Economic analysis of prescribed burning in the south-west of Western Australia

by

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ABSTRACT

In recent years, the frequency and severity of large wildfires have increased in Australia and elsewhere. These large fires occur despite advances in fire-fighting technology, considerable suppression efforts, and record expenditures on wildfire suppression. Suppression expenditures in Australia have increased substantially in the last few decades, and this trend is expected to continue. However, increasing suppression capacity alone will not solve the wildfire problem. There is a risk that Australia will continue to increase its fire-fighting capacity and expenditures without improving its management of fire in the landscape. However, wildfires are also a natural element of the Australian environment that cannot be eliminated. Therefore, they need to become an integral part of land and ecosystem management.

One way to manage wildfire risk is through the application of prescribed burning. However, its use to mitigate wildfire effects has generated considerable debate. Much of this debate revolves around the efficacy of prescribed burning in reducing wildfire extent and severity, but relatively little attention has been given to the economic impacts of prescribed burning programs and the trade-offs in the allocation of resources between different fire management activities. Despite much land being prescribed burned in some parts of Australia, there has been almost no evaluation of the costs and benefits of the practice. Without the help of sound economic analyses, it is not clear which strategy for prescribed burning yields the best return on investment.

This study aims to fill these gaps in research and provide a framework through which the trade-offs between prescribed burning, wildfire suppression and wildfire damages can be brought to light and evaluated. I used an economic model in conjunction with a wildfire simulator to test different prescribed-burning strategies and identify the strategy that yields the highest return to society under different scenarios. I conducted a short-term analysis (one year) and a long-term analysis (15+ years) in order to evaluate and compare the impacts of short-term and long-term approaches to decision making in fire management. I applied this model to a case study area located in the south-west of Western Australia (WA).

I found that for a short-term analysis, there is not a significant difference in the estimated economic net benefits when the level of prescribed burning is varied over a wide range of values. What changes in the short term is the proportion of management costs and the proportion of damages for different prescribed-burning rates, but an increase in one

approximately cancels out a decrease in the other. Compared to a no-prescribed-burning scenario, short-term investments in prescribed burning generate a benefit of AU$0.7 to AU$1.5 per dollar invested.

The long-term analysis shows that not doing any prescribed burning for several years can be very costly for the south-west of WA, with large increases in damages and suppression expenditures. The net benefits of prescribed burning appear much larger in a long-term analysis compared to a short-term analysis. As a result, the long-term analysis shows a more clear-cut answer. It suggests that substantial benefits that can be gained from increasing the amount of area prescribed burned per year in the region. There are not large economic differences between prescribed burning 10, 15 and 20% of public land in the long term, but each of these is greatly superior to doing nothing. The results from the long-term model indicate that prescribed burning may generate between AU$10 and AU$47 benefits every year per dollar invested compared to doing nothing.

Dedicated to

Jean-Joseph Florec

Wherever you are, I know you are taking care of me

Marleny Correa and Patrick Florec

I love you dearly

Fabian Linge

Life is beautiful with you

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

Chapter 1.

INTRODUCTION

In the last few decades, the frequency of large, intense wildfires has increased in many places around the world, including Australia. These fires have caused significant social, economic and environmental impacts. One of those impacts is the substantial increase in suppression expenditures. However, there is a risk that suppression expenditures will continue to increase without reducing the potential for mega-fires to occur. A strategy that has been put forward for the management of wildfires is the purposeful application of fire to the landscape under milder climatic conditions, that is, prescribed burning. To decide on the appropriate level of investment in prescribed burning, fire managers need to have a clear picture of the benefits and costs of the practice and the consequences of changing the level of investment. However, studies that elucidate these benefits and costs are rarely available. In an effort to increase the availability of this type of studies, this thesis explores the application of economic analysis to wildfire management in the south-west of Western Australia (WA) and aims to evaluate trade-offs between prescribed burning, wildfire suppression and wildfire damages in the region. This analysis can be of interest to fire managers and policy makers in WA and elsewhere as it provides them with useful information for making decisions regarding the allocation of resources for wildfire management. It can also be relevant to fire scientists working on the economic or ecological aspects of fire management.

1.1. Background and rationale

Wildfires are a worldwide phenomenon that can cause significant damage to ecosystems, life and property (Gill 2005; Bowman *et al.* 2009). Every year in Australia, the United States (US), and in many other parts of the world, large uncontrolled fires burn in a variety of landscapes destroying economic, environmental and social assets (Williams *et al.* 2011). Total wildfire losses are difficult to estimate (Abt *et al.* 2008), but there have been some estimates of global and per-country annual wildfire losses. On average, it is estimated that wildfires have been the cause of approximately US$2 billion (AU$2.5 billion) economic losses per annum at a global scale for the last 30 years, or US$4.7 billion (AU$6 billion) per annum for the last 10 years (CRED 2015). However, these figures may be considerably underestimating total losses due to wildfires. In the 1997-1998 fire season alone, the Indonesian fires resulted in economic losses between US$ 8.8 to US$9.3 billion (AU$11.3 to AU$11.9 billion) and fires in Latin American caused an estimated US$10 to US$15 billion (AU$12.8 to AU$19.2 billion) in damages (Bowman *et al*. 2009), but the Emergency Events Database (CRED 2015) recorded a total damage of US$10.9 billion for all of 1997 and 1998 together (the entire two years). In Australia, it has been estimated that wildfires cost the country more than AU$80 million (US$61.4 million) annually on average (DCCEE 2010). But these estimates are far from accurate. According to the International Disasters Database, wildfires have caused on average US$185 million (AU$237 million) damages per year for the last 30 years (CRED 2015). However, the database recorded AU$1.3 billion (US$1 billion) damages for all of 2009, while the total cost of the Black Saturday fires alone (February 2009) was estimated to be AU$4.2 billion ($US3.2 billion) (Attiwill and Adams 2013). Economic consequences aside, in Australia wildfires are the most dangerous type of natural disaster in terms of risk to human life (Bureau of Transport Economics 2001; Teague *et al.* 2010). They have resulted in the deaths of more than 230 people and 1000 injuries in the last 30 years (CRED 2015).

Over the past two decades, the frequency and severity of large wildfires have increased in most vegetated landscapes around the world (Bowman *et al*. 2009). Also, in developed countries, house values have increased, and more houses have been built in areas of greater fire risk. As a result of these changes, the social, economic and environmental impacts of wildfires have increased substantially (Morgan 2009; Williams *et al.* 2011). In the US for instance, in the period since 2003 the states of Washington, California, Georgia, Florida, Utah, New Mexico, among others, have all had their largest recorded wildfire in history and/or the most destructive wildfire in history (NIFC, 2014). In Australia, the fires of Black Saturday in 2009 caused the highest loss of life and property from a wildfire in Australian history (Teague *et al*. 2010). These large fires and losses occur despite advances in fire-fighting technology, greater suppression capacity, considerable suppression efforts, and record expenditures on wildfire suppression (Toman *et al.* 2011; Attiwill and Adams 2013).

In an effort to protect human communities and environmental assets, fire agencies spend large amounts of money in suppressing and preventing fires (Stockmann *et al.* 2010a; Preisler *et al.* 2011; Gude *et al.* 2013). In recent years, these costs, and in particular the costs of wildfire suppression, have increased substantially in many parts of the world (Morgan *et al.* 2007; Cochrane *et al.* 2012; Gude *et al.* 2013; Thompson *et al.* 2013a). In countries like the US and Australia, fire suppression expenditures have reached record highs (Morgan *et al.* 2007; Thompson *et al.* 2013a). For instance, in Australia the total expenditure of fire service organisations has increased from $1.7 billion in 1998 (in 2013AU$) to AU$3.7 billion in 2013 (SCRGSP 2000, 2014).[1] In the US, suppression expenditures by the US Forest Service have increased from less than US$300 million in the early 1980s to more than a billion dollars (in real terms) in most years since 2000 (Calkin *et al.* 2005; Thompson *et al.* 2013a; Hand *et al.* 2014). These increases in suppression costs have been attributed to the cumulative effects of: (1) the increase in the number, size and intensity of wildfires due to changes in weather patterns, (2) the continuous expansion of the wildland-urban interface, and (3) decades of fire exclusion and aggressive suppression that resulted in extensive fuel build-ups in fire-prone landscapes (Liang *et al.* 2008; Stetler *et al.* 2010; Stockmann *et al.* 2010b; Cochrane *et al.* 2012). This increasing trend in suppression expenditures is expected to continue in the US (Thompson *et al.* 2013a) and in Australia (Morgan *et al.* 2007). However, increasing suppression capacity alone will not solve the problem of the increasing fire threat (Marino *et al.* 2014). There is a risk that Australia will continue to increase its fire-fighting capacity and suppression expenditures without improving its management of fire in the landscape (Morgan *et al*. 2007).

Fire management draws resources from a pool of limited human and financial resources, competing for these resources with other areas of public spending (Ganewatta 2008). Government budgets are limited, and as wildfire management costs increase and represent a higher proportion of Governments' budgets, less money is available for other services that are useful to society (Thompson *et al.* 2013a). Similarly, as suppression expenditures increase and utilise a higher portion of fire agencies' fixed budgets, future budgets for other fire-management responsibilities, such as prescribed burning and education campaigns, are reduced (Calkin *et al.* 2005; Liang *et al.* 2008; Thompson *et al.* 2013a). As a result, fire agencies are under increasing pressure to justify their investment decisions and manage wildfires in a cost-effective manner (Stetler *et al.* 2010; Calkin *et al.* 2011). But with increases in fire size, fire severity, management costs and risk to assets, making decisions regarding the allocation of resources for fire management has become more challenging (Wilson *et al.* 2011; Clayton *et al.* 2014).

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[1] The total expenditure of fire service organisations reported by the Steering Committee for the Review of Government Service Provision (Productivity Commission) includes expenditures relating to wildfires, structural fires and other emergency services. However, in most States and Territories in Australia, expenses associated with wildfires represent the majority of services provided by fire services organisations. Therefore, the figures reported by the Productivity Commission closely illustrate the trend in total expenditure on wildfires at a national level.

With more severe wildfire impacts and suppression expenditures at record highs, other strategies need to be implemented for the management of wildfires. A strategy available to fire managers is managing fuel levels through prescribed burning (Mercer *et al.* 2008; Stockmann *et al.* 2010b; Toman *et al.* 2011). The purposeful application of fire to the landscape in mild weather conditions is used in many fire-prone landscapes for wildfire management and protection of human assets (Fernandes and Botelho 2003; Penman *et al.* 2011). More recently it has been used for the protection of ecological assets and ecosystems restoration (Driscoll *et al.* 2010; Bradstock *et al.* 2012b; McCaw 2013). However, the increased focus on prescribed burning to mitigate potential wildfire effects has generated considerable debate (Penman *et al.* 2011; Altangerel and Kull 2013; Ryan *et al.* 2013). Much of this debate revolves around the efficacy of prescribed burning in reducing wildfire extent and severity (Fernandes and Bothelo 2003), but little attention has been given to the economic impacts of prescribed burning programs and the trade-offs in the allocation of resources between different fire management activities. Despite much land being prescribed burned in some parts of Australia, there has been almost no evaluation of the costs and benefits of the practice. Without the help of sound economic analyses, it is not clear which option yields the best returns on investment.

Fire-management resources may be utilised in a variety of ways and their alternative uses have different implications for economic, environmental and social assets, which may be profound (Ganewatta 2008). In order to identify the most efficient allocation of resources and the strategies that generate the highest benefits to society and the environment, fire managers need more information about the implications of different uses of these resources in the short and the long term (Ganewatta and Handmer 2006). They need to know the returns on investment of different strategies and the trade-offs between available options (Ganewatta 2008). However, this information is rarely available to them (Clayton *et al*. 2014). National enquiries and government reports that recommend changes in prescribed burning levels (e.g. Teague *et al.* 2010; Keelty 2012) do not include economic analyses that assess the potential economic impacts of such changes. In the scientific literature, despite abundant theoretical research on the economics of fire management, and a number of studies that evaluate one single management action, empirical studies that look at fire management programs as a whole and attempt to quantify the trade-offs between different management activities are still scarce (see Chapter 2).

This study aims to fill this gap in research and provide a framework through which the trade-offs between prescribed burning, wildfire suppression and wildfire damages can be brought to light and interpreted. Through the application of economic analysis to fire management in the south-west of Western Australia (WA), my main goal is to appraise the impacts of changing the prescribed burning strategy under different scenarios and different timeframes. This study seeks to provide improved understanding about the implications of different uses of limited resources in fire management, taking into account the following facts: (1) prescribed burning costs change with the size and location of treatments (Berry *et al.* 2006; Calkin and Gebert 2006), (2) varying the size and the location of the treatments can strongly affect their efficiency (Fernandes and Bothelo 2003), and (3) the assets protected differ in nature and value (Ganewatta 2008). As an applied study, this thesis also proposes measures that may help improve the use of prescribed burning in the south-west of WA.

This study is not an analysis of the effectiveness of prescribed burning in reducing wildfire hazard. Rather, that information is an input to this study. The main focus is on the evaluation of economic impacts of alternative prescribed burning strategies, and for this purpose the study draws on the current available knowledge in the literature about the effects of prescribed burning on wildfire behaviour. Some studies have shown that prescribed burning reduces the intensity and extent of wildfires (Rawson *et al.* 1985; Boer *et al.* 2009) and can facilitate fire suppression efforts (McCarthy and Tolhurst 2001). However, the scientific evidence regarding the effectiveness of prescribed burning programs is sparse (Oliveras and Bell 2008), and significant uncertainties remain over the long-term effectiveness of prescribed burning in mitigating wildfire effects, notably under the effects of climate change and the continuous development of the wildland-urban interface (Fernandes and Botelho 2003; Gill and Stephens 2009). These uncertainties are acknowledged and, with the help of sensitivity analyses, an effort is made to elucidate how they affect the results of this study.

1.2. Research objectives

The main objective of the analysis presented in this thesis is to evaluate the economic impacts of modifying the prescribed burning strategy in the south-west of WA, including the effects on suppression expenditure and wildfire damages. The aim is to identify the prescribed burning strategy that maximises the benefits to society and the environment, and to ascertain the economic and spatial attributes of such a strategy in terms of the area that is prescribed burned, location of treatments, and dollars invested. With this broad objective in mind, the study addresses the following research questions:

1. What prescribed-burning regime minimises the sum of fire management costs and damages in the South West forest region of WA?
2. Is the answer clear-cut or is there a range of strategies that are close to being optimal?
3. What is the difference in the results between an economic analysis of prescribed burning in the short term and in the long term, and how could these results impact decision making?
4. What difference does it make to take into account the differences in asset values of different types of assets in the landscape, and the difference in prescribed-burning costs depending on size and location?
5. Given the inherently stochastic nature of the problem, what can an analysis of the distribution of results tell us in addition to the analysis of their means?

1.3. Methodology

In this study, prescribed burning in the South West forest region of WA has been evaluated using a combination of economic techniques and simulation. To test the effects of prescribed burning on wildfire impacts, a large number of wildfires were simulated under varying climatic conditions and different prescribed burning levels using a wildfire simulator. The economic impacts of modifying the prescribed burning strategy were evaluated using a modified version of an existing economic model: the Cost plus Net Value Change (C+NVC) model. This section briefly outlines the methodology that has been used in this thesis. A more detailed explanation of the methods is presented in Chapter 3.

1.3.1. Economic model

Most economic analyses of wildfire management policy have been based on the C+NVC model, or its predecessor, the Least Cost plus Loss (LC + L) (Mercer *et al.* 2007; Gebert *et al.* 2008). The C+NVC model evaluates the welfare changes that result from a change in investment in the pre-suppression strategy. In this study, the C+NVC was modified to have prescribed burning as the only pre-suppression strategy that is evaluated. Hence, the area prescribed burned is the independent variable and suppression and net damages are the dependent variables. Other fire management activities that are also part of pre-suppression, such as education campaigns, detection or pre-positioning of fire-fighting resources, and the consequences of changing the level of investment in these activities are not examined in this thesis.

Previous studies evaluating the costs and benefits of different fire management strategies have focused on minimising area burned by wildfires or intensity-weighted area burned (see for instance Mercer *et al.* 2007; Butry *et al.* 2010). The problem with minimising area burned is that such an objective does not take into consideration the differences in value of the assets protected. It is built on the assumption that all areas have equal value, whether they are endangered-species habitat, agricultural land, densely populated areas, or some other category. In the past, an objective of minimising area burned may have been similar in practice as minimising economic losses, but in today's complex environment it is unlikely to be (Abt *et al.* 2008). Therefore in this study, the objective function minimises the sum of management costs and damages, instead of area burned or intensity-weighted area burned. The model has been designed to take into account the differences in value of different assets in the landscape. In this manner, this study can determine which assets are mostly burned in the landscape and which assets generate the largest proportion of damages. This is one of the key contributions of this thesis.

Previous applications of the C+NVC model use simulations of average fire seasons. However, it is the non-average fire seasons that most influence costs and damages estimates. Although extreme fire events are rare, they account for most fire costs and losses (Handmer and Proudley 2008). Thus, the study of non-average fire seasons and the inclusion of catastrophic events are of paramount importance for the evaluation of wildfire management programs. In this study, the C+NVC model was extended to incorporate rare/extreme fire events. Estimated costs are weighted averages, depending on the frequencies of fires of different severities and sizes. This is another important contribution of this thesis.

1.3.2. Wildfire simulator

To derive the C+NVC curve, a large number of fire seasons were simulated for each prescribed-burning regime. The fire simulator used for the simulations in this study is the AUSTRALIS wildfire simulator, which was developed at The University of Western Australia by members of the School of Computer Science and Software Engineering (CSSE) (Johnston *et al.* 2008; Kelso *et al.* 2015; Simulation and Modelling Research Group n.d.).

1.3.3. Estimation of the C+NVC curve

The C+NVC curve is the sum of three curves: pre-suppression cost, suppression cost and net value change. The minimum of the curve corresponds to the prescribed-burning regime that maximises the benefits to society. In the modified version of the C+NVC model used in this study, those curves are: prescribed-burning costs, suppression costs and net value change (or net damages). Prescribed-burning costs are calculated as a function of treatment size and the proximity of the treatments to a town. Suppression costs depend on fire size, which is the factor that has the most significant influence on suppression expenditures (Calkin *et al.* 2005; Gebert *et al.* 2007; Liang *et al.* 2008; Abt *et al.* 2009). Damages are calculated with the output from the AUSTRALIS simulator. The area burned by each fire is overlaid on a raster dataset that contains the value in dollars for different categories of land (i.e. urban areas, agricultural areas, forests, etc.). Depending on the intensity with which each cell burns, a percentage of the area overlapped is considered destroyed and added to the total damages.

One advantage of this approach is that it uses a common unit of measure to evaluate wildfire effects on different types of assets, so wildfire impacts on human infrastructure and ecological values are appraised in the same currency (i.e. dollar values). By using a common unit of measure, it is possible to estimate total expected wildfire effects and evaluate the cost-effectiveness of different management activities that may be proposed (Finney 2005). Another advantage of this approach is that it makes possible the quantification of damages for different types of assets.

1.4. Outline of the study

Chapter 2 is a review of the literature on the economics of fire management. Chapter 3 presents a review of fire management in the south-west of WA, including a historical outlook of fire management and current challenges. Chapter 4 presents the detailed exposition of the model used in this thesis. Results of the first application of the C+NVC to fire management in the south-west of WA are presented in Chapter 5. In this first application of the model, the short-term economic impacts of changing the prescribed-burning strategy are evaluated. For the short-term analysis, the spatial arrangement of the treatments is also modified and evaluated.

