challenges, and impact. Fire, 1, 9. https://doi.org/10.3390/fire1010009.

Thompson, M.P., Bowden, P., Brough, A., Scott, J.H., Gilbertson-Day, J., Taylor, A. et al. (2016). Application of wildfire risk assessment results to wildfire response planning in the southern Sierra Nevada, California, USA. Forests, 7(3), 64. https://doi.org/10.3390/f7030064.

Thompson, M.P., MacGregor, D.G., Dunn, C.J., Calkin, D.E. and Phipps, J. (2018). Rethinking the wildland fire management system. Journal of Forestry, 116(4), 382-390. https://doi.org/10.1093/jofore/fvy020.

Thompson, M.P., Rodríguez y Silva, F., Calkin, D.E. and Hand, M.S. (2017). A review of challenges to determining and demonstrating efficiency of large fire management. International Journal of Wildland Fire, 26(7), 562-573. https://doi.org/10.1071/WF16137.

Victorian Traditional Owner Cultural Fire Knowledge Group (2020). The Victorian Traditional Owner Cultural Fire Strategy. Federation of Victorian Traditional Owner Corporations. https://www.fvtoc.com.au/country.

Welch, J.R. and Coimbra Junior, C.E.A. (2019). Indigenous fire ecologies, restoration, and territorial sovereignty in the Brazilian Cerrado: the case of two Xavante reserves. Land Use Policy, 104, 104055. https://doi.org/10.1016/j.landusepol.2019.104055.

Whittaker, J. and Mercer, D. (2004). The Victorian bushfires of 2002–03 and the politics of blame: a discourse analysis. Australian Geographer, 35(3), 259-287. https://doi.org/10.1080/0004918042000311313.

Wildland Fire Executive Council (2014). The National Strategy: The Final Phase in The Development of The National Cohesive Wildland Fire Management Strategy. https://www.forestsandrangelands.gov/documents/strategy/strategy/CSPhaseIIINationalStrategyApr2014.pdf.

Williamson, G.J., Bowman, D.M.J.S., Price, O.F., Henderson, S.B. and Johnston, F.H. (2016). A transdisciplinary approach to understanding the health effects of wildfire and prescribed fire smoke regimes. Environmental Research Letters, 11(12), 125009. https://doi.org/10.1088/1748-9326/11/12/125009.

Appendix

Best, M.J., Pryor, M., Clark, D.B., Rooney, G.G., Essery, R.L.H., Ménard, C.B. et al. (2011). The Joint UK Land Environment Simulator (JULES), model description. Part 1: energy and water fluxes. Geoscientific Model Development, 4(3), 677-699. https://doi.org/10.5194/gmd-4-677-2011.

Clark, D.B., Mercado, L.M., Sitch, S., Jones, C.D., Gedney, N., Best, M.J. et al. (2011). The Joint UK Land Environment Simulator (JULES), model description. Part 2: carbon fluxes and vegetation dynamics. Geoscientific Model Development, 4(3), 701-722. https://doi.org/10.5194/gmd-4-701-2011.

Frieler, K., Lange, S., Piontek, F., Reyer, C.P.O., Schewe, J., Warszawski, L. et al. (2017). Assessing the impacts of 1.5 °C global warming: simulation protocol of the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2b). Geoscientific Model Development, 10(12), 4321-4345. https://doi.org/10.5194/gmd-10-4321-2017.

Harper, A.B., Cox, P.M., Friedlingstein, P., Wiltshire, A.J., Jones, C.D., Sitch, S. et al. (2016). Improved representation of plant functional types and physiology in the Joint UK Land Environment Simulator (JULES v4.2) using plant trait information. Geoscientific Model Development, 9(7), 2415-2440. https://doi.org/10.5194/gmd-9-2415-2016.

Intergovernmental Panel on Climate Change. (2018). Global Warming of 1.5°C: An IPCC Special Report on the Impacts of Global Warming of 1.5°C Above Pre-Industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty. Masson-Delmotte, V., Zhai, P., Pörtner, H.-O., Roberts, D., Skea, J., Shukla, P.R., Pirani, A., Moufouma-Okia, W., Péan, C., Pidcock, R., Connors, S., Matthews, J.B.R., Chen, Y., Zhou, X., Gomis, M.I., Lonnoy, E., Maycock, T., Tignor, M. and Waterfield, T. (eds.) https://www.ipcc.ch/site/assets/uploads/sites/2/2019/06/SR15_Full_Report_High_Res.pdf.

Kelley, D.I., Bistinas, I., Whitley, R., Burton, C., Marthews, T.R. and Dong, N. (2019). How contemporary bioclimatic and human controls change global fire regimes. Nature Climate Change, 9(9), 690-696. https://doi.org/10.1038/s41558-019-0540-7.

Kelley, D.I., Burton, C., Huntingford, C., Brown, M.A., Whitley, R. and Dong, N. (2021). Low meteorological influence found in 2019 Amazonia fires. Biogeosciences, 18(3), 787-804. https://doi.org/10.5194/bg-18-787-2021.

Kelley, D.I. and Harrison, S.P. (2014). Enhanced Australian carbon sink despite increased wildfire during the 21st century. Environmental Research Letters. 9(10), https://iopscience.iop.org/article/10.1088/1748-9326/9/10/104015

Lange, S. (2019). EartH2Observe, WFDEI and ERA-Interim Data Merged and Bias-Corrected for ISIMIP (EWEMBI). GFZ Data Services. https://doi.org/10.5880/pik.2019.004. Accessed 8 October 2021.

Mathison, C., Burke, E., Robertson, E., Hartley, A., Burton, C., Williams, K. et al. (In preparation). Description and Evaluation of the JULES-ES setup for ISIMIP2b. (In preparation for the Journal of Advances in Modeling Earth Systems).

O'Neill, B.C., Kriegler, E., Riahi, K., Ebi, K.L., Hallegatte, S., Carter, T. et al. (2014). A new scenario framework for climate change research: the concept of shared socioeconomic pathways. Climatic Change, 122(3), 387-400. https://doi.org/10.1007/s10584-013-0905-2.

Rogelj, J., Luderer, G., Pietzcker, R.C., Kriegler, E., Schaeffer, M., Krey, V. et al. (2015). Energy system transformations for limiting end-of-century warming to below 1.5 °C. Nature Climate Change, 5(6), 519-527. https://doi.org/10.1038/nclimate2572.

Sellar, A.A., Jones, C.G., Mulcahy, J.P., Tang, Y., Yool, A., Wiltshire, A. et al. (2019). UKESM1: Description and evaluation of the U.K. Earth system model. Journal of Advances in Modeling Earth Systems, 11(12), 4513-4558. https://doi.org/10.1029/2019MS001739.

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