In Chapter 6, the model is expanded to investigate the long-term economic effects of different investments in prescribed burning. The aims of this analysis is to estimate the cost to the South West forest region of not doing any prescribed burning for several years, compare the results with the short-term analysis, and highlight the implications for short-term and long-term decision making in relation to prescribed-burning investments.

In the literature examining the long-term economic effects of prescribed burning, most studies use a measure of central tendency (usually the mean) of stochastic simulations to present the results (e.g. Mercer *et al*. 2007; Butry *et al*. 2010). The distribution of the results is not analysed or reported, despite potentially providing valuable information to fire managers for decision making on prescribed-burning investments. The distribution of the results for a prescribed-burning rate that is applied in the long term may have implications for management and the protection of valuable assets. In Chapter 7 the variance of the results and the skewness of the distributions are analysed.

Finally, Chapter 8 provides a summary of the main conclusions and implications of the study. Also in this Chapter, some of the limitations of the study are reviewed and directions for future research suggested.

Chapter 2.

Literature review

The main purpose of this literature review is to identify areas of research relating to the economics of wildfire management that have been well covered in the literature and those that require further study. Another aim of this review is to identify insights in the existing literature that can be useful in addressing the research objectives of this thesis. Since the focus of this thesis is on the application of economic analysis to fire management, the studies reviewed in this Chapter contain some economic analysis in them. Other studies that can be relevant to fire management, such as decision making frameworks and ecological assessments, if they do not include any economics, are excluded. Non-economic studies are only mentioned when they provide knowledge that may significantly influence the way economics analysis is applied to fire management or when uncertainties in fire science may affect the robustness of the results of economic studies.

The use of economics in the wildfire literature is not new, but it remains relatively limited. Since the development of the first economic models for fire management by Headley (1916) and Lovejoy (1916), the wildfire economics literature has expanded, but today it still is relatively small compared to other fields of wildfire research, such as wildfire behaviour, wildfire risk and fire regimes. However, concerns about the growing number of mega–fires (Williams *et al*. 2011) and the considerable increase in suppression expenditures (Calkin *et al*. 2005) have prompted the fire economics field to expand. More research on fire economics has been conducted in the last 16 years than in the previous 50 years combined (see Table 2.1). Economic analysis is now being applied to fire management in countries where none was previously applied, such as Australia, Portugal and Spain (e.g. Bennetton *et al.* 1998; Pedernera *et al.* 2008; Rodriguez y Silva and Gonzalez-Caban 2010). However, there is still a considerable need for more economic analysis of fire management programs. In particular, there is a need for more empirical work that elucidates trade-offs between available options. With increased interest from fire managers and policy makers to integrate economics into the decision making process (Clayton *et al*. 2014), this field of study will most likely continue to grow.

In economic terms, fire management activities and fire impacts may be encapsulated in a three–stages cycle: (1) the pre–fire or pre–suppression stage, which is defined as all expenditures associated with activities carried out before the fire event occurs (or before the start of the fire season) such as prevention, education, detection, fuel management (e.g., prescribed burning, mechanical fuel reduction) and pre–positioning of fire–fighting resources; (2) the during–fire or suppression stage, which is defined as all expenditures associated with activities carried out during a fire event (i.e. after the fire–fighting resources have been deployed); and (3) the post–fire stage, which encompasses the resulting net damages and expenditures for rehabilitation projects (Gebert *et al*. 2008).[2] Although these stages are presented in a linear sequence, they should be envisaged as a circle, where the post–fire stage corresponds to the pre–fire stage of the next period, and so on (Gebert *et al*. 2008).

Table 2.1. Summary of studies published in the field of fire economics since early 1900s

Decade	Publications relating to fire economics*
Before 1950s	Beall (1949); Craig *et al.* (1945); Craig *et al.* (1946a); Craig *et al.* (1946b); Flint (1928); Headley (1916); Hornby (1936); Lovejoy (1916); Sparhawk (1925)
1950s	Arnold (1950)
1960s	Mactavish (1965); Parks (1964)
1970s	Davis (1971); Gorte and Gorte (1979); Mills (1979); Noste and Davis (1975); Simard (1976)
1980s	Baumgartner and Simard (1982); Gonzalez-Caban (1984); Gonzalez-Caban and McKetta (1986); Gonzalez-Caban *et al*. (1986); Healey *et al*. (1985); Jackson *et al.* (1982); Mills and Bratten (1982); Rich (1984); Rich (1989)
1990s	Bennetton *et al*. (1998); Botti (1999); Boxall *et al*. (1996); Boxall *et al*. (1999); Cleaves and Brodie (1990); Cleaves *et al*. (1999); Donovan *et al.* (1999); Englin *et al*. (1996); Fried *et al.* (1999); Gebert and Schuster (1999); Glover and Jessup (1999); Gonzalez-Caban (1997); Gonzalez-Caban and Omi (1999); Hesseln and Rideout (1999a); Hesseln and Rideout (1999b); Hesseln *et al.* (1998); Jones *et al.* (1999); Loomis and Gonzalez-Caban (1994); Loomis and Gonzalez-Caban (1997); Loomis and Gonzalez-Caban (1998); Loomis and Gonzalez-Caban (1999); Loomis *et al.* (1996); Macleod and Ludwig (1991); Mees *et al.* (1994); Omi *et al.* (1999); Rideout and Omi (1990); Rideout and Omi (1995); Steele and Stier (1998); Winter *et al.* (1996)

* Theoretical, empirical, review and discussion publications relating to fire economics are included in this table

[2] Initial attack, which corresponds to the action taken by the first fire–fighters that arrive at the fire incident, is sometimes considered to be part of the pre–suppression effort and in other cases considered to be part of suppression activities (direct fire–fighting expenditures).

Table 2.1 contd.

Decade	Publications relating to fire economics*
2000s	Abt *et al.* (2008); Abt *et al.* (2009); Berry and Hesseln (2004); Berry *et al.* (2006); Bureau of Transport Economics (2001); Butry (2009); Butry *et al.* (2001); Calkin and Gebert (2006); Calkin *et al.* (2005); Calkin *et al.* (2008); Cardoso de Mendonça *et al.* (2004); Cardoso de Mendonça *et al.* (2006); Crowley *et al.* (2009); Dale (2006); Donahue (2004); Donovan and Brown (2005); Donovan and Rideout (2003a); Donovan and Rideout (2003b); Donovan *et al.* (2008); Drucker *et al.* (2008); Dyer and Stafford Smith (2003); Englin *et al.* (2001); Ganewatta (2008); Ganewatta and Handmer (2006); Gebert *et al.* (2007); Gebert *et al.* (2008); Gonzalez-Caban (2007); Gonzalez-Caban *et al.* (2007); Haight and Fried (2007); Handmer and Proudley (2008); Hartsough *et al.* (2008); Hesseln (2000); Hesseln *et al.* (2003); Hesseln *et al.* (2004a); Hesseln *et al.* (2004b); Howe and Cochrane (2005); Kaval (2009); Kaval and Loomis (2008); Kaval *et al.* (2007); Liang *et al.* (2008); Loomis (2004); Loomis *et al.* (2001); Loomis *et al.* (2003); Loomis *et al.* (2004); Loomis *et al.* (2005); Loomis *et al.* (2008); Loomis *et al.* (2009); Loureiro *et al.* (2004); Lynch (2004); Martin *et al.* (2007); Marzano *et al.* (2008); Mavsar and Farreras (2009a); Mavsar and Farreras (2009b); McKee *et al.* (2004); Mercer and Prestemon (2005); Mercer and Prestemon (2008); Mercer *et al*. (2007); Mercer *et al.* (2008); Morgan (2009); Morgan *et al.* (2007); Mueller *et al.* (2009); Omi (2008); Pedernera *et al.* (2008); Prestemon *et al.* (2001); Prestemon *et al.* (2008); Rideout and Ziesler (2008); Rideout *et al.* (2008); Rittmaster *et al.* (2006); Rodriguez y Silva (2008); Schaaf *et al.* (2008); Smalley (2003); Snider *et al.* (2006); Spring and Kennedy (2005); Spring *et al.* (2008); Walker *et al.* (2007); Wei *et al.* (2008); Winter and Fried (2001); Winter *et al.* (2004); Yoder (2004); Yoder and Blatner (2004); Yoder *et al.* (2003a); Yoder *et al.* (2003b)
2010s	Bar Massada *et al.* (2011); Busby and Albers (2010); Busby *et al.* (2013); Butry *et al.* (2010); Clayton *et al*. (2014); Christman and Rollins (2015); Donovan *et al.* (2011); Gibson and Pannell (2014); Gonzalez-Caban *et al.* (2013); Gude *et al.* (2013); Heckbert *et al.* (2012); Hesseln *et al.* (2010); Kochi *et al.* (2010); Liu *et al.* (2010); McFarlane *et al.* (2011); Mell *et al.* (2010); Milne *et al.* (2014); Mutch *et al.* (2011); North *et al.* (2012); Penman *et al*. (2014); Petrovic and Carlson (2012); Preisler *et al.* (2011); Prestemon *et al.* (2010); Prestemon *et al.* (2012); Richardson *et al.* (2012); Rodriguez y Silva and Gonzalez-Caban (2010); Stetler *et al.* (2010); Stockmann (2007); Stockmann *et al.* (2010a); Stockmann *et al.* (2010b); Thompson and Anderson (2015); Thompson *et al.* (2013a); Thompson *et al.* (2013b); Thompson *et al*. (2015); Wibbenmeyer *et al.* (2013); Wilson *et al.* (2011)

* Theoretical, empirical, review and discussion publications relating to fire economics are included in this table

To date, most wildfire economics applications have focused on a single aspect or stage of the fire problem (Gebert *et al*. 2008; Milne *et al*. 2014). The vast majority of wildfire economic studies have concentrated on pre–suppression activities, predominantly dealing with prescribed burning (Gebert *et al*. 2008). A number of studies have applied economic analysis to suppression activities and fire impacts, and a few have estimated wildfire losses. A very small number of studies have dealt with the 3 stages at once, and hardly any has quantified trade-offs between different management activities. Such analyses of fire management programs that simultaneously evaluate pre-suppression, suppression and damages, although much needed, remain particularly scarce. Furthermore, throughout the fire economics literature, there is a noticeable disparity between the estimation of costs and benefits; most studies have estimated management costs, but only a handful have looked at the benefits that different management activities can generate.

This Chapter starts by reviewing the theoretical developments in fire economics since the development of early models in the first quarter of the 20th century. The subsequent sections review the empirical literature, which has been divided here between the costs of fire management activities, the benefits of fire management, the consequences of wildfires, integrated assessments, and incentive structures. Each one of these areas of research is presented in a separate section.

2.1. Evolution of fire economic models

Since the early works of Headley (1916), Lovejoy (1916) and Sparhawk (1925), there have been irregular but ongoing additions to the literature, and today there is a considerable number of studies on the theory of fire economics. To summarise the key theoretical developments in this field, this section provides a brief chronological overview of key studies. A large part of this section focuses on the development and reformulation of the "Least Cost Plus Losses" (LC+L) model, which subsequently became the "Cost Plus Net Value Change" (C+NVC) model. The latter is currently the most accepted economic model for evaluating fire management programs (Mercer *et al*. 2007; Ganewatta 2008; Gebert *et al*. 2008; Rodriguez y Silva and Gonzalez-Caban 2010).

2.1.1. Early studies and the development of the Least Cost Plus Losses model

Economists outlined the economic principles of wildfire management in the early 1900s. Since its onset, the wildfire economics discipline has aimed to identify the optimal level of investment in fire management (i.e. the quantity and quality of resources or the amount of money that can justifiably be spent in fire management). Early work on the economics of fire management focused on the protection of productive resources of the thriving timber industry in the first half of the 20th century. These early studies on the economic efficiency of management programs formulated the economic questions as a minimisation problem. Headley (1916) suggested the application of the principle of minimum total damage and cost to wildfire management. In his manual for fire management, he provided examples of different levels of investment in suppression which indicated that the investment should only be increased (or decreased as the case may be) if it generates enough benefits to cover the additional cost. He also recommended the application of economic analysis to individual fire events in order to determine whether they could be left to burn additional area, subject to the economic risk of letting the fire burn given the climatic conditions present.

In an analysis of the costs and benefits of forest protection, Lovejoy (1916) identified the costs of fire protection plus the costs in the form of fire losses as the charges to be added to a forest business (which could be the government), since insurance for the forests cannot be purchased.[3] These charges can be justified for the business as long as they are not prohibitive (i.e. as long as their payment does not result in the elimination of profits). Hence, in his analysis the economic problem of forest fires was also expressed as a minimisation problem, where the sum of costs and damages is to be minimised. He recognised, however, that with the data available at the time, the ratio between cost and effectiveness in protection could not be calculated. But he was confident that, given the value of the forests to be protected, an increase in the level of investment in fire protection by the Forest Service at the time would be justified.

Sparhawk (1925) expanded this minimisation concept and formalised it in to what later came to be known as the Least Cost Plus Losses (LC+L) model. In this model, the most-efficient investment in primary fire protection (also called pre-suppression, which includes prevention, detection and preparation of suppression forces) is the amount that minimises the sum of protection costs and total liability (total liability corresponds to the sum of suppression costs and damages). Here, pre-suppression expenditure is treated as the independent variable that determines suppression costs and damages. Sparhawk's main objective in conducting his study was to identify a scientific method that would help answer the following questions relating to fire management: (1) how much money spent on fire protection in the national forests of the

[3] In early work on the economics of fire management, the term *fire protection* was often used, but its meaning varied greatly from one study to another. For some authors it denoted fire suppression (Headley 1916; Parks 1964; Mactavish 1965), for others (Sparhawk 1925; Flint 1928) it represented fire prevention or pre-suppression, and for some others (e.g. Lovejoy 1916) it meant both fire prevention and fire suppression.

western US can be justified by the benefits it generates, and (2) what is the best way to distribute this money between the different units of a fire management organisation.

In his study, Sparhawk calculated damages depending on area burned and suppression costs depending on the average time that it takes to control the fires. He demonstrated that the time that elapses between fire detection and fire suppression has a significant influence on the final area burned by the fire, and thus on the resulting damages. In addition, he recognised that not all areas have equal value and that fires can burn areas of low liability and areas where the damages and costs can be much greater. Thus in his analysis, damage estimates are different for different areas. The only forests values considered in Sparhawk's study are timber values of the national forests and no consideration is given to values on private land. Although he recognises the existence of non-market values in forested areas, he considers that they would be negligible for forests that produce merchantable timber. Non-market values in grasslands and bushland are included, but he assigned them an arbitrary value.

In the 1930s and up to the 1960s, the LC+L model continued to be used for determining the most efficient level of fire protection. During this period, some authors introduced minor changes to the LC+L model (Flint 1928; Hornby 1936) but kept the fundamental principles of the model unchanged. For instance, Flint (1928) applied a very similar version of Sparhawk's model, but he expressed pre-suppression expenditures in cents per acre protected and losses in cents per acre burned instead of using total expenditure or total losses. Hornby (1936) used a modified version of the LC+L model, with acres burned as the independent variable. In his modified version of the model, suppression costs and damages increase with the quantity of acres burned and pre-suppression expenditure decreases with increasing area burned. Hornby used acres burned because he found that damages were difficult to estimate as a function of pre-suppression effort. The general conclusions of these studies tended towards increases in pre-suppression and suppression efforts and generally reinforced the objective of fire exclusion.

Other studies moved away from the analysis of pre-suppression and focused on the optimisation of suppression resources, like optimising the number of fire-fighters to be deployed for a fire (Arnold 1950; Parks 1964; Mactavish 1965). They used the LC+L method to analyse fire control organisation. Arnold (1950) came back to Sparhawk's demonstration of the increase in area burned with increasing time elapsed between fire detection and fire suppression. He considered this concept of elapsed time to be of critical importance and developed an economic model for fire control based on it. In his model, Arnold measured suppression costs depending on the number of men dispatched and the length of time the fire burns before they arrive (elapsed time). A suppression costs plus damage curve is calculated for different levels of elapsed time. He optimised the suppression force size for each level of elapsed time and then derived a total cost plus losses curve that includes the costs of getting the crews to the fire at different elapsed times, prevention costs, suppression costs and losses.

Parks (1964) focused on the analysis of initial attack and developed decision rules for determining the optimal amount of human resources required for suppressing wildfires. His objective was to identify the level of suppression that minimise total costs of suppression plus fire damage, and therefore he did not include detection and prevention costs in his analysis. He recommended a significant increase in the number of fire-fighters to be deployed for fires occurring in the Plumas National Forest in California.

Mactavish (1965) recognised that the suppression capacity of a given crew and its equipment depends on many other factors (apart from the size of the crew) such as fuel type, soil, topography, proximity to water supplies, and fire intensity. He extended Arnold's (1950) model by incorporating intensity levels in the analysis. Mactavish's model derived a different suppression costs plus damages curve for different ranges of elapsed time, where the minimum of each curve indicated the optimal crew size for a given elapsed time, and he did so for different intensity levels. He then included pre-suppression costs and derived a LC+L curve for each intensity level. Mactivish did not apply his model, but pointed out the data requirements for its application to the Canadian forest sector.

In the 1940s, a set of studies by Craig and others (Craig *et al.* 1945; Craig *et al.* 1946a; Craig *et al.* 1946b) introduced some key changes to the LC+L model. Although their studies were primarily empirical, the theoretical contributions to the model were substantial. They were the first to estimate damages for multiple resources and one of the very few at the time to include values other than timber. In addition, they recognised the need for marginal analysis and suggested that additional investments in pre-suppression and prevention should only be done if they result in (at least) an equivalent reduction in losses (suppression plus damages).

However, early on, some authors also recognised that there were some limitations in the use of the LC+L model. Beall (1949) considered that the data available at the time and the knowledge of the factors studied were insufficient to be able to deduce practical objectives from the LC+L model. Hornby (1936) found that it was very difficult to link pre-suppression effort with damages. However, he believed that the LC+L method was sound and suggested experimenting with different levels of pre-suppression, suppression and damages to find the combination that produced the LC+L figure. Similarly, Davis (1971) pointed out at the difficulties in estimating the changes in damages following variations in costs. Davis drew attention to the fact that up to that point most studies had only included market commodities in the estimation of damages and very little attention had been given to intangible damages. Simard (1976) found that there was little consistency among previous studies in the definition of the strategies evaluated. Like Beall (1949) and Davis (1971), he highlighted the problem of linking strategy costs with damages and pointed out that the relationship between the cost and damage functions and their production functions had never been explicitly investigated.

In an effort to examine the relationship between the cost and damage functions, Simard (1976) developed a hypothetical production function for area burned depending on fire management (which corresponded to fire suppression in his model) and related this curve to the damage curve and the cost curve. He also highlighted the importance of applying marginal analysis and equating marginal costs with marginal benefits to find the optimal investment in fire suppression. In addition, Simard took into account the beneficial effects of fire in his model, thus adopting a more comprehensive approach than any other model to date.

Simard pointed out some problems that require the model to be modified and had not been examined in previous literature: (1) the indivisibility of resources: since a manager cannot dispatch half an airtanker, the cost function actually increases in discrete steps and not in a straight line. Therefore, unless the optimal level of fire management effort is equal to one of these discrete steps, there will always be some inefficiency. (2) Imperfect information: at no time do fire managers know everything about a fire situation. When faced with imperfect information, managers tend to assume the worse for a given situation and their decisions result, on average, in greater expenditures than what would be spent if perfect information were available. (3) Risk: there is always the possibility that something unexpected will happen, such as spotting (i.e. embers flying ahead of the fire front and starting new fires), or injures to personnel or equipment breakdown, which happen with a given probability. As in the case of imperfect information, this usually results in higher expenditures in fire management. Nonetheless, as Simard explains, the fact that fire managers tend to spend more than what the economic theory indicates does not invalidate the theory, but it means that these issues need to be taken into account when solving for the optimum level of fire management effort.

2.1.2. The Cost Plus Net Value Change model and its reformulation

Fires do not only cause losses. They also regenerate the vegetation and are necessary for some species. In the 1980s it was recognised that the beneficial effects of fire needed to be accounted for in the economic model. In an effort to include these benefits and incorporate non-market values, the LC+L model was modified and renamed the cost plus net value change (C+NVC). In the C+NVC the damages curve is net of benefits from wildfires (Rideout and Omi 1990; Gonzalez-Caban 2007). From its initial emphasis on pre–suppression, the C+NVC model has been extended to deal with all aspects of fire management (Gebert *et al*. 2008). Today, the C+NVC is the most accepted model of fire economics (Gebert *et al*. 2008).

Since the LC+L was formulated as the C+NVC in the 1980s, the model has gone through some further reformulation work. Research to improve various aspects of the model have focused on rewriting the model as a maximisation framework (Rideout and Omi 1990), reformulating the relationship between pre–suppression and suppression (Donovan and Rideout 2003b; Rideout and Ziesler 2008), redefining suppression as an independent variable (Donovan and Rideout 2003b), improving the efficiency of the fire management mix (Mills 1979; González-Cabán *et al*. 1986; Donovan *et al*. 1999), specifying how optimal suppression budgets are to be used (Donovan and Rideout 2003a), and accounting for long–term effects (Hesseln and Rideout 1999b; Mercer *et al.* 2007).

Rideout and Omi (1990) questioned the use of a minimisation framework. They compared the traditional efficiency criterion of minimisation of the C+NVC with a profit maximisation criterion in order to discern the insights for fire management that can be drawn from each approach. They found that both criteria are identical, but that the traditional formulation of the C+NVC obscures the role of fire benefits. Rideout and Omi modified the expression of the C+NVC in order to clarify the role of benefits by including them in its mathematical expression. Moreover, their analysis suggest that there are input substitution effects between pre-suppression and suppression that would need further research to be clarified.

Donovan and Rideout (2003b) reviewed and analysed what they consider inherent errors in Sparhawk's formulation of the LC+L model that have been perpetuated into the C+NVC model. They identified two major errors: (1) suppression expenditure is modelled as an output of the model, and (2) suppression and pre–suppression expenditures are incorrectly modelled as negatively correlated. Donovan and Rideout (2003b) argued that both pre–suppression and suppression should be modelled as independent inputs because decisions on suppression investments are made independently of the level of pre-suppression applied. In fact, fire managers would probably not limit their suppression expenditures to the optimal level suggested by economic models, especially if they face an unusually difficult fire season. In most cases, fire managers show an aversion to loss and avoid risk more than is socially or ecologically optimal in the long term (Calkin *et al*. 2005; Wilson *et al*. 2011). As a result, they usually overinvest in suppression. Hence, without taking fire managers' risk aversion into account, the C+NVC model may indicate suppression levels that are actually hardly ever implemented. Regarding the second concern identified, Donovan and Rideout (2003b) explain that suppression and pre-suppression are not necessarily always negatively correlated and that the results obtained under such an assumption either do not provide a minimum in the C+NVC curve or may indicate an erroneous optimal level of pre-suppression effort.

Rideout and Ziesler (2008) identified three common misperceptions in wildfire management that resulted in the misapplication of the C+NVC theory. The three misperceptions are: (1) fuel treatments always reduce optimal fire-fighting effort, (2) with initial attack as the independent variable, the minimum of the C+NVC provides an answer to the fire-fighting problem, and (3) high initial attack success rates are preferable to low rates of success. However, in their analysis of these misperceptions they examine the case where fire-fighting effort is the independent variable and net value change (NVC) and extended attack are the dependent variables. In their analysis, pre-suppression costs (other than initial attack) are not included, and their conclusions could be very different if they had included the costs of pre-suppression activities such as risk reduction efforts and fuel treatments.

While the C+NVC model indicates the most efficient level of pre-suppression, it does not specify the specific mix of pre-suppression resources. To apply it at an operational level, it is necessary to know both the most efficient level of pre-suppression and the corresponding optimal mix of resources. Mills (1979) and Gonzalez-Caban *et al*. (1986) recognised the importance of optimising the mix for pre-suppression resources and developed methods to translate optimal pre-suppression levels into numbers of fire-fighting inputs for initial attack for a specific fire service area. In these models, only initial attack is evaluated and other pre-suppression activities such as prevention, detection and fuel management are not examined, but a similar process can be used to determine the optimal mix of resources for these activities.

Later, Donovan *et al*. (1999) suggested improvements for the resource mix optimisation of the National Fire Management Analysis System (NFMAS). They proposed a system where it is possible to vary one input while holding all the others constant and where the total suppression budget (including initial and extended attack) can be fixed. With a fixed suppression budget, initial attack resources can then be ranked in order of importance to determine which resources should be deployed first given a particular budget. In further work on the mix of fire-fighting resources for extended attack, Donovan and Rideout (2003a) proposed an integer programming model to optimise resource allocation for wildfire suppression. They applied the integer programming technique to a single event and found that a specific fire-fighting mix that minimises the C+NVC within realistic fire management constraints could be identified. The model can be extended to include multiple fire events over a period of time (e.g. a complete fire season), but uncertainties regarding future fire events would add complexity to the analysis.

Besides these, other aspects of the C+NVC model have also been found to be limiting. For instance, the annual budgeting approach of the model can be very restrictive. Fire scars, whether caused by fuel reduction treatments or unplanned fires, affect fire behaviour and intensity for longer than one year. Likewise, natural fuels cumulate over time and if left undisturbed, they gradually increase the risk of catastrophic fires year by year. Hence, by analysing a single fire season, the long-term effects of natural fuel accumulation processes, unplanned fires, fuel treatments and land management strategies on wildfire risk are overlooked (Hesseln and Rideout 1999b). However, most of the theoretical work on the C+NVC and the applications of the model do not address long-term sustainability of fire management programs. To date, only three studies have directed their efforts towards incorporating long-term effects in the economic model: Hesseln and Rideout (1999b), Mercer *et al*. (2007), and Butry *et al.* (2010).

To address the long-term effects of wildfire management, Hesseln and Rideout (1999b) proposed a theoretical extension of the C+NVC model using control theory. They explored the long-term relationships between fire management activities and physical and economic damage and investigated the suitability of control theory for the development of a long-term optimisation model. Control theory is used to find solutions to optimisation problems where decisions are related through time. It recognises that the choice of a decision variable in one time period affects future choices of that decision variable, and the objective is to determine the optimal path of decision variables over a specified period of time. Since fire management decisions are dynamic, the principles of control theory may be applied to fire management in order to find optimal paths of pre-suppression and suppression over a time-period that result in an optimal path of NVC. Hesseln and Rideout (1999b) linked NVC with wildfire area and fire-line intensity, which allowed them to incorporate the erratic and unpredictable characteristics of fire behaviour in their model and also include ecological and environmental factors.

Mercer *et al*. (2007) made several theoretical improvements to the model. First, they adopted a long-term approach. Their model estimates the long-term economic consequences of alternative prescribed-burning policies under different climate scenarios in Volusia County (Florida, US). Second, they used a new measure of wildfire output to estimate the damages. Instead of using only area burned to calculate expected wildfire damages, the authors used intensity-weighted area burned, which is more directly linked to economic damages from wildfire. And third, they compared the distributions of the welfare criterion for different prescribed burning strategies instead of comparing the mean, which is the usual approach of the C+NVC. They compared the results using the new measure of wildfire output with the results obtained with the area burned measure and concluded that ignoring intensity levels can lead to a suboptimal application of prescribed burning.

Butry *et al*. (2010) used long-run model that evaluates the effects of prescribed burning on wildfire ignition risk and area burned, and the effects of wildfire prevention education on the ignition risk of accidental human ignitions. Their model put together the expected ignitions and the expected fire size in several time-periods and estimated the changes due to prescribed burning and education programs.

Some critical elements for wildfire management, however, have not been addressed in the literature on wildfire economics and the reformulation of the C+NVC; for instance, the growth of the wildland-urban interface (WUI) and the mega-fire phenomenon.[4] The WUI has become a major challenge for fire managers and policy makers (Marzano *et al*. 2008). Indeed, wildfire

[4] The wildland-urban interface (WUI) is broadly thought of as the area of transition between urban areas and unoccupied land (rural or wildland areas) where flammable vegetation intermingles with human built structures. Although the term is often used in fire economics and fire behaviour research, its definition varies greatly in the literature (Smalley 2003; Tolhurst *et al*. 2013). In this study, the WUI is defined as the zone where a well-defined urban area (i.e. house density greater than one house per hectare) meets a rural-residential area (house density between one and 16 houses per hectare), or a wildland area (house density less than one house per 16 hectares) (house densities for the different areas are defined in Tolhurst *et al*. 2013). It includes the properties within 50 metres of this separation.

risk reduction is more complicated and costly in the WUI than in wildland areas, because of the spatial interactions between housing and fuels (Bar Massada *et al*. 2011). With the expansion of the WUI, fire agencies have increasingly directed their efforts towards the protection of life and property from catastrophic fires (Calkin *et al*. 2005; Wei *et al*. 2008) and are likely to continue to do so (Smalley 2003; Morgan *et al*. 2007; Mell *et al*. 2010; Mutch *et al*. 2011). But with more and more houses located in areas of flammable vegetation, objectives of life and property protection become more difficult to achieve. The development of the WUI also requires a long-term perspective. Thus, optimal fire management strategies need to reflect these developments and balance short- and long-term objectives of fire prevention and protection for WUI areas. The C+NVC model needs to be extended to account for short-term and long-term fire protection objectives in an increasingly complex fire-fighting environment with private development likely to continue expanding in fire prone areas.

The mega-fire phenomenon also needs to be investigated from an economic perspective. However, applications of the C+NVC model in its current expression use simulations of average fire seasons. While modelling average fire seasons can provide valuable insights into the fire management problem, the issue of catastrophic fires should not be overlooked. It is the non-average fire seasons that most influence costs and damages estimates (Handmer and Proudley 2008). In Australia and elsewhere, the frequency of disastrous fires appears to follow an increasing trend (Morgan 2009) and this tendency could be enhanced by an increase in the number of extreme weather days owing to climate change. Hence, the risk of rare and extreme events in the long run needs to be incorporated in the model.

There is a great need for more empirical applications of the model. Despite the abundance of theoretical studies on the subject, integrated economic assessments of wildfire management that compare the costs and benefits of different management options remain scarce (Mercer *et al*. 2007). Although the number of empirical studies has considerably increased in the last two decades, the vast majority of these studies has concentrated on either the costs of fire management, or the benefits of fire management, or the consequences of wildfires. The remaining sections of this Chapter review the empirical literature on wildfire economics, which has been here classified according to the five major areas of research focus: costs, benefits, wildfire consequences, integrated assessments and incentive structures.

2.2. The costs of fire management

The costs of fire management activities have been relatively well covered in the fire economics literature. Most of the studies analysing fire management costs have focused on pre-suppression activities, dealing primarily with prescribed burning (Gebert *et al*. 2008) and sometimes also including mechanical fuel reduction (see Table 2.2). Other pre-suppression activities, such as community education and fire detection systems, have been largely overlooked. A handful of studies have paid attention to the costs of suppression, analysing suppression expenditure trends and attempting to forecast wildfire suppression costs. Practically all the research has been done in North America, with only two exceptions in Spain and Australia (see Table 2.2).

This section on the costs of fire management is divided in two subsections: first, the literature on the costs of pre-suppression activities is reviewed, then the second subsection focuses on the costs of suppression.

2.2.1. The costs of pre-suppression activities

Studies analysing the costs of pre-suppression activities have worked primarily on estimating per acre costs of fuel treatments (Hartsough *et al*. 2008; Rodriguez y Silva 2008) or identifying the factors that influence these costs (Rich 1984; Rideout and Omi 1995; Gonzalez-Caban 1997; Cleaves *et al*. 1999; Berry and Hesseln 2004; Berry *et al*. 2006; Calkin and Gebert 2006). Most of these analyses identifying the factors that influence the costs of fuel treatments have used regression models, testing the significance of different factors, from biophysical and topographic elements, to managerial and scale factors. As mentioned before, most of this research on fuel treatment costs has focused on prescribed burning. These studies have been conducted in a variety of landscapes and management environments, but they all seem to agree in a number of points: prescribed-burning costs per unit area are lower for larger treatments and higher in the WUI, are influenced by both physical-topographic and managerial factors, and are very variable.

First, per acre costs of fuel treatments decrease as the size of the treatment area increases (Rideout and Omi 1995; Cleaves *et al*. 1999; Berry *et al*. 2006; Calkin and Gebert 2006). For instance, Cleaves *et al*. (1999) found prescribed-burning costs decrease with larger-scale burning. They extracted data on costs and cost influences from a survey conducted to characterise and quantify prescribed burning activity in the Forest Service's National Forest System between 1985 and 1994. The authors included questions in the survey that required survey participants to rank the importance of selected resource targets and cost influences. With this information they listed treatment size as the most important determinant of per acre costs of prescribed burning. Berry *et al*. (2006) also found that per acre costs of prescribed burning decrease with treatment size. In a study conducted to analyse the effects of the WUI on prescribed-burning costs, they also included treatment size as one of the explanatory variables in their model. Their results indicate that the number of acres burned in a prescribed burning treatment is negatively correlated with per acre costs of prescribed burning.

Rideout and Omi (1995) found that per acre costs decrease with treatment size, but this decrease is not linear. They examined fuel treatments cost data for the continental US obtained from the National Park Service database and used multiple regression analysis to test the hypothesis that a constant cost elasticity can be applied to fuel reduction treatments. This implies that the slope of the cost-per-acre curve depends on the number of acres treated. They confirmed this hypothesis and explained that while cost per acre decrease with increases in treatment area, changes in cost per acre with acres treated are greater for small treatments than for large treatments, so the importance of scale depends on the level of scale.

Table 2.2. Summary of empirical studies relating to fire management by research focus

Area of research focus*		Publications[ϒ]
Pre-suppression	Fuel treatments	Bar Massada *et al*. (2011); Berry and Hesseln (2004); Berry *et al*. (2006); Butry *et al*. (2010); Calkin and Gebert (2006); Cleaves and Brodie (1990); Cleaves *et al*. (1999); Drucket *et al*. (2008); Dyer *et al*. (2003); Gibson and Pannell (2014); Gonzalez-Caban (1997); Gonzalez-Caban and McKetta (1986); Hartsough *et al*. (2008); Heckbert *et al*. (2012); Hesseln (2000); Jackson *et al*. (1982); Jones *et al*. (1999); Loomis *et al*. (2003); Lui *et al*. (2010); Mercer *et al*. (2007); Mercer *et al*. (2008); Omi (2008); Omi *et al*. (1999); Penman *et al*. (2014); Prestemon *et al*. (2001); Prestemon *et al*. (2012); Rich (1984); Rich (1989); Rideout and Omi (1995); Rodriguez y Silva (2008); Stockmann (2007); Stockmann *et al.* (2010a); Stockmann *et al.* (2010b); Wei *et al*. (2008); Yoder (2004)
	Education	Butry *et al*. (2010); Gibson and Pannell (2014); Prestemon *et al*. (2010)
	Detection	Steele and Stier (1998)
	Initial attack	Gonzalez-Caban *et al*. (1986); Mills (1979)
Suppression		Abt *et al*. (2009); Arnold (1950); Calkin *et al*. (2005); Christman and Rollins (2015); Donahue (2004); Donovan and Rideout (2003b); Donovan and Rideout (2003a); Donovan *et al*. 1999; Donovan *et al*. (2008); Donovan *et al*. (2011); Gebert and Schuster (1999); Gebert *et al*. (2007); Gonzalez-Caban (1984); Gude *et al*. (2013); Hesseln *et al*. (2010); Liang *et al*. (2008); Mactavish (1965); Mees *et al*. (1994); Parks (1964); Petrovic and Carlson (2012); Prestemon *et al*. (2008); Thompson *et al.* (2013b); Thompson *et al*. (2015)
Benefits of implementing fire management activities		Fried *et al*. (1999); Gonzalez-Caban *et al*. (2007); Gonzalez-Caban *et al*. (2013); Kaval (2009); Kaval and Loomis (2008); Kaval *et al*. (2007); Loomis and Gonzalez-Caban (1994); Loomis and Gonzalez-Caban (1997); Loomis and Gonzalez-Caban (1998); Loomis and Gonzalez-Caban (1999) ; Loomis *et al*. (2004); Loomis *et al*. (2005); Loomis *et al*. (2008); Loomis *et al*. (2009); Loureiro *et al*. (2004); Mavsar and Farreras (2009a); Mavsar and Farreras (2009b); Walker *et al*. (2007); Winter and Fried (2001)

[ϒ] This table includes theoretical, empirical, review and discussion publications.
* Studies that can be classified into two areas of research focus have been included in both (e.g. integrated economic assessments that investigate changes in fuel treatment investments have been included in two areas of research focus: fuel treatments and integrated assessments).

Table 2.2 (contd.)

Area of research focus*	Publications[ϒ]
Wildfire impacts	Boxall *et al*. (1996); Butry *et al*. (2001); Calkin *et al*. (2008); Cardoso de Mendoza *et al*. (2004, 2006); Englin *et al*. (1996); Handmer and Proudley (2008); Hesseln *et al*. (2003); Hesseln *et al*. (2004a); Hesseln *et al*. (2004b); Kochi *et al*. (2010); Loomis (2004); Loomis *et al*. (2001); Lynch (2004); Mueller *et al*. (2009); Richardson *et al*. (2012); Rideout *et al.* (2008); Rittmaster *et al*. (2006); Stetler *et al*. (2010)
Integrated assessments	Bar Massada *et al*. (2011); Bennetton *et al*. (1998); Butry (2009); Butry *et al*. (2010); Craig *et al*. (1945); Craig *et al*. (1946); Dale (2006); Davis (1971); Drucket *et al*. (2008); Dyer and Stafford Smith (2003); Flint (1928); Gibson and Pannell (2014); Headley (1916); Heckbert *et al*. (2012); Hornby (1936); Jones *et al*. (1999); Liu *et al*. (2010); Loomis *et al*. (2003); Lovejoy (1916); Mercer *et al*. (2007); Mercer *et al*. (2008); North *et al*. (2012); Omi (2008); Omi *et al*. (1999); Pedernera *et al*. (2008); Prestemon *et al*. (2001); Prestemon *et al*. (2010); Prestemon *et al*. 2012; Rodriguez y Silva and Gonzalez-Caban (2010); Schaaft *et al*. (2008); Simard (1976); Snider *et al*. (2006); Sparhawk (1925); Stockmann *et al.* (2010a); Stockmann (2007); Wei *et al*. 2008; Yoder (2004)
Incentive structures and risk attitudes	Busby and Albers (2010); Busby *et al*. (2013); Crowley *et al*. (2009); Donovan and Brown (2005); McFarlane *et al*. (2011); Thompson *et al.* (2013a); Wibbenmeyer *et al*. (2013); Yoder and Blatner (2004); Yoder *et al.* (2003b)
Other (review and discussion papers, or theoretical papers with multiple economic research focuses)	Clayton *et al*. (2014); Donovan and Rideout (2003b); Ganewatta (2008); Gebert *et al*. (2008); Gonzalez-Caban (2007); Hesseln (2000); Hesseln and Rideout (1999a); Hesseln and Rideout (1999b); Hesseln *et al*. (1998); Kochi *et al*. (2010); Marzano *et al*. (2008); Mell *et al*. (2010); Milne *et al*. (2014); Morgan (2009); Morgan *et al*. (2007); Mutch *et al*. (2011); Rideout and Ziesler (2008); Rideout and Omi (1990); Smalley (2003); Thompson and Anderson (2015); Wilson *et al*. (2011)

[ϒ] This table includes theoretical, empirical, review and discussion publications.
* Studies that can be classified into two areas of research focus have been included in both (e.g. integrated economic assessments that investigate changes in fuel treatment investments have been included in two areas of research focus: fuel treatments and integrated assessments).

Many studies have examined the costs of prescribed burning but mechanical treatments have generally been overlooked. One study found similar cost results for mechanical treatments and prescribed burning. Calkin and Gebert (2006) developed regression models to estimate the costs of fuel reduction treatments based on biophysical factors, treatment type, treatment objective, and USDA Forest Service management region. Like Cleaves *et al*. (1999), Calkin and Gebert (2006) also surveyed fuel managers in the Forest Service to obtain cost data on fuel treatments. Using this data, they examined which factors influence the costs of different types of fuel treatments and found that treatment size explained most of the variation in costs per acre for both mechanical removal and prescribed burning treatments. Their results show that on average, per acre costs diminish with increased treatment size: a 1 percent increase in treatment size reduces costs per acre by 0.35 percent for prescribed burning and by 0.30 percent for mechanical treatments.

Second, prescribed burning treatments in or close to the WUI have higher per acre costs than treatments in wildland areas (Berry and Hesseln 2004; Berry *et al.* 2006; Calkin and Gebert 2006; Hesseln *et al.* 2006). Wildfire risk reduction is more complicated in the WUI than in wildland areas, because of the spatial interactions between housing and fuels (Bar Massada *et al*. 2011). For treatments within the WUI, there are usually concerns regarding air quality, aesthetic appearance, structure protection and risk that make them more expensive than wildland treatments (Berry and Hesseln 2004). A few studies have analysed the effects of the WUI on per acre costs of fuel reduction treatments. Berry and Hesseln (2004) analysed cost data for prescribed burns and mechanical treatments in the Pacific Northwest Region of the USDA Forest Service. They developed a regression model for each type of fuel reduction and showed that treating fuels within the WUI significantly increases both prescribed-burning costs and mechanical treatment costs. According to Berry and Hesseln's (2004) model, the costs of mechanical treatments within the WUI are almost four times greater than in wildland areas and the costs of prescribed burning inside the WUI are about 43 percent more expensive than outside the WUI.

Berry *et al*. (2006) and Hesseln *et al.* (2006) also used regression analysis to examine which factors influence prescribed-burning costs in the States of Washington and Oregon in the US and estimate the effect of the WUI on prescribed-burning costs. Their results show that when prescribed burning is applied in the WUI, ceteris paribus, costs increase by 139 percent relative to wildland areas.

Using survey data on fuel treatment costs, Calkin and Gebert (2006) reached similar conclusions, although with smaller percentage changes. In their study, they also investigated the effect of the WUI on the costs of both prescribed burning and mechanical treatments.

Their results indicate that prescribed burning treatments located within the WUI are 34 percent more expensive than treatments located in wildland areas. Mechanical treatments were found to be 62 percent more expensive within the WUI compared to treatments outside the WUI.

Third, prescribed-burning costs are commonly influenced by a number of key physical factors, which usually include slope, elevation, fire regime, fuel type and prevalent weather conditions (Cleaves and Brodie 1990; Rideout and Omi 1995; Berry *et al*. 2006). In an economic analysis of prescribed burning, Cleaves and Brodie (1990) determined that fuels, topography, and weather had a significant influence on prescribed-burning costs. As the terrain gets more difficult to access in places with high elevation and steep slopes, additional time and equipment are required to access and monitor prescribed burning treatments, hence per acre costs of prescribed burning usually increase with slope and elevation (Rich 1984; Cleaves and Brodie 1990). Rich (1984) analysed prescribed-burning costs in the Powell District of the Clearwater National Forest, Idaho, in 1982. He found that prescribed-burning costs increase with increases in elevation and concluded that elevation was the most important physical site factor influencing costs. Berry *et al*. (2006), however, obtained the opposite result and found that each increase of 100 feet in elevation reduced prescribed-burning costs by 1.6 percent. But they found a positive correlation with slope where each additional degree in slope increased the costs of treatments by about one percent. Berry *et al*. (2006) also found other factors, such as fuel type and the fire regime applied in the area to significantly influence per acre costs of prescribed burning. Similarly, Rideout and Omi (1995) found that fuel type significantly affected costs per acre.

Fourth, fuel reduction costs are also influenced by other non-physical factors such as managerial factors and land-use types. Several studies have found managerial variables to be significant determinants of fuel-reduction costs. For instance, Gonzalez-Caban (1997) found that managerial and institutional factors significantly affect prescribed-burning costs. Similarly, Jackson *et al*. (1982), in a study examining the costs of prescribed-burning projects that have as primary goal the improvement of wildlife habitat, found managerial factors to be significant. Cleaves and Brodie (1990) showed that the management objective selected significantly influences the costs of prescribed-burning projects. They also lay emphasis on risk and how risk perceptions can influence prescribed-burning costs, including the risk of escape, the risk of smoke affecting surrounding areas, and fire managers' risk attitude. The treatment method selected, which is also part of a managerial decision, has been shown to significantly affect per acre costs of prescribed burning (Rideout and Omi 1995; Berry *et al*. 2006). Similarly, Berry and Hesseln (2004) found that primary project objectives, another managerial variable, are an important determinant of per acre costs of prescribed burning. The fire managers surveyed by Cleaves *et al*. (1999) reported safeguards to minimise escaped fires and compliance with environmental laws as important determinants of prescribed-burning costs per acre.

In some cases managerial factors have been found to influence costs more than physical factors (Rich 1984, 1989). In his 1984 study of prescribed-burning costs in the Powell District, Rich investigated the impact of both physical site characteristics and managerial factors on prescribed-burning costs and concluded that managerial variables, such as decisions regarding the size of the treatment and the mop-up category, are more important.[5] This conclusion was reinforced by his 1989 study where he analysed similar information from the Powell District between 1982 and 1985.

In addition to managerial factors, land-use type can also affect fuel reduction costs. For instance, when fuel treatments are applied in designated protection areas, the costs per acre increase (Berry and Hesseln 2004). Berry and Hesseln (2004) found that mechanical treatments are 60 percent more expensive in designated protection areas than treatments in non-protected areas, and per acre costs of prescribed burning are 35 percent higher in designated protection areas than in non-protected areas.

Finally, prescribed-burning costs vary greatly from one place to another (Rideout and Omi 1995; Cleaves *et al*. 1999; Berry *et al*. 2006; Hartsough *et al*. 2008). Because prescribed-burning costs are affected by local site characteristics, the costs can considerably vary from one landscape to another. For instance, Berry *et al*. (2006) reported that operating costs varied between the east and west slopes of the Cascade Range, being 34 percent lower on the east slopes. In the survey Cleaves *et al*. (1999) conducted, they obtained cost data from 95 of the 114 National forests in the Forest Service's National Forest System, covering all nine different Regions. From the information reported by fire managers, the authors found that per acre costs of prescribed burning varied greatly between regions, ranging between US$11 and $270 per acre. Rideout and Omi (1995) found that per acre costs of prescribed burning were

[5] Although in most studies, treatment size has been considered a physical characteristic, Rich (1984) argued that it should be considered a managerial decision, because the size of a treatment is decided by fire managers and not mandated by the physical environment. For Rich (1984), physical factors correspond to factors such as slope, elevation, aspect, fuel age in the treatment unit, fuel age around the treatment unit, distance from the district office to the site by road, and cover type.

significantly influenced by location variables such as the region where the treatment was applied. Hartsough *et al*. (2008) collected data on the costs of different types of fuel reduction operations at seven experimental sites in the western US. They reported very different costs per hectare between these seven sites, ranging between US$310 and $1,210 for prescribed burning and between $1,730 and $5,150 for mechanical treatments. In addition, they compared these costs with published costs in other areas and found that costs for treatments involving prescribed fire are generally higher in the west than other regions of the US.

The relative economic performance of prescribed burning

Because of the ecological differences between landscapes and the differences in costs of treatments, the optimal type of treatment and treatment size (i.e. the treatment that simultaneously minimises costs of treatment and wildfire risk) can vary from one area to another. Rodriguez y Silva (2008) investigated this issue to help with decision making for fuels management in Mediterranean ecosystems. He conducted a comparative cost analysis of mechanical treatments and prescribed burning, emphasising the attributes of each type of fuel reduction treatment based on fuel modelling criteria. In this study, Rodriguez y Silva (2008) first classified the prescribed burning management units of the study region in Andalusia, Spain, according to their physical characteristics to determine the treatment that would more closely meet the conditions required to proceed safely and efficiently with the treatment.[6] In addition, he developed a methodology to identify the ideal forest areas for the use of prescribed burning. For the different fuel categories, he then ascertained the ideal burning conditions, ignition methods, and size of treatment, and with the cost data he then determined the cost-effective burn area for each fuel category. In locations where the ecological conditions of the forest are suitable for prescribed burning, Rodriguez y Silva (2008) showed that prescribed burning had an advantage over mechanical methods.

Hartsough *et al*. (2008) compared the costs of prescribed burning and mechanical treatments. In addition, the authors analysed the effects of each treatment on potential fire behaviour in order to estimate their capacity for reducing wildfire severity and determine the optimal treatment for each area. They found that the costs per hectare of mechanical treatments were substantially higher than prescribed burning, but since some of the material cut could be

[6] The physical characteristics included elements such as the existence of a timber canopy, the horizontal continuity of the fuel, the diameter of the fuel and the vertical continuity between the distinct vegetation strata.

recovered and generate revenue, the net costs per hectare of mechanical treatments were actually lower than prescribed burning. The results from the study sites indicated that all types of treatment mitigated potential fire severity to some degree, but the authors found that the most effective treatment for reducing fire severity was a combination of both treatments, where prescribed burning is applied after mechanical treatments in the same site. The second most effective treatment was prescribed fire only.

2.2.2. The costs of suppression

The study of suppression expenditures was pioneered by Gonzalez-Caban (1984), who estimated the costs of mopping-up activities, which correspond to the suppression activities that start when the fire has been contained. Since then, work on the costs of suppression has focused on analysing expenditure trends (Calkin *et al.* 2005; Thompson *et al.* 2013b), identifying factors that influence suppression expenditures (Gebert *et al*. 2007; Donovan *et al*. 2008; Liang *et al*. 2008; Donovan *et al*. 2011; Gude *et al*. 2013), investigating options for a more efficient allocation of suppression resources (Donovan and Rideout 2003a; Hesseln *et al.* 2010), and forecasting suppression spending (Gebert and Schuster 1999; Prestemon *et al*. 2008; Abt *et al*. 2009). In the same way as the pre-suppression costs literature, most of the studies analysing suppression costs have developed regression models to test the significance of different factors and also to predict suppression expenditures. For this area of study, all of the published research has been conducted in North America.

One of the most recurring statements in the literature examining suppression costs, which has often been the impetus for the application of economics to fire management, is that suppression expenditures have increased substantially since the mid-1980s (Omi *et al.* 1995; Calkin *et al.* 2005; Liang *et al.* 2008; Prestemon *et al.* 2008; Donovan *et al.* 2011; Thompson *et al.* 2013a). In the US for instance, the Forest Service suppression expenditures increased from less than 400 million US dollars per annum during most of the 1970s to over a billion dollars (in real terms) in nearly all years since 2000 (Calkin *et al*. 2005; Prestemon *et al*. 2008). Aviation resources, which are the most expensive suppression resources and account for a large share of suppression expenditures, have been used extensively by fire managers and their costs are increasing (Thompson *et al.* 2013b).

In view of the dramatic increase in the United States' federal expenditures on suppression in recent years, Liang *et al*. (2008) mentioned that "there is an urgent and immediate need to address the excessive cost of large fires" (p. 650). But to address those costs, it is necessary to know what drives suppression expenditures and identify the areas where government efforts should focus to contain suppression costs.

One element that has been identified as a significant driver of suppression costs is fire size (Calkin *et al*. 2005; Gebert *et al*. 2007; Donovan *et al*. 2008; Liang *et al*. 2008). Donovan *et al*. (2008) examined factors that influence suppression costs in the States of Oregon and Washington. Their regression analysis of a sample of 58 fires occurring in 2002 showed that fire size was the most significant determinant of suppression costs. Calkin *et al*. (2005) examined data from the US Department of Agriculture Forest Service on suppression expenditures, number of wildfires and area burned by wildfires between 1970 and 2002. They concluded that the increase in total suppression expenditures appears to be associated with increases in area burned and not with suppression costs per acre, which remained constant during the period studied. Using a regression model, Gebert *et al*. (2007) also identified area burned as one of the variables having a significant influence on suppression costs. Liang *et al*. (2008) tested 16 non-managerial factors that could influence suppression expenditures by the US Department of Agriculture Forest Service, including wildfire size and shape, private properties, public land attributes, forest and fuel conditions, and geographic settings. Their regression shows that fire size has a significant effect on suppression expenditures and that as the area burned by wildfires increases, suppression cost increase monotonically.

In addition to fire size, private land or proximity to houses have also been found to have a strong effect on suppression expenditures (Gebert *et al*. 2007; Liang *et al*. 2008; Gude *et al*. 2013). Gebert *et al*. (2007) regression models identified total housing value within 20 miles (32 kms) of the ignition point as one of the variables that has a significant effect on suppression costs in Regions 1 to 6 of the National Forest System managed by the USDA Forest Service, and the distance to the nearest town was a significant variable in Regions 8 and 9. Liang *et al*. (2008) found that private ownership of land significantly affects suppression expenditures per fire, even if the proportion of private land burned is relatively small. However, their results show that suppression costs per fire do not increase monotonically with the percentage of private land burned by wildfires. First, suppression expenditures per fire increase with the percentage of private land burned, then decrease slightly when the percentage reaches 20 percent, and then stabilise after 50 percent of private land burned. In a more recent study, Gude *et al*. (2013) adopted a different approach and analysed suppression costs at the daily level (instead of using aggregated data for each fire event). They used linear mixed models to estimate the effect of the number of homes on daily suppression expenditures. In accordance with studies using aggregated data on fire suppression, they found that an increase in the number of houses within 6 miles (9.7 kms) of the active fire increases daily suppression costs. The only study analysing the effect of housing on suppression costs that did not find a relationship between the number of houses or housing density and suppression cost is Donovan *et al*. (2008). However, the size of sample used in the study and the types of fires chosen for the sample might explain the difference in their findings. They only tested 58 fires occurring in one single year and all of the fires in their sample pose a threat to housing, since the region is sufficiently densely populated that all ignitions may potentially burn houses. Therefore, the effects of housing may not be detected in their study if there are no fires in the sample for which the costs are not already strongly affected by housing.

Other factors can also affect suppression expenditures, such as extreme terrain conditions (Gebert *et al*. 2007; Donovan *et al*. 2008; Donovan *et al*. 2011), fire intensity level and fuel type (Gebert *et al*. 2007), and managerial factors (Donahue 2004), but none of these variables are as significant as fire size and private ownership of land (Liang *et al*. 2008). According to Liang and colleagues, this can be explained by the fact that fuel type and terrain conditions indirectly affect suppression expenditures through fire size, because these variables determine the extent of wildfires. An important finding of Liang and colleagues' study is that it shows that when fire size and private land are accounted for, all other variables have no significant effect on suppression costs. In their model, fire size and private land combined explained 58 percent of variation in suppression costs. This has important implications for fire management. It suggests that in order to address the excessive suppression costs of wildfires, fire agencies need to focus on the management of wildfires on private land, which is a complex and politically sensitive issue (Liang *et al.* 2008; Penman *et al.* 2011; Gibbons *et al.* 2012).

This emphasises the challenges of the WUI, which not only relate to the high fire risk of houses that are closely surrounded by flammable vegetation and are more difficult to protect because of the interactions between buildings and fuels (Marzano *et al*. 2008; Bar Massada *et al*. 2011), but also to community and political pressures (Attiwill and Adams 2013). In fact, media coverage of wildfires and political pressure have also been found to significantly affect suppression costs (Donovan *et al*. 2011). Donovan and colleagues used regression models to estimate the effect of newspaper coverage and political pressure on suppression expenditures and found that both variables have a significant effect. Additionally, when political pressure and newspaper coverage were accounted for, other variables relating to housing near the fires lost their significance.

Forecasting suppression costs

One of the major challenges for fire agencies is the prediction of suppression costs. Forecasts of suppression costs are an important part of fire management, but the methods usually adopted by fire agencies yield inaccurate forecasts (Abt *et al*. 2009; Prestemon *et al*. 2008). The USDA Forest service uses the 10-year suppression costs average to calculate the budget for the following year, but there is not a probability of occurrence attached to that forecast. Suppression costs forecasts in the form of probability distributions can provide a better picture of expected suppression costs and the likelihood of unusually expensive fire seasons (Prestemon *et al*. 2008). Because of the lack of use of probabilities for different levels of suppression expenditure, two studies have shown that the estimates obtained using the 10-year average end up being far away from actual expenditures. Abt *et al*. (2009) and Prestemon *et al*. (2008) used regression models to predict suppression expenditures in different regions managed by the USDA Forest Service. They showed that their models' predictions were much closer to actual expenditures than figures obtained using the 10-year average method.

2.3. The non-market benefits of implementing fire management activities

There is a substantial proportion of the fire economics literature that has focused on estimating the non-market benefits of implementing fire management activities in the form of people's willingness to pay (WTP) for publicly funded programs that reduce the risks of wildfires. Nearly all the work in this area has used the contingent valuation method and only a few relatively more recent studies have used choice experiments or the travel-cost method. All of this research has been conducted in the US, with the exception of one single choice experiment study conducted in Europe.

Studies using the contingent valuation method conducted in different locations in the US (Michigan, California, Florida, Montana, Colorado and Oregon) have shown that WUI residents express positive WTP for wildfire risk reduction activities. Usually, respondents are well informed of the risk (Kaval *et al*. 2007; Kaval 2009), they perceived wildfire risk to be high and rank it higher than the risk of other natural hazards (Fried *et al*. 1999). In a study conducted in

Michigan by Fried and colleagues, nearly all respondents were insured against property loss, so the fact that they were willing to pay additional money to reduce the risk of losing their houses to wildfire suggests that there are non-market losses experienced when a house is burned that are not covered by insurance (Fried *et al*. 1999). Although in this case, some of the WTP could be explained by the fact that residents may be expecting a reduction in insurance premiums in the long term because of reduced fire risk, the high amounts respondents are willing to pay for risk reduction activities may still be higher than the reduction in insurance premiums they could get (50% of respondents are willing to pay more than US$400 per annum for risk reduction activities within their properties).[7] Even after accounting for the negative effects of prescribed burning programs resulting from smoke emissions and the willingness to accept compensation for these negative effects, net willingness to pay estimates remain positive (Loureiro *et al*. 2004). A choice modelling study in Spain suggests that people in Catalonia also obtain non-market benefits from additional fire prevention (Mavsar and Farreras 2009a, 2009b).

People's WTP estimates are influenced by several factors, including property values, income, the area burned by wildfire that is reduced by the fuel reduction program, people's previous experiences with wildfire, perceived risk of wildfires, and some spatial variables. Some research has found that the amount that WUI residents are willing to pay for reductions in risk is related to the value of their property (Winter and Fried 2001) and the income of the household (Winter and Fried 2001; Walker *et al*. 2007). In a study using the contingent valuation method to analyse WTP for fuel reduction programs by White and Hispanic households in California, Florida and Montana, Loomis *et al*. (2009) found that people's WTP for reducing area burned by wildfire increases with the size of the fire. In other words, they are willing to pay more for a reduction in the area burned by the largest wildfires. Using a choice experiment, Gonzalez-Caban *et al*. (2013) found that homeowners in Florida with prior experience of wildfire impacts were willing to pay more for protection programs than those without prior experience. Also, people living in areas that they considered at high risk of wildfires showed higher WTP estimates. Kaval *et al*. (2007) used the contingent valuation method to estimate WUI residents' WTP for prescribed burning programs in Colorado. They showed that support for prescribed burning programs depended on respondents' perception of wildfire frequency. Kaval and Loomis (2008) evaluated the influence of spatial forest fire risk variables on people's WTP for fire prevention in Colorado. Their results identified spatial

[7] According to local insurers, homeowners living in the city of Grayling, Michigan (located in the study area for Fried *et al*. 1999), pay an average of $839 per annum for a standard home insurance policy.

variables that have a significant influence on WTP estimates, including having a defendable space within 10 meters of the house and having a high fire danger within 100 meters of the house. Other studies have found that support for prescribed burning depends on the perceived frequency of wildfires or the perceived risk (Kaval *et al*. 2007; Walker *et al*. 2007; Kaval 2009), and proximity to forested areas (Walker *et al*. 2007).

Other forest values, such as preservation of biodiversity and protection of endangered species, and the benefits of using fire management programs to protect them have also been quantified using the contingent valuation method. Loomis and Gonzalez-Caban (1994) estimated households' WTP for fire prevention and control programs that protect critical habitat for the northern spotted owl in Oregon, US. They showed that people have high WTP for programs that reduce fire hazard to the species' critical habitat, and that this translates into an average value of USD$28 per acre protected. In a follow up study in 1997, Loomis and Gonzalez-Caban found that households in California are willing to pay similar amounts to households in Oregon for fire prevention to reduce fire intensity and area burned in spotted owl's habitat. Loomis and Gonzalez-Caban have also shown that the area of habitat protected is a significant variable in the WTP function (Loomis and Gonzalez-Caban 1998). They indicate that fire managers can use these WTP estimates to assess the benefits of fire management programs that achieve additional reductions in area burned (Loomis and Gonzalez-Caban 1998) and have developed tools that fire managers can use to easily calculate these benefits (Loomis and Gonzalez-Caban 1999).

Because households' WTP for fire prevention programs is very similar for populations across different States (Loomis *et al*. 2005), for different ethnic groups (Loomis *et al*. 2004; Loomis *et al*. 2008), and for Native Americans and the general public (Gonzalez-Caban *et al*. 2007), some authors have investigated whether the WTP functions are transferable (e.g. Loomis *et al*. 2005). However, there is mixed evidence of transferability for mean WTP for prescribed burning and mechanical treatments.

2.4. Wildfire impacts on market and non-market values

A portion of the fire economics literature is centred around the estimation of wildfire impacts on market and non-market values (i.e. the proportion of market and non-market goods and services that is destroyed by the burning of forests and property). On this topic as well, most of the research has been conducted in the US, with a few studies conducted in Canada, Australia and south-east Asia. The majority of analyses of wildfire impacts have focused on non-market values, for the most part estimating the effects of wildfires on recreational demand, property values and human health. The methods most commonly used are travel cost and hedonic pricing. Only a few studies have investigated the impacts of fires on market goods and services.

Forest fires have been shown to have a significant impact on recreation demand. Two studies using the travel cost models analysed the effects of forest fires on canoeing in Canada (Boxall *et al*. 1996; Englin *et al*. 1996). They revealed that canoers are willing to pay to avoid fire-damaged forests (Boxall *et al*. 1996) and the amount of forest burned in the last 10 years significantly influenced the chosen canoeing route (Englin *et al*. 1996). Also using the travel cost method, several studies have evaluated the effects of fire (both prescribed burning and wildfires) on hiking and biking demand in the US (Loomis *et al.* 2001; Hesseln *et al.* 2003; Hesseln *et al.* 2004a, 2004b). They have found that hiker's demand decreases slightly in areas recovering from wildfires in Colorado and Montana (Hesseln *et al*. 2004Montana, Hesseln *et al*. 2004a) but is not statistically significant (Loomis *et al*. 2001), whereas the presence of crown fires had a significant and negative effect on mountain biking demand (Hesseln *et al*. 2004b; Loomis *et al*. 2001). In contrast, in New Mexico, both user groups' demand decreases in areas recovering from wildfires (Hesseln *et al*. 2003). Also, demand by both user groups decreases as area burned increases (Hesseln *et al*. 2004b). Moreover, the effects of wildfire on annual recreation values changes with time since wildfire, but the evolution of recreational values with time after the fire is non-linear (Englin *et al*. 2001).

Wildfires can also reduce property values or decrease homebuyer's demand for homes. Using the hedonic price framework, Stetler *et al*. (2010) and Loomis (2004) found that following a wildfire, there is statistically significant and persistent negative effect on property values. Repeated wildfires cause a decrease in house prices in the surrounding areas, but not at the same rate; the first fire reduces property values by 10 percent, while the second fire reduces property values by 23 percent (Mueller *et al*. 2009).

The health effects of wildfire can also be significant, but there are very few economic studies that estimate them and there is a need for more research in the area (Kochi *et al*. 2010). For instance, Rittmaster *et al*. (2006) found that the health effects from forest fires are substantial and second only to timber losses. Richardson *et al*. (2012) estimated a cost of illness of USD$9.50 per person per day from exposure to wildfire smoke, but they find that this measure underestimate the true economic costs of health effects from smoke exposure. Cardoso de

Mendoça *et al*. (2004, 2006) also identified significant costs from respiratory ailments provoked by wildfire smoke in the Amazon region.

The economic impacts of wildfires on other forest values can also be significant, such as damages to forest watershed values (Lynch, 2004), lost soil productivity, and increased vulnerability to invasive weeds (Calkin *et al*. 2008).

Of the studies focusing on estimating the impacts of wildfires, only a small number have included wildfire impacts on market goods and services. For instance, Butry *et al*. (2001) modelled and analysed the economic impact of six large wildfires in the St. John's River Water Management District, Florida, US. They included several categories of market goods and services, such as disaster relief expenditures, timber losses, property damage and tourism-related losses. Their results indicate that the largest amount of losses from the six fires analysed was a result of losses in the timber industry (between US$322 and US$509 million in 1998 dollars). Other studies have also estimated the losses caused by wildfires, but this has not been the focus of their analysis. Instead, this estimation was done as part of an integrated assessment of fire management options. These studies are reviewed in the following section.

2.5. Integrated economic assessments of fire management

There is a portion of the fire economics literature that looks at the costs of fire management activities and compares them with the benefits that can potentially be generated from those activities. We call these studies integrated economic assessments. These are studies that, one way or another, relate the costs of fire management to its associated benefits. Although the articles reviewed in this section might only analyse one of the activities carried out by fire agencies; that is, only prescribed burning, or only education, or only suppression, they distinguish themselves from the rest of the literature in that they include both the costs *and* the benefits of the management activity studied, as opposed to the literature presented in the previous sections, which looked at either the costs, the benefits, *or* the damages separately. The integrated studies reviewed below have all attempted to link the benefits to the money invested in different fire management activities and estimate the gains (or losses) of changing the level of investment.

Most of these integrated studies have combined simulations with optimisation techniques (Mercer *et al*. 2007; Butry *et al*. 2010; Prestemon *et al*. 2010; Prestemon *et al*. 2012), cost-benefit analyses (Jones *et al*. 1999; Dyer and Stafford Smith 2003; Drucket *et al*. 2008; Schaaft *et al*. 2008) or cost-effectiveness analyses (Omi *et al.* 1999; Liu *et al.* 2010; Stockmann *et al.* 2010a). Other methods that have been used for integrated studies include propensity scores (Butry 2009) and mixed integer programming (Mercer *et al*. 2008). The vast majority of these analyses have been conducted in the US, with a few exceptions for prescribed burning in private land in northern Australia (Dyer and Stafford Smith 2003; Drucket *et al*. 2008) and prescribed burning in public land in China (Liu *et al*. 2010).

2.5.1. The costs and benefits of prescribed burning

A large proportion of the integrated economic assessments have focused on evaluating the costs and benefits of prescribed burning, occasionally including other fuel treatment techniques. Some studies developed models to determine the optimal rate of prescribed burning (Prestemon *et al*. 2001; Mercer *et al*. 2007) or the optimal timing for applying the treatment (Jones *et al*. 1999), while others attempted to identify the most cost-effective strategy to reduce wildfire risk (Omi 1999; Lui *et al*. 2010; Stockmann 2007). Most analyses have focused on prescribed burning in public land and only a very small number of economic analyses have looked at prescribed burning in private land.

Prescribed burning in public land

Investigating the optimal rate of prescribed burning, Prestemon *et al*. (2001) obtained a relatively low rate for the Volusia county in Florida. Using simulation and optimisation techniques in their economic analysis, they found that the amount of prescribed burning that maximises public welfare is about 3 percent of the total forest area. Their model however, only estimated welfare impacts based on timber markets. In a similar study of the same area but using a different measure of wildfire damages, Mercer *et al*. (2007) found a much higher optimal rate of prescribed burning. They estimated a wildfire risk model and used it in Monte Carlo simulations to evaluate the welfare changes of alternative prescribed burning strategies over a 100-year future. Using intensity-weighted area burned as the measure of wildfire damages, instead of just area burned by wildfires (which is the measure used by Prestemon *et al*. 2001), Mercer *et al*. (2007) found that the optimal rate of prescribed burning in the Volusia county is 13 percent of the total forest area. In addition, the welfare impacts in their model included not only impacts on timber markets, but also on housing and the tourism sector revenues. In this study, the authors demonstrated that the results depend strongly on the measure of wildfire damages used: area burned or intensity-weighted area burned. The area-burned measure indicated that net economic losses from wildfire are minimised when 4.8 percent of the forest is prescribed burned, compared to 13 percent for the intensity-weighted area measure. The authors conclude that since net economic damages from wildfires depend on both the size and the intensity of the fires, it is crucial to take intensity levels into account, because relying only on the area burned measure may lead to the application of less prescribed burning in the landscape than it is optimal.

Besides the optimal rate, some authors have investigated the optimal timing for prescribed burning treatments. For instance, Jones *et al*. (1999) used a stochastic simulation modelling system to assess wildfire risk over five decades under different scenarios of prescribed burning in the Stevensville West Central area of the Bitterroot National Forest in Montana, US. They compared wildfire risk without fuel treatments (but with fire suppression) and wildfire risk with fuel treatments and evaluated different scenarios of prescribed burning, including scenarios that changed the timing of the treatments. The only scenarios that generated positive net present values were those for which the fuel treatments were arranged to minimise wildfire risk in the third decade, because it provided the opportunity to implement fuel treatments in the form of commercial timber harvests in earlier decades, which generated revenue that could offset the costs of fuel treatments. But again, like Prestemon *et al*. (2001), this model only estimates the impacts of wildfires on timber production.

Instead of explicitly estimating the benefits of prescribed burning or other fuel treatments, some studies have looked into the cost-effectiveness of these treatments to reduce wildfire occurrence or area burned. For example, Lui *et al*. (2010) showed that to reduce wildfire occurrence and area burned, it is more cost-effective to use mechanical treatments in combination with prescribed fire rather than using prescribed burning alone. Because the costs of these options are different, their cost-effectiveness depends on the area treated. For larger treatment areas, prescribed burning is more cost-effective, but for smaller treatment areas, mechanical treatments and prescribed fire combined are more cost-effective.

Focusing on the WUI problem, Stockmann (2007) and Stockmann *et al.* (2010a) used cost-effectiveness to evaluate the change in wildfire risk to houses in fire-prone areas from different strategies. The strategies compared in this study differed in their location (close or far from houses) and in their nature (reducing fuels or improving the houses). Their results show that to reduce wildfire risk to WUI residences, it is more cost-effective to invest in strategies that modify the level of fuels close to houses or strategies that improve building design and construction features than to invest in silvicultural forest treatments in the surrounding forests.

Another example of a cost-effectiveness study is Omi *et al*. (1999). In this study, Omi and colleagues used a fire simulator to evaluate fire size with and without fuel treatments. They included suppression costs in order to compare the costs of prescribed burning plus suppression when the treatment was applied with the costs of suppression when there was no treatment. Their result is a frontier that illustrates the relationship between treatment costs vs. reduction in area burned and can be used to select the most effective fuel treatment strategy that yields the highest reduction in area burned subject to the available budget.

Omi *et al*. (1999) estimated a second frontier that links treatment cost vs. suppression cost savings. The estimated suppression costs savings generated by the application of prescribed burning ranged between US$8,000 and $62,000 depending on the segments selected for treatment. Their results show that the cost savings per hectare are higher in some locations and lower in others, which could also provide very useful information to fire managers for the prioritisation of the treatments and allow them to identify the areas where they can obtain the highest returns on investment. However, the authors do not provide an explanation for the differences in cost savings per hectare and do not investigate the subject any further.

The issue with the above three studies is that they do not estimate the damages avoided, so they were unable to draw conclusions about whether any strategy generated benefits in excess of its costs. They only allowed comparisons of different strategies to achieve similar outcomes.

Apart from avoided timber losses and reduced damages to housing and tourism, prescribed burning has been shown to generate other benefits, such as reducing sediment runoff. Fire-prone ecosystems are susceptible to post-fire erosion that can result in increased sediment loads in water catchments (Lane *et al.* 2006; Feikema *et al.* 2011). These sediments have to be cleaned out by water management authorities, which can be a very expensive activity. Hence, there can be economic benefits in the form of reduced sediment clean up from changing the fire regime through prescribed burning. Loomis *et al*. (2003) investigated this topic and found that changing the fire regime from low-frequency, high-intensity wildfires, to a shorter interval of prescribed burns in the Angeles National Forest would reduce sediment yield and result in

USD $24 million cost savings for Los Angeles County Public Works because of reduced debris clean out.

Prescribed burning can also generate economic benefits in the form of greenhouse gas offsets. Heckbert *et al*. (2012) examined the potential for fire management to generate offsets to carbon markets in northern Australia. They found that prescribed burning can contribute to greenhouse gas abatement targets and yield net benefits even when the treatment is applied over large areas.

Prescribed burning in private land

The studies examining the costs and benefits of prescribed burning which have been reviewed up to this point, have all looked at prescribed burning on public land. There has also been some work on the costs and benefits of prescribed burning in private land holdings, although it is much scarcer. For instance, Yoder and colleagues (Yoder *et al.* 2003a; Yoder *et al.* 2003b; Yoder 2004; Yoder and Blatner 2004) developed a dynamic economic model of prescribed fire use in private land that reproduces the case of forested land in the Rocky Mountains, US. The model estimates the optimal level of precaution that an owner needs to exert in order to minimise the likelihood of damage to neighbouring properties and the optimal length of time between prescribed burns that maximises the expected net present value for private landowners. He compared different scenarios changing the level of potential damages, the level of wildfire risk and the total liability faced by the prescribed burners. His results suggest that in areas of high potential damages, such as the WUI, or in areas of high wildfire risk, it may be beneficial to engage in relatively frequent prescribed burns.

Other work on the economics of prescribed burning in private land has been done in the northern parts of Australia. One example is Dyer and Stafford Smith (2003), who developed a simulation model to evaluate the economic impacts of different prescribed burning treatments on pastoral lands in the Northern Territory, Australia, which are predominantly grazed by cattle. In this area, pastoral managers face difficult choices about trade-offs between short-term maximisation of pasture productivity for grazing and vegetation management for the long-term balance of grass and tree cover. They found that prescribed burning results in significant long-term net benefits that stem from reduced tree cover and improved pasture condition and production, which ultimately leads to increased animal performance and enterprise profitability. They also found that applying treatments in the late dry season and allowing some wildfires to burn, instead of applying a fire exclusion strategy, is a more effective option for controlling tree populations, but pastoral managers have little incentive to integrate fire in their grazing management strategy because of the absence of significant economic benefits in the short-term from using fire.

In another study of pastoral properties, but this time in the Cape York Peninsula in northern Queensland, Australia, Drucker *et al*. (2008) showed that fire and fire management have a significant effect on the profitability of cattle properties. Unlike Dyer and Stafford Smith (2003), who found the largest returns from late dry season treatments, Drucker and colleagues' model generated the largest returns from early dry season burning. The differences in their results are partly explained by the differences in the frequency of wildfires and the fire-fighting efficiency between the two locations, which result in significantly different wildfire impacts on pastoral properties in each region. This shows the considerable effects that local environmental and management conditions can have on optimal management strategies and highlights the importance of tailoring these analyses to the specific local conditions.

2.5.2. The costs and benefits of other pre-suppression activities

The costs and benefits of other pre-suppression activities, such as mechanical fuel reductions, education campaigns and detection systems, have also received some attention in the literature, but considerably less than prescribed burning.

Mechanical fuel reductions

For mechanical fuel reductions, Prestemon *et al*. (2012) quantified the net economic benefits of mechanical treatments on the western US timberlands. They showed that in forests that produce timber, the benefits that mechanical treatments can generate depend mostly on the type of products that can be sold (i.e. if the products obtained from mechanical treatments can be sold or not and if large trees can be harvested). Interestingly, they found that when wildfire costs and benefits are considered, the results do not change significantly.

Education

Although education campaigns are regularly conducted by fire agencies, wildfire education has received very little attention from the fire economics literature. On this topic, we only know of three studies that have investigated its costs and benefits. The first one is Prestemon *et al*. (2010), who analysed the effect of education campaigns on the number of wildfires ignited in Florida, US, and estimated the subsequent benefits that could potentially be generated from a change in wildfire education efforts. They estimated the benefits of wildfire prevention education by calculating the damages caused by the current numbers of fires per year and estimating the amount of damages that would have been averted if there were fewer fires as a result of increased education. Their results indicate that with current levels of education effort, marginal benefits are significantly greater than marginal costs, hence government decisions to increase spending on wildfire prevention education could yield immediate positive net benefits, even with modest alterations to spending.

The second study is Butry *et al*. (2010), who simultaneously analysed education and prescribed burning. Butry and colleagues investigated how prescribed burning and wildfire prevention education can be jointly organised in Florida, US, to minimise the sum of management costs and expected wildfire damages. They found that to minimise expected costs-plus-losses, both the investment in prescribed burning and in education need to be increased substantially. Expected costs-plus-losses were minimised with an increase of 74 percent in the number of hectares prescribed burned and a 164 percent increase in wildfire prevention education spending state-wide.

Thirdly, Gibson and Pannell (2014) evaluated education of town dwellers and training for farmers in best practice burning methods as two of 10 potential strategies in the Otago region of New Zealand. They found that community education in the town was highly attractive, with a benefit:cost ratio of 12:1. Although not included in the report, their model shows that the training program for farmers had a Benefit: Cost Ratio ranging from 0.4 to 9, depending on the location of farmers relative to the high-value assets in the region (David Pannell, personal communication).

Wildfire detection

On the topic of detection, we only know of one study that has assessed the economic benefits of detection. Steele and Stier (1998) evaluated the costs and benefits of fixed lookouts for wildfire detection compared to public detection alone in Winsconsin, US. They estimated that each dollar invested in lookout detection generates six dollars of benefits in the form of reduced suppression costs and reduced property damages, but the economic effectiveness of lookouts varied greatly with location. However, their study does not take into account the long-term consequences of earlier detection and the subsequent potential increase in suppression success, which could lead to a more rapid accumulation of fuels in the landscape and ultimately result in higher intensities for wildfires and increased suppression costs.

2.5.3. The costs and benefits of wildfire suppression

A few integrated assessments have focused only on suppression activities, either determining the optimal investment in suppression (Mees *et al*. 1994; Petrovic and Carlson 2012), or the optimal mix of suppression resources (Donovan and Rideout 2003a), or assessing whether it is worth investing in costly technologies for suppression (Hesseln *et al*. 2010). In these studies, prescribed burning and other fire mitigation activities are excluded from the analysis.

The optimal level of investment in suppression is the level that minimises suppression costs plus damages. We only know of two studies that have investigated the optimal suppression investment. The first one is Petrovic and Carlson (2012). In this study, the authors used simulation to analyse trade-offs between wildfire suppression and damages and developed a modelling framework to determine the optimal suppression response depending on suppression costs, the number of fires attended simultaneously, and the potential damages. Their results indicate that suppression investment should increase as the density of assets at risk increases and the costs per unit of suppression resources decrease. The second study is Mees *et al*. (1994). They developed a model to estimate the optimal expected costs of a wildfire (suppression plus damages) that accounts for uncertainties in fire behaviour and uncertainties in the efficiency of the suppression resources. They show how the optimal strategy that minimises potential suppression costs plus damages varies with uncertainty in the predicted flame-length and uncertainty in the width of the fire-line that can be constructed by the fire-fighting resources deployed. The optimal solution appears to be more sensitive to uncertainty in the fire-line width, which can explain why fire managers often deploy a large amount of resources in order to increase the certainty of success.

Other integrated assessments of suppression have focused on addressing resource allocation for wildfire containment, investigating how to distribute suppression resources in order to achieve the most efficient outcome. One example is Donovan and Rideout (2003a). They developed an integer programming model that determined the optimal mix of fire-fighting resources under different types of budget constraints and conducted a sensitivity analysis to see how the costs of fire-fighting resources, the total damages and the sum of costs and damages would change with different suppression strategies. The results of their sensitivity analysis indicate that when fire-fighting forces take a long time to arrive to the fire, although it does not increase suppression costs, it significantly increases total resulting damages. More importantly, putting a constraint on fire-fighting resources has a small impact on the sum of costs plus losses, but the reduction in suppression costs is achieved at the expense of large increases in total damages.

Another area of study researched is the use of geospatial technologies, which are often costly. Examples of these technologies are time-dated briefing maps, incident action-plan maps, fire progression maps, and area and transportation maps. With these technologies increasingly being used in fire suppression, it is important to ascertain whether their high costs are outweighed by the benefits they generate. Hesseln *et al*. (2010) is the only study that has researched this area. They evaluated the costs and benefits of geospatial technologies used by the USDA Forest Service to fight large fires in the Northern Rocky Mountains. They found that, despite their costs, geospatial technologies do not seem to significantly increase suppression expenditures, possibly because they allow fire managers to allocate fire-fighting resources more efficiently. Thus the costs of geospatial technologies may be offset by improvements in the use of suppression resources.

More recently, research has also investigated the benefits of using daily information on the spatial variation of wildfire-potential to mobilise suppression resources (Christman and Rollins 2015). This research has found that there can be significant savings in suppression costs when wildfire-potential information is used and suppression resources are moved throughout the fire season accordingly, instead of staying in their home locations all season.

2.5.4. Economic trade-offs in fire management

We know that if there was no fire management at all, the damages would be significantly greater. Compared to a no-intervention scenario, investing in fire management yields net benefits in most situations (Bennetton *et al*. 1998). However, a scenario with no intervention at all (i.e. no suppression, no fuel managements and no education campaigns) would probably never occur, and in today's context of climate change and the expanding WUI, examining whether it is beneficial to have a fire management program (compared to not having one) becomes irrelevant. What government and fire agencies need to know, is whether the current levels of investment in fire management maximise welfare to society, or whether a different level of investment would be more advantageous, and how to use their budgets to get the highest return on investment.

For more than a century now, economists have analysed these complex issues. They have attempted to answer two important economic questions: (1) what is the optimal level of investment in fire management, and (2) what is the best way of distributing it between all the different fire management activities. To answer these questions, it is necessary to identify the trade-offs between different management activities and wildfire damages. In other words, to assess what we gain from increasing expenditure in one activity and what we lose from reducing it somewhere else.

Integrated economic assessments that have explicitly looked at trade-offs between different management activities and have put some effort into quantifying them are very rare. Only a handful of studies have attempted to do so.

Some analyses have examined the trade-offs between fuel treatments and suppression. For instance, Schaaft *et al*. (2008) used a simulation model to assess trade-offs between fuel treatments and suppression in the Angeles National Forest, US. They combined and evaluated five prescribed-burning rates and two suppression-program options, evaluating the costs of each alternative against the benefits it generated in terms of reduced area burned and avoided resource losses. They found that a strategy that combines a high suppression investment with a low rate of prescribed burning significantly reduces area burned by wildfires, but the benefits of this strategy in terms of reduced resource damages do not compensate for the increase in management costs. Likewise, a high rate of prescribed burning combined with a low investment in suppression is costly and the benefits do not make up for the increase in costs. They concluded that a strategy involving a lower investment in suppression combined with a moderate rate of prescribed burning that targets areas with high fuel loads yielded the highest net benefits.

Butry (2009) estimated the returns of both fire suppression and fuels management using propensity score matching techniques. His study shows that there are substantial financial benefits of fire management, but the estimated returns have high margins of error. The returns on investment for prescribed burning in terms of damages avoided may be estimated using propensity scores, but Butry indicates that more research is required to do the same for suppression. He estimated a benefit-cost ratio of damages avoided per dollar spent in prescribed burning of 1.53, but indicates that this estimate is likely to vary geographically. He emphasises the fact that prescribed burning appears to reduce wildfire intensity, but may be less effective in reducing total wildfire area burned.

Gibson and Pannell (2014) compared the benefits and costs of various fire prevention strategies in two cases studies: Otago, New Zealand, and Adelaide, South Australia. In the New Zealand case study the strategies evaluated included community education, fuel removal within a town, regulation, on-ground support to farmers conducting burns for land management, training of farmers, fire breaks, and incentive payments to farmers. Their analysis measured benefits resulting from reductions in fire incidents, and reductions in the spread of fires. They found a wide range of Benefit: Cost Ratios, from highly favourable to highly unfavourable. The most favourable strategies tended to involve taking actions close to the high-value asset (the town and a commercial forest) rather than on farms some distance away.

The Gibson and Pannell (2014) case study in South Australia also measured benefits resulting from reductions in the cost of fire suppression as a result of prescribed burning. They examined prescribed burning strategies of different intensities. Again they found that the Benefit: Cost Ratio of an intervention tended to be higher if it was physically closer to high-value assets. Prescribed burning generated benefits in excess of costs in some cases, but not in others (where the burning was remote from assets). This suggests that a targeted approach to prescribed burning may be warranted.

Other studies have compared investment in suppression alone vs. investing in both suppression and prescribed burning. Snider *et al*. (2006) compared the costs of a fire management program that would involve only suppression activities over 40 years with the costs of a program that would combine suppression and fuel treatments. His analysis shows that investing in mechanical and fire treatments for restoration-based wildfire hazard reduction results in benefits in the form of reduced future costs of suppression. This justifies an investment of $238 to $601 per acre in hazard reduction treatments.

One study assessed the benefits of prescribed burning for initial attack. Mercer *et al*. (2008) showed that there are gains from fuel treatments for suppression because they reduced the chances of a starting fire escaping initial attack (and hence requiring extended attack), especially when the resources available for initial attack are limited.

Finally, some authors have suggested the use of wildfire for vegetation management (Dale 2006; North *et al*. 2012) which may result in ecological benefits such as improved watershed and forest conditions and improved wildlife habitat, and economic benefits in the form of reduced suppression and fuel treatment costs in the long-run. However, there are considerable barriers to the use of wildfire as a land management tool, particularly regarding institutional and public support (Dale 2006) and cultural and regulatory barriers (North *et al*. 2012).

2.6. Incentive structures

A small part of the fire economics literature has investigated people's incentives to do certain things relating to fire management. These studies have evaluated incentives either from a landowner perspective or from a fire-manager perspective. From a landowner perspective, research has focused on analysing landowners' incentives to engage in wildfire risk reduction activities such as prescribed burning inside their properties, or investigating the effectiveness of liability law regulating prescribed fire in the US. From a fire-manager perspective, studies have mostly concentrated on evaluating fire managers' incentives to contain management costs, particularly for suppression.

The analysis of landowners' incentives to prescribe burn revealed that several factors affect landowners decisions, such as initial fuel loads, public provision of wildfire suppression, insurance cover, whether their land is located close to high-value assets (i.e. in the WUI), fear, perceptions on responsibility, and the perceived effectiveness of mitigation (Yoder and Blatner 2004; McFarlane *et al*. 2011; Busby *et al*. 2013) Studies examining liability laws relating to prescribed burning in private land have found that the effectiveness of these laws depends on the capacity of landowners and their neighbours to limit the probability and extent of damage. If the neighbours of the burner have a low capacity to reduce expected damages, then strict liability laws that make the burner entirely responsible for the damages of an escaped fire are more effective and lead to efficient precaution. But if both burner and neighbours can reduce expected damage, then strict liability law is less effective and other laws based on the carefulness adopted by the landholder whose land is burned are more effective in producing efficient precaution (Yoder *et al.* 2003a; Yoder *et al.* 2003b). The structure of liability laws has also been found to affect the timing of landowners' burns (Yoder 2004).

With respect to risk reduction in public land, studies have found that landowners have a tendency to "free ride" on public protection and invest too little in fuel treatments (Busby and Albers 2010), even when information about wildfires improve (Busby *et al*. 2013). Some authors have suggested that to reduce the social costs of landowners sub-optimal decisions, they must be required to share the government's costs of suppression (Crowley *et al*. 2009).

Fire managers' risk attitudes generally tend to result in the over-allocation of fire-fighting resources, even when the probability or extent of damages from fires are low (Wibbenmeyer *et al*. 2013). In addition, the incentive structures in which fire managers operate has generally been thought to encourage aggressive suppression and the overutilisation of fire-fighting resources (Donovan and Brown 2005; Thompson *et al.* 2013a). In order to halt this tendency and introduce incentives for fire managers to contain management costs some authors have suggested that suppression budgets be fixed and that fire managers be allowed to carry over surpluses and deficits from year to year (Donovan and Brown 2005). Other authors have suggested the use of a new performance measures, such as risk-based metrics for annual suppression costs based on statistical expectations for fire season expenditures (Thompson *et al.* 2013a; Thompson *et al.* 2015). This would involve looking at the probability distributions for suppression costs, which is more congruent with a risk-based management approach.

2.7. Conclusion

As this review shows, existing economic theoretical models relating to wildfire management need to be extended in order to develop cost-effective and sustainable wildfire management strategies that effectively balance short- and long-term fire risks to ecosystems, life and property. The economic theory of wildfire management has been the subject of study since the "Least Cost Plus Loss" model was first expressed by Sparhawk (1925), but the theory still needs to better reflect the dynamics of wildfires.

In the empirical literature on fire economics, the costs of pre-suppression and suppression activities and the factors that influence these costs have been relatively well covered. Most authors seem to agree that focusing on suppression as the principal wildfire mitigation strategy and continuing increasing the investment in suppression resources and activities to manage the wildfire problem is not likely to yield the greatest net benefits in the long-run.

There is also an adequate amount of research estimating the indirect and non-market benefits of wildfire mitigation programs, particularly analysis dealing with WTP estimates for reductions in wildfire risk and area burned to inhabitants of fire-prone areas. There are some integrated economic studies of fire management, although less abundant than other parts of the literature. Despite the abundance of studies estimating people's WTP for fire prevention programs, indicating the existence of substantial potential non-market losses and society's growing concerns about biodiversity and species protection, these estimates have not been included in integrated analyses. Some integrated studies recognise the importance of non-market values, but they do not incorporate estimates of non-market losses caused by wildfires or the non-market benefits of implementing fire protection programs. Moreover, there is no analysis of trade-offs between the values at risk that fire management programs aim to protect. Although some integrated studies attempt to elucidate the trade-offs between the different fire-management activities, the trade-offs have not been clearly quantified between pre-suppression activities, suppression activities and damages. Considerable additional research is required in order to elucidate these trade-offs and there is also a need for more empirical economic analyses of wildfire management that account for the long-term effects of wildfires and fuel management strategies (Mercer *et al*. 2008).

Chapter 3.

WILDFIRE MANAGEMENT IN THE SOUTH-WEST FOREST REGIONS OF WESTERN AUSTRALIA

The case study area selected for this study is located in the south-west of Western Australia (WA), which is one of the most fire-prone regions in the world (McCaw and Hanstrum 2003; Hamilton *et al.* 2009). The forests, woodlands and heathlands of the south-west of WA are subject to a Mediterranean type climate with very hot summers and long periods of drought. As a result, the vegetation in the region produces large quantities of dry fuels that cumulate over the years and can easily be ignited during the drier months of the year, from November to February (McCaw and Hanstrum 2003; Burrows and McCaw 2013). Lightning strikes, accidental ignitions and arson provide the spark that results in recurrent wildfires (McCaw and Hanstrum 2003; Hamilton *et al.* 2009). For millions of years, wildfires have constituted a natural process and a vital element for the region's biota that today cannot be totally prevented or eliminated from the landscape (Pyne *et al.* 1996; Pyne 1998, 2006; Burrows and McCaw 2013). Therefore, in order to protect the region's ecosystems and human communities, wildfires need to be managed and become an integral part of land and ecosystem management (Burrows 2008; Bradstock *et al.* 2012b; Burrows and McCaw 2013). However, the management of wildfires can only happen through those few elements over which we can have some control; that is, fuel levels, suppression effort and, to a certain extent, the quantity and type of human-caused ignitions (Pyne *et al.* 1996).

Throughout history, these three aspects of fire (fuels, suppression and ignitions) have been managed in varying ways in the region. The reduction of fuels through prescribed burning, which is the focus of this study, has been promoted at times as a means to re-integrate fire in the landscape, manage fuel accumulation and restore ecosystems. But the application of prescribed burning raises the question of the fire regime (Gill 2005): what is the most appropriate fire regime for the region? That is, what frequency, intensity and size of treatments are the most suitable for its forests? Is the optimal fire regime a *natural* one, like the regime that was in place before humans arrived to Australia, or is it better to reproduce the practices adopted by the Aborigines that lived in the region before European settlement?

Or an entirely different one? And who would benefit most from a given fire regime: the region's ecosystems or the human communities living in it?

The history of wildfires in the region and the different management techniques used by both Aboriginal and post-colonial communities may provide some insights about these questions, improve our understanding of the challenges associated with fire management today, and provide information that could be used to improve current and future fire management policies. By taking an evolutionary and historical approach, we can investigate how the biodiversity in the region has lived with fire, how it has responded to different fire regimes, and examine the varied consequences of human use of fire. As Hopper (2003) put it, we "might derive [from evolutionary science and environmental history] principles and hypotheses worthy of consideration as we attempt to achieve life and property protection, and biodiversity conservation through fire management across south-west WA ecosystems" (p.10). For this purpose, this Chapter provides a historical review of fire and its management in the south-west of WA and an overview of current practices and challenges.

The Chapter is organised as follows. Section 3.1 reviews the evolution of fire regimes before humans, after the first humans arrived to Australia and after European settlement, and how these changes in fire regimes affected the region's ecosystems and human communities. Section 3.2 looks at the current context where there is an effort to reintroduce fire in the landscape and the challenges associated with the use of prescribed burning in the region today. Finally, Section 3.3 provides a detailed presentation of the case study area used for this study.

3.1. The history of fire and its management in the south-west of WA

3.1.1. Fire before humans

Fire has been present in Australia for millions of years and it has been a key element in the evolution of Australian ecosystems (Bowman 2003; Hopper 2003). How fire became such a central element of the landscape is still a question (Bowman 2003) but there is no doubt that fire was already present in Australia before humans arrived in the continent (Pyne 1998, 2003, 2006). There is no evidence to support the theory that fire became important in the Australian continent only after the arrival of humans (Kershaw *et al.* 2002; Bowman 2003; Hassell and

Dodson 2003; Hopper 2003). However, the frequency, the intensity and the spatial distribution of pre-human or “natural” fire was very different from the human fire regime.

The fire regime before humans arrived to Australia was characterised by lightning ignitions, a patchy frequency and fires that burned very large areas (Bowman 2003). Natural fire was patchy partly because the only source of ignition was lightning, which is inherently random. Many areas with accrued flammable fuels could remain unburned for long periods of time if they were not hit by dry-storm lightning, and conversely, lightning could strike areas where fuel loads were low and discontinuous and the fires would die without spreading (Pyne 2003). The size of these fires was much larger than contemporary fires because there were fewer elements to stop them: no humans with organised suppression resources and technology, no major topographic barriers over most of the continent, and more continuity of the fuels in the landscape (i.e. no patches of cleared vegetation). As a result, only a change in the prevailing weather conditions could limit the size of a fire (Bowman 2003). In the south-west of WA, natural fire burned the land at varying intervals, between 25 and 150 years on average for mallee-heath (Hopper 2003). When humans arrived in the continent however, their presence significantly altered the fire regimes of the landscapes they inhabited (Bowman 2003; Kershaw *et al*. 2002).

3.1.2. Humans and fire before European settlement in the south-west WA

Humans are believed to have caused a significant increase in fire activity. Most researchers consider this change from pre-human to human fire to be of fundamental importance in shaping the fire ecology of Australian ecosystems (Bowman 2003; Kershaw *et al*. 2002). With the arrival of humans in Australia, there was a new source of ignition that changed the patterns of burning that had existed until then (Bowman 2003; Kershaw *et al*. 2002). Although the exact time of arrival of the first humans to the Australian continent is still a subject of debate, it is generally agreed that it occurred between 70,000 and 40,000 years before present. There is evidence that around this time, there is a noticeable increase in fire activity, and in the absence of major climate change, this increase is most likely due to burning by Aboriginal communities (Kershaw *et al*. 2002), for whom fire was an essential tool in their way of life.

The first humans that inhabited the Australian continent used fire frequently to support their needs. Practically all aspects of Aboriginal life involved the use of fire (DEC 2008). Aboriginal communities used fire as a tool for hunting, managing the vegetation, clearing tracks and campsites, providing light and warmth, cooking, signalling, and for ceremonial and spiritual practices (Mills 1989; Pearce 1989; Bowman 1998, 2003). This new source of ignition significantly altered the fire regime. Lightning ignitions coupled with human use of fire resulted in more frequent fires in smaller patches.

All around Australia, there is evidence of human control of fire by Aboriginal communities (Pearce 1989; Bradstock *et al.* 2012a). However, the details about the extent, frequency, intensity and homogeneity of Aboriginal burning are still under debate (Bowman 1998; Abbott 2003; Bowman 2003). It is believed that each community used fire in a different way (Abbott 2003). The contemporary theory of burning by Aboriginals has been principally derived from looking at practices in the north of Australia, but for the Aboriginal fire regime in the south-west of WA there is less information in the literature (DEC 2008).

The indigenous people of the south-west of WA, the Noongars, applied fire to relatively small sections of the landscape at a time and at changing intervals for all areas (DEC 2008). It is uncertain how systematic or casual they were about the application of fire (Abbott 2003). The early recounts of the first Europeans indicate that the Noongars near Albany burned the landscape in consecutive patches, whereas the Noongars near Perth were not particular about where they lit fires (Abbott 2003). There is more knowledge about the season of burning, since early records of observations of fires were dated (Abbott 2003). Their burning occurred mainly during the dry summer months between December and March, which coincided with the peak period of lightning ignition (Burrows and McCaw 2013). On suitable days, there could be many ignitions and the resulting fires could burn large patches in the order of tens and hundreds of hectares (Abbott 2003). This maintained a mosaic of fuel ages across the landscape, from recently burned to long unburned, that limited the impact of large, intense wildfires that may have threatened the life and resources of the Noongar communities. The first Europeans described the Northern Jarrah forest (located south of Perth) as a park-like environment in which "a horse and cart could move between the giant [eucalyptus] trees" (Mills 1989, p.234). However, this changed dramatically when Europeans settled in south-west WA.

3.1.3. Humans and fire after European settlement in the south-west WA

There is unmistakable evidence that the fire regime resulting from Aboriginal burning was very different from the regime that was imposed by European settlers. Most of the settlers that founded the first colony around the Swan River area (where Perth is today) in 1829 came from Britain, a place where fire was not an important part of land management at the time (Mills 1989; Abbott and Burrows 2003). Most of them were upset about the extent of Aboriginal burning around Perth and Albany (Mills 1989). They mostly saw fire as a threat to their newly acquired assets. Progressively, with the growth of the colonies, the new perceptions about fire and the land use changes produced by the early settlers, the landscape and the fire regime were drastically changed in the south-west forests of WA.

Early settlers had plans for the regions that were to change the face of the landscape. The region's climate and soils provided a suitable location for the development of the livestock industry and other agricultural enterprises. The new colonies were to supply the Royal Navy with beef and pork and the East India Company with horses (Mills 1989). As a result, large parts of the forests disappeared in a relatively short period of time. Parts of the forests were used for timber production, other parts were cleared for agriculture. These land-use changes had profound implications for the fire regime of the region. Most of the new land uses were at risk of being destroyed by fire (Burrows and McCaw 2013). The timber industry wanted to protect the production of timber, the livestock industry wanted to reduce the land covered by forests in order to have more land for grazing and protect their fences, and other farmers wanted to protect their agricultural land (Mills 1989). More and more settlements were developed in the area, which led to more human assets being at risk. These new perspectives resulted in a new fire policy where fire was to be avoided at all costs in order to protect human assets in the landscape. Not only would all fires be stopped as soon as possible, but the purposeful application of fire in the landscape like the Noongars did was also avoided (Mills 1989). This no-fire culture was passed on to the following generations of settlers, who also aimed to eliminate fire from the landscape without taking into account the ecology of the local ecosystems.

In order to avoid fires as much as possible, the colonies developed systems that would help them achieve this task. They developed the first fire-fighting forces, detection system and other practices that would increase their chances of rapidly detecting the fires so that they could promptly react to stop them from spreading (Havel 1989). More trucks were acquired and increasingly more people and resources were used to detect and fight the fires. Fire research also began around this time (1933) in the town of Dwellingup (south-east of Perth). The first fire hazard maps were created and forester Roy Wallace developed a simple method to predict daily fire risk in order to have fire-fighting resources ready and available when most needed (Abbott and Burrows 2003).

This emphasis on fire exclusion had long-term impacts on fire behaviour and ecological consequences that later created a new set of problems. In places where fire used to occur relatively frequently, the reduction in fire occurrence led to a substantial accumulation of fuels. Where before the forest had a park-like aspect due to Aboriginal burning, there was now re-growth of the undergrowth. The increase in the interval between fires changed the region's ecosystem structure and composition (Hopper 2003). The rapid accumulation of fuels in the landscape produced more severe wildfires, both in intensity and extent. As a consequence, wildfires became more difficult to control (Havel 1989).

With an increase in the extent and intensity of wildfires, the fire exclusion policy started to be questioned (Burrows and McCaw 2013). Large and damaging wildfires in 1949 and 1950 made foresters realise that fire could not be completely avoided and that a fire exclusion policy was impractical and unsustainable in the long run (Burrows and McCaw 2013). In 1954, the Forests Department (the early predecessor of today's WA Department of Parks and Wildlife) recognised prescribed burning as a fundamental component of fire management and began to re-introduce fire in the landscape.[8]

3.1.4. Reintroducing fire through prescribed burning

The second half of the 20th century saw the progressive reintroduction of fire in the landscape through the application of prescribed burning (Hamilton *et al*. 2009). Although the importance of broad-acre prescribed burning was recognised in the 1950s, its implementation was initially limited due to insufficient resources and little understanding of weather and fire behaviour (Burrows and McCaw 2013). There was a change in attitudes but not enough investment in the practice. A series of catastrophic fires in 1961 that destroyed the town of Dwellingup provided the impetus for an increase in the extent of the practice. These fires instigated a Royal Commission that recommended the improvement and the expansion of the prescribed burning

[8] The Forests Department was created in 1918 and existed through to 1985 when the Department of Conservation and Land Management (CALM) was created. CALM was replaced by the Department of Environment and Conservation (DEC) from 2006 to 2013, and since 1 July 2013 the Department of Parks and Wildlife is responsible for prescribed burning in the public lands it manages in the south-west of WA.

program (Hamilton *et al*. 2009).[9] The few areas that had been prescribed burned before the fires of 1961 provided evidence of the capacity of the treatment to reduce fire intensity and damage to the forests (Burrows and McCaw 2013). New understandings gained from fire ecology research in the 1960s also led to the realisation that fire had an important role in fire-prone ecosystems (Burrows and McCaw 2013). Since 1961, the Forests Department and its successors have fully integrated prescribed burning into their fire management programs, although the extent of the practice has varied greatly over the decades (Hamilton *et al*. 2009; Burrows and McCaw 2013).

3.1.5. Concluding remarks on the history of fire and its management in the south-west of WA

The historical outlook presented in this section brought to light four key points that have important implications for present-day management of wildfires in the south-west of WA:

1. Fire has been part of the landscape for millions of years and has always played an important role in the evolution and composition of the region's ecosystems. The most important implication of this is that fire cannot be avoided or eliminated from this landscape, and thus needs to be managed as part of land and ecosystems management.
2. Since the first humans arrived in the continent, their presence has provided a new source of ignition that has effectively changed the fire regime in the region. This needs to be taken into account in fire management, because it implies that introducing a "natural" fire regime is no longer possible due to the presence of humans. In addition, the fire regime produced by natural fires of less frequent but very large and intense wildfires would probably not be tolerated today by the communities living in the area.
3. A fire exclusion policy results in the substantial accumulation of fuels in the landscape and a fire regime of extensive and intense wildfires. This suggests that excluding fire from the landscape is not an effective way of protecting the human and environmental assets in the region.
4. Since fire cannot be avoided and is important for the region's biota, it needs to be managed. One way of managing it is by purposefully applying fire to reduce fuel loads.

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[9] Royal Commissions are a type of public inquiry held in Australia where the executive government appoints temporary ad-hoc bodies to investigate–and usually to provide advice–on a defined issue. Royal Commissions are different from other types of public inquiries in that legislation under which they are established gives specific powers of investigation to them.

But depending on how this is done, it will affect the people, human assets and ecological assets of the region differently.

3.2. Fire management today

The history of fire in the south-west of WA has shown that prescribed burning has the potential to reduce the extent and intensity of wildfires. However, applying fire in the landscape to the levels that were used by Aboriginal communities prior to European settlement is unlikely to be possible in today's context. The current landscape looks entirely different from the landscape that the Noongars lived in. Before the Europeans arrived to the region, it is estimated that only 1200 to a few thousand people lived in the area, which was mostly covered by forests (Pearce 1989). Today, more than 174,000 people live in the South West (South West Development Commission n.d.), and as of 2000, about 90% of the pre-European vegetation in the Biodiversity Hotspot (which covers an area of 310,000 km^2) had been cleared (Bradshaw 2012), mostly for use in agriculture. Present-day fire management in the south-west of WA is different to what it was in the past and new challenges have emerged. This Section describes current fire management practices in the region and the major challenges that fire managers face with regards to the application of prescribed burning.

3.2.1. Prescribed burning

The purpose of the application of prescribed burning in the south-west forests of WA has evolved through time. Initially prescribed burning was principally applied to reduce fuel loads and protect the forests from high-intensity wildfires that could cause serious damage to property and the State's timber reserves. With additional information available on the effects of prescribed burning and more research on the impacts of different types of fires on the plant and animal species of the region, the treatment started to be also applied for the protection of the rich ecosystems in the area, many of which depend on fire for their survival. The government of WA now recognises prescribed burning as a valid technique for both land management and the treatment of wildfire risk (DEC 2008). The treatment has been shown to reduce wildfire extent in the south-west of WA and scientific enquiry supports the technique (Boer *et al.* 2009). Today, the application of prescribed burning has several aims for land management, including strategic fire risk reduction to protect life, property and other human assets from wildfires, biodiversity conservation and silviculture (Burrows 2008; DEC 2008;

Hamilton *et al*. 2009; Burrows and McCaw 2013). However, the appropriate fire regimes to meet these objectives continue to be debated.

Since the onset of the broad-acre prescribed burning program in the 1960s, the extent of the area prescribed-burned and the area burned by wildfires has varied greatly. Between 1951 and 1969, the amount of area prescribed burned in the south-west increased linearly. However, since then, the area treated has been declining in a linear way to the levels that were applied at the start of the program (Gill and Moore 1997; Burrows and McCaw 2013). The area burned by wildfires shows an inverse relationship with the area prescribed burned over this time that is visible at the regional scale (Boer *et al*. 2009; Burrows and McCaw 2013). Between 1962 and 1990, the proportion of the region prescribed burned per year ranged from 8% to 18%, with an annual average of 13%. Over the same period, the proportion of the region burned by wildfires varied between 0.1% and 1.1%, with an annual average of 0.3%. Between 1991 and 2012, the proportion of area prescribed burned ranged from 4% to 9%, with an annual average of 7%, while the proportion of area burned by wildfires varied between 0.2% and 4.7%, averaging 1.1% annually (Burrows and McCaw 2013). Individual wildfires of more than 20,000 hectares were relatively rare in the region in the 1970s and 1980s, but since 1997 there has been at least one of these large wildfires every second or third fire season (Burrows and McCaw 2013).

Todays' DPaW fire management target for the south-west forests of WA is to prescribe burn 10% of the land it manages in the region per year (about 200,000 hectares annually for the greater south-west), which will allow for 60% of DPaW-managed land to have fuels less than 6 years old in any one year. However, the target has been rarely achieved in the last 2 decades. Since 1996, the annual target of burning 200,000 hectares in the south-west has only been achieved four times (three times between 1996 and 2005, and only once between 2006 and 2015).

3.2.2. Wildfire suppression

Fire suppression is carried out by two agencies in the region: the Department of Parks and Wildlife (DPaW) and the Department of Fire and Emergency Services (DFES). DPaW is responsible for the suppression of wildfires that occur in the land that it manages and DFES is responsible for suppression in other lands (mostly private land). For fire-fighting resources, DPaW relies on permanent paid staff that in many cases may also have conducted prescribed burns during the cooler months of the year. DFES relies mostly on volunteers that are organised in Local Bushfire Brigades in rural and pastoral areas or are part of the Volunteer Fire and Emergency Services brigades that respond to a number of emergencies and natural disasters in the State. Each Department may also provide support in fire suppression to the other when the fire occurs near the lands they manage.

Most fires are rapidly suppressed after they are detected by either agency. By law under the WA Bush Fires Act 1954 and the Fire Brigades Act 1942, all necessary steps must be taken for preventing and extinguishing fires and protecting and saving life and property from fire. DPaW's Code of Practice for Fire Management (DEC 2008) indicates that the Department "will respond to fires occurring on or near Department-managed lands to a degree that is appropriate to the values at risk" (p. 12). This is a departure from what was indicated in the Department's Fire Management Policy in 2005, where it was specified that in circumstances where the impacts of a wildfire are likely to be low or the resources are limited, DPaW's response "may mean observation, rather than active suppression" (CALM 2005, p. 5). However this statement was removed from subsequent policy documents.

3.2.3. Other management activities

Other fire management activities currently carried out in the south-west include education campaigns, arson prevention activities and fire detection. For instance, DFES delivers community education and fire awareness programs State-wide and provides teaching resources for wildfire education in primary schools in the south-west of WA (DFES 2015). DPaW engages in wildfire surveillance and detection through the deployment of 'spotter' aircrafts during the peak fire season; during the 2013-14 fire season, 10 wildfire spotter aircrafts were deployed for a total of 4,300 hours (DPaW 2014). For arson prevention, DFES works closely with the WA Police and DPaW to reduce arson, by developing campaigns that encourage the community to report suspicious behaviour, and identifying and convicting arsonists (DFES 2015).

3.2.4. Current challenges

Prescribed burning generates much debate among scientists, ecologists, policy makers and the community, and the relative importance placed on prescribed burning by these different stakeholders varies with the time since the last large fire event (Attiwill and Adams 2013; Burrows and McCaw 2013). Recent Royal Commission and Special inquiries (Teague *et al.*

2010; Keelty 2012) have emphasised the protection of the community as the primary goal of fire management in the populated forested landscapes of southern Australia. Today, fire managers face the challenge of meeting community expectations for fire protection while simultaneously conserving biodiversity.

The use of prescribed burning for fuel reduction and other land management objectives in the south-west of WA is today much lower compared to the levels of the 1960s and 1970s. Fire managers now face several challenges that limit the use of prescribed burning in the region, including: (1) a growing WUI, (2) increased complexity generated by a variety of land uses in public forests, (3) a narrower climatic window for the application of the treatment, and (4) community resistance and sensitivity to the smoke generated by the burns.

The population in the south-west of WA has increased substantially since the 1960s. Over the past five years, the population in the region has increased at a rate of 2.8% per year and by 2026 it is expected to be 32% larger than in 2014 (South West Development Commission n.d.). This growing population and the ensuing urban development have made the implementation of prescribed burning considerably more complex. Indeed, fuel reductions using prescribed burning are more complicated in the WUI than in wildland areas, because of the spatial interactions between housing and fuels (Bar Massada *et al*. 2011).

Land use changes over the past 60 years have resulted in greater fragmentation of the forests of the south-west of WA (Burrows and McCaw 2013). At present, the forests in public land have a variety of purposes, including biodiversity conservation, recreation, timber harvesting and mining. As a result, prescribed burning needs to integrate all these land uses, and the complexity of the issue has been increased.

A changing climate has caused decreases in average annual rainfall and increases in temperatures in the region, which has modified the timing and duration of weather patterns suitable for prescribed burning (McCaw 2013). Today, the window of opportunity during which the climatic conditions are suitable for the application of prescribed burning is relatively narrow and achieving the same number of burns in fewer days becomes increasingly difficult.

With a larger population living in the area, there is now more awareness of smoke from prescribed burns (McCaw 2013). Communities are concerned about poor air quality (Reisen *et al.* 2011) and some industries have experienced losses attributed to the impacts of smoke from prescribed burns (McCaw 2013). Most of the Australian population is now disconnected from the reality that living in a fire-prone environment might mean experiencing fires regularly, and instead of viewing fire as a natural part of the Australian environment, fire is generally viewed as a threat to life, property and the environment (McCaw 2013). As a result, there is resistance from the community regarding the application of the treatment.

To be successful in the long run, prescribed burning programs require the support from the communities affected (Toman *et al.* 2011). However, public support for or resistance to prescribed-burning programs is not necessarily based on the potential economic or ecological impacts, but usually depends on the nature of the most recent fire event. An increase in prescribed burning to mitigate the negative effects of wildfires is often advocated following disastrous wildfire events (Penman *et al*. 2011). In contrast, restrictions on prescribed burning treatments are imposed after prescribed fires escape and destroy human assets (DEC 2012). Recent wildfire inquiries, escaped prescribed burns and concerns raised by the communities have made prescribed-burning planning, risk management and decision making more complex, further hindering the implementation of the treatment in the south-west of WA (Keelty 2012; Burrows and McCaw 2013).

3.2.5. Concluding remarks on today's fire management in the south-west of Western Australia

The previous sections have shown that today prescribed burning has to balance multiple objectives and integrate additional land management issues. The critical issue is to find the appropriate fire regime that meets all these objectives. In their foreword, Abbott and Burrows (2003) asked a pertinent question: what fire regime do we use? "Is a 'natural' fire regime most appropriate to use? If so, do we adopt a fire regime that applied before the arrival of Aborigines, the practices adopted by Aborigines, some blend, or do we adopt a new regime?" (p. vii). The information provided by the history of fire in the south-west and the current challenges faced by fire managers may provide an answer to this question.

In today's context, a "natural" fire regime would probably not be tolerated by the communities of the south-west of WA, and with humans in the landscape, the risk of fire ignition is always present (Pyne 2003), thus a "natural" fire regime is no longer possible. Fire cannot be avoided in the region, thus it is unreasonable to consider a fire exclusion policy for the region, such as the one applied by the early European settlers. The answer would be that a new regime needs to be implemented, but to find a regime that meets the current multiplicity of objectives and challenges, all the different aspects of the issue (environmental, political, technical and economic) need to be evaluated. This study focuses on one of these aspects: the economic issues relating to prescribed burning. The following section presents the case study area to which the economic analysis has been applied.

3.3. Case study area

The case study area selected for this study is the South West forest region, one of the three forest regions managed by DPaW (see Figure 3.1). The South West forest region covers a total of 1,880,000 hectares and contains a mix of forests, agricultural land, and residential areas (see Section 3.3.1 on Land use below). The region has a long history of prescribed burning and complex fire management issues because of the combination of different land uses and tenures, where human assets are intermingled with flammable vegetation. This makes the South West forest region an interesting area of study for the application of economic analysis to prescribed burning. In this Section, some of the key features of the region are described, and the contemporary fire history and fire management expenditures are reviewed.

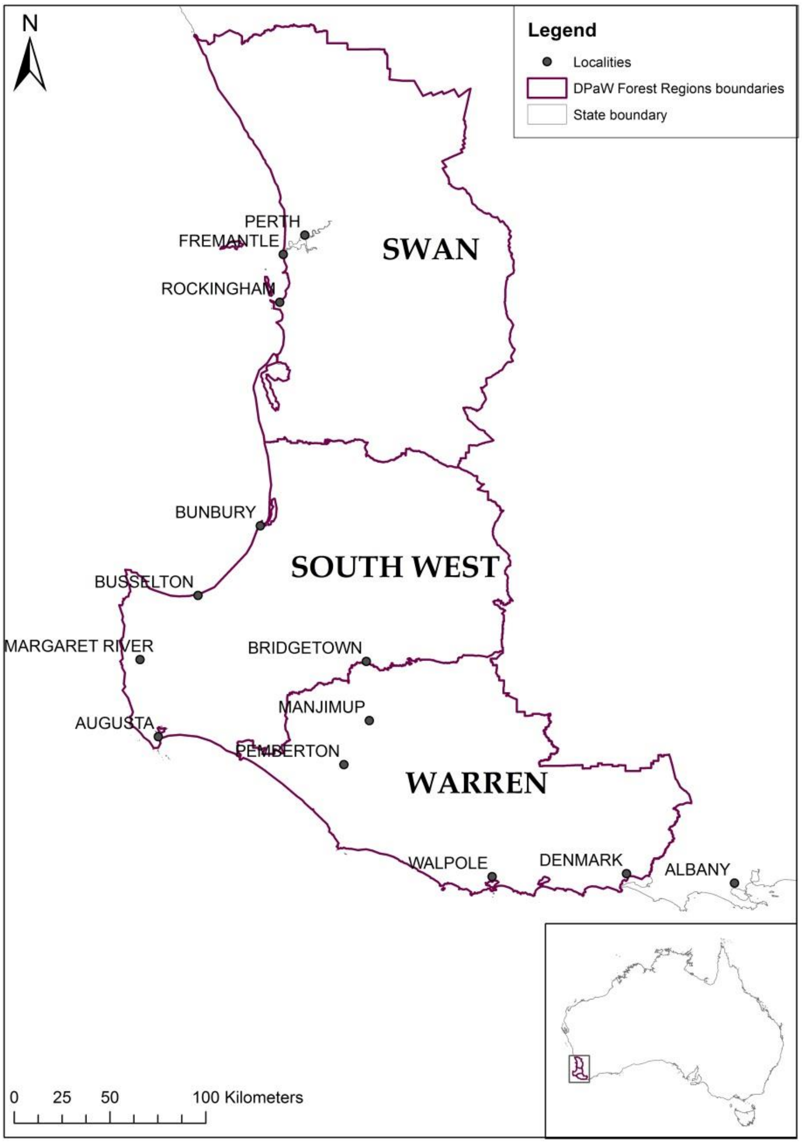

Figure 3.1. DPaW forest regions in the south-west of Western Australia

3.3.1. Land use

The most prominent land use in the South West forest region is Conservation areas and State forests (about 57% of the region, see Figure 3.2 and Table 3.1). These areas include national parks, nature reserves, other protected areas of remnant native vegetation and State forests. The second most prominent land use in the region is Agriculture (about 37%), mostly comprised of pastures for grazing, cropping, seasonal horticulture and irrigated vines. About 78,000 hectares are used for plantation forestry, which corresponds to nearly 5% of the region. Only a small proportion of the land in the region corresponds to urban areas (0.6%). There are two cities located in the coastal areas (Busselton and Bunbury) and several towns in the agricultural and forested areas (see Figure 3.3). About 128,000 people live permanently in the region, but since the region is a major tourist destination, the population in the region increases significantly during the holiday seasons. The region receives on average about 106,000 international visitors and 1.5 million domestic visitors per year (South West Development Commission n.d.).

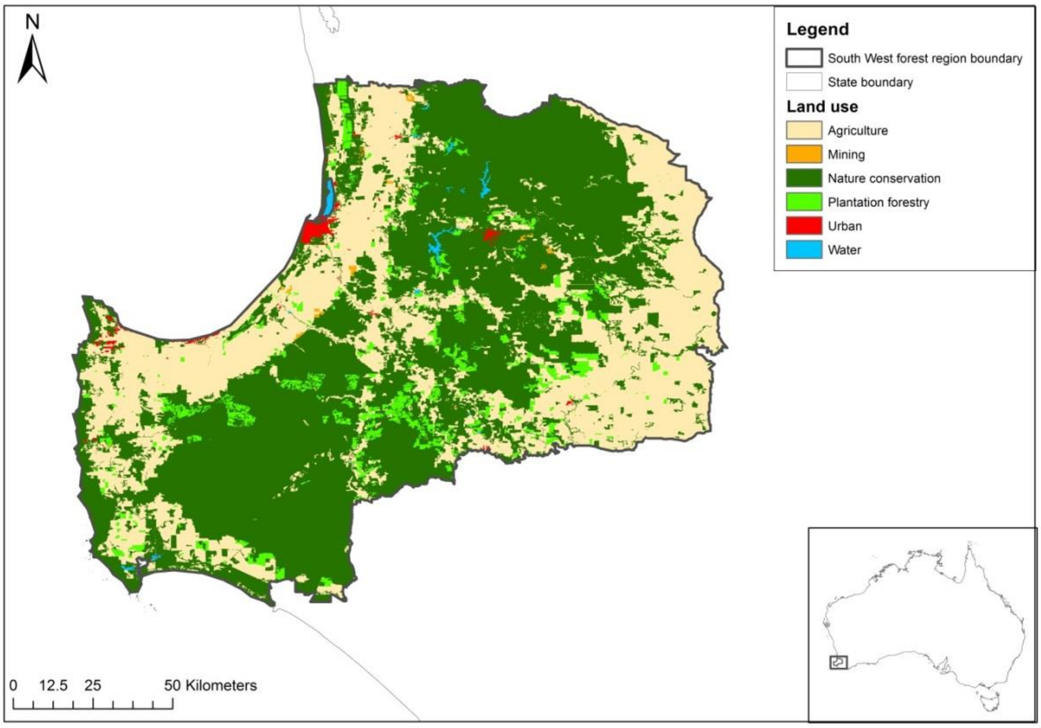

Figure 3.2. Land use in the South West forest region

Table 3.1. Proportions of different land uses in the South West forest region

Land use	Proportion
Conservation areas and State forests	56.9%
Plantation forestry	4.6%
Agriculture	37.3%
Urban	0.6%
Mining	0.2%
Water	0.4%

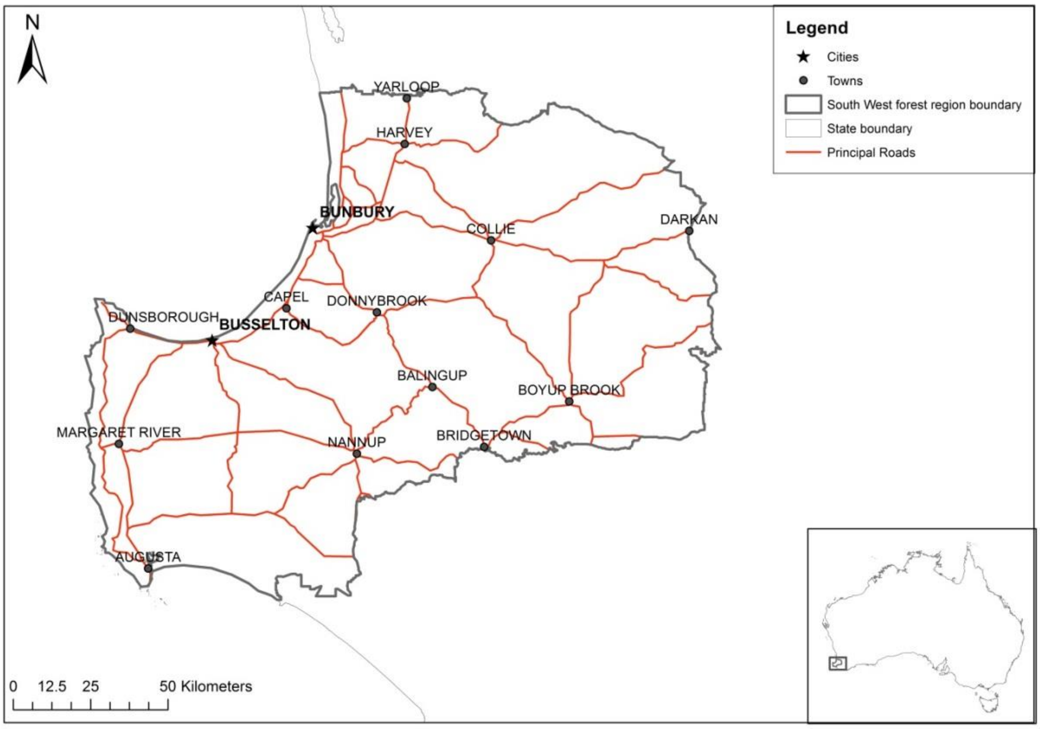

Figure 3.3. South West forest region cities, towns and principal roads

3.3.2. Vegetation and topography

The landscape of the South West forest region is mostly flat with some undulating land in the forested areas. Most of the area (55%) is covered by Eucalypt forests and woodlands (see Figure 3.4), primarily dominated by jarrah (*Eucalyptus marginata*), marri (*Corymbia calophylla*) and karri (*Eucalyptus diversicolor*) (Burrows and McCaw 2013). The vegetation closer to the coastline in the south-western corner of the region is mostly covered by shrublands. Around 42% of the region is either non-native vegetation or has been cleared for agriculture and buildings.

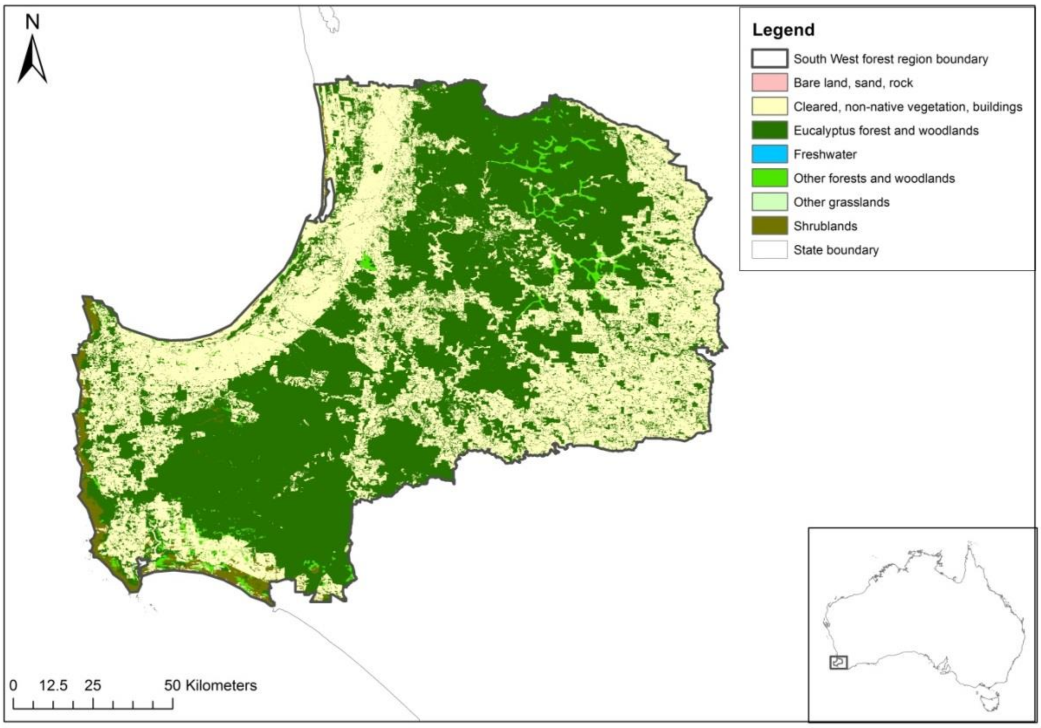

Figure 3.4. Vegetation in the South West forest region

3.3.3. Climate

The climate of the south-west of WA is a Mediterranean type climate (Burrows and McCaw 2013). The region has cool moist winters and dry hot summers. Areas within 50 kilometres of the south coast receive more than 1000 mm of rainfall annually, but rainfall decreases rapidly with distance from the coast. The eastern and northern parts of the South West forest region receive between 600 mm and 1000 mm of rainfall annually. Rainfall in the region is seasonal, with more than 80% of annual rainfall occurring during the wettest months from May to October. During the peak of the fire season (December to February), mean monthly rainfall in the region averages less than 25 mm (Burrows and McCaw 2013). However, since the 1970s, the climate of the south-west of WA has been in a continuous drying phase. Annual rainfall has been reduced by around 20%, mainly due to a reduction in rainfall in autumn and early winter (Burrows and McCaw 2013).

Average maximum temperature from October to April is 21^{o}C to 27^{o}C, but maximum temperatures regularly exceed 35^{o}C during the hottest months of the year (December to February). The rest of the year (May to September) average maximum temperatures are <18^{o}C. For most of the year, prevailing winds are generally easterly and of moderate strength between 20 and 30 kilometres per hour (McCaw and Hanstrum 2003). Between October and

March, dry lightning storms provide a source of ignition for wildfires in the south-west forests. Because of the small amount of rainfall recorded during summer and the high temperatures occurring during this period, the vegetation is usually dry enough to burn for 6 to 8 months of the year and provide the region with a particularly flammable environment (Bowman 2003; McCaw and Hanstrum 2003; Burrows and McCaw 2013).

3.3.4. Biodiversity

Due to its climatic diversity and geomorphology, the south-west of WA is a region of remarkable biological diversity (Burrows 2008). A large area in the south-west of WA has been designated as one of the 35 global biodiversity hotspots in the world (see Figure 3.5, the total area of the Southwest Australia global biodiversity hotspot is indicated in light orange). The case study area (the South West forest region) is located within this hotspot. The forest ecosystems of the region are an important element of the hotspot, which includes approximately 8,000 plant taxa, of which more than 75% are endemic (Burrows and McCaw 2013). Because of the rich biological diversity of the region, biodiversity conservation is an important objective of land management in the area and is now also integrated in fire management.

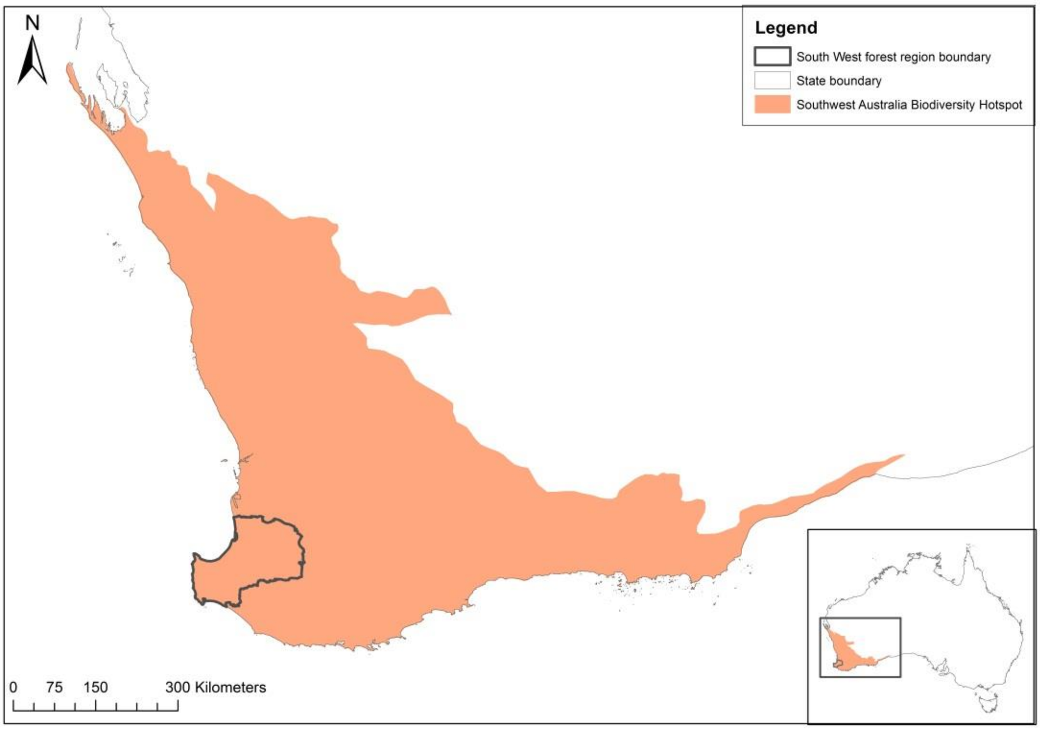

Figure 3.5. Southwest Australia Biodiversity Hotspot

3.3.5. Land managed by the Department of Parks and Wildlife

42% of the South West forest region (790,000 hectares) is managed by DPaW (see Figure 3.6), which for the most part are located in forested areas (compare Figure 3.4 and Figure 3.6). In the land that it manages, the Department undertakes a coordinated approach to prescribed burning.

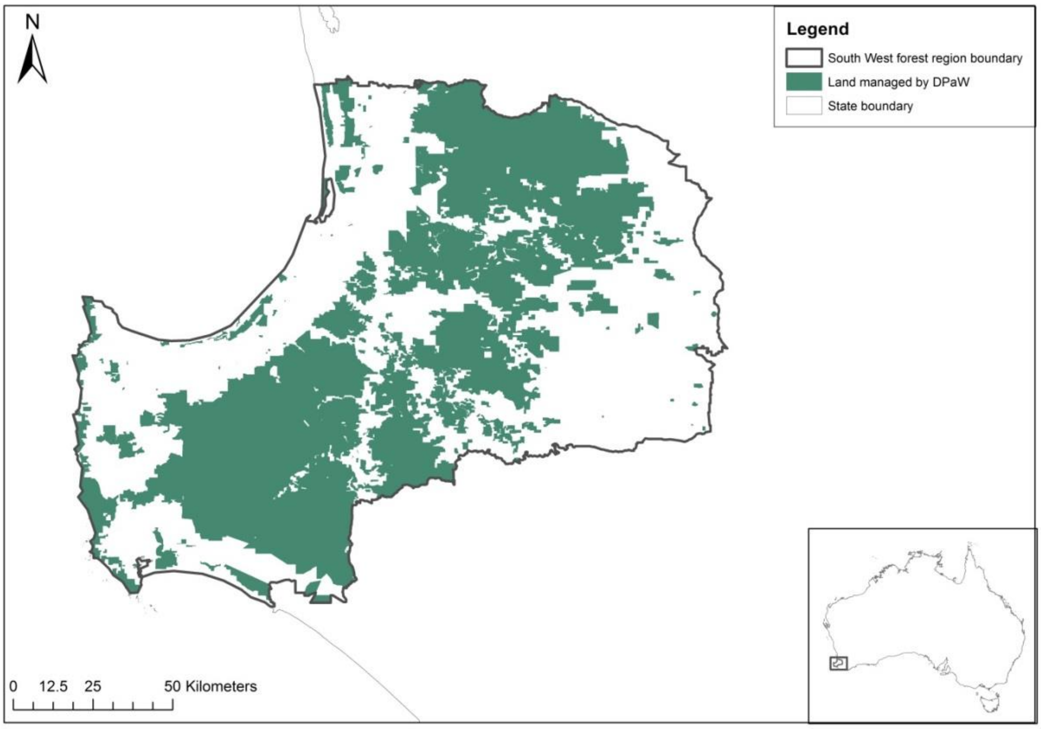

Figure 3.6. Land managed by DPaW in the South West forest region

3.3.6. Contemporary fire history and frequency

Wildfires occur every year in the South West forest region. The region experiences on average about 110 wildfires per year larger than one hectare.[10] Since 1951, wildfires have burned on average 9,600 hectares per year, but the average per decade has varied substantially (see Table 3.2). Large wildfires represent a small number of all fires, but constitute the bulk of the total area burned. Since 1951, about 0.4% of all wildfires burned 47% of total area. Today, most wildfires are caused by humans (about 62%, see Table 3.3) with the vast majority of

[10] Many other ignitions are reported in fire agencies' databases that are smaller than one hectare but these ignitions do not represent a wildfire ignition. Ignitions smaller than one hectare are also reported because fire agencies keep a record of all the incidents that triggered a response from the agency (e.g. after an emergency call a fire officer goes to the area to inspect it and look for a fire but finds nothing).

human-caused fires being deliberately lit (more than 2/3). Lightning is responsible for about 18% of wildfire ignitions.

Table 3.2. Average number of hectares burned by wildfires per year in each decade since the 1960s

Decade	Average hectares burned by wildfires per year
1960s	12,000
1970s	5,300
1980s	3,100
1990s	5,000
2000s	6,500

Source: DPaW fire history database

Table 3.3. Proportion of wildfires by cause 2003-2013

Wildfire cause	Percentage
Human (arson)	44%
Human (accidental)	18%
Unknown*	9%
Escaped prescribed burn (DPaW)	2%
Escaped private burn	8%
Lightning	18%

* Lightning is not suspected

The area prescribed burned per year in the South West region has varied greatly since 1951 (Figure 3.7). The target of prescribed burning 10% of DPaW managed land in the case study area (i.e. 79,000 hectares per year for the South West forest region) has only been reached once in the past two decades. However, up to the 1990s this target was either achieved or exceeded by a significant amount (see Table 3.4). Partly because of the reduction in prescribed burning and aggressive suppression of fires in the last few decades, fuels have accumulated in the landscape of the south-west of WA. As of June 2012, about 70% of the forests in the South West forest region had fuels that had not been burned for over seven years (see Figure 3.8).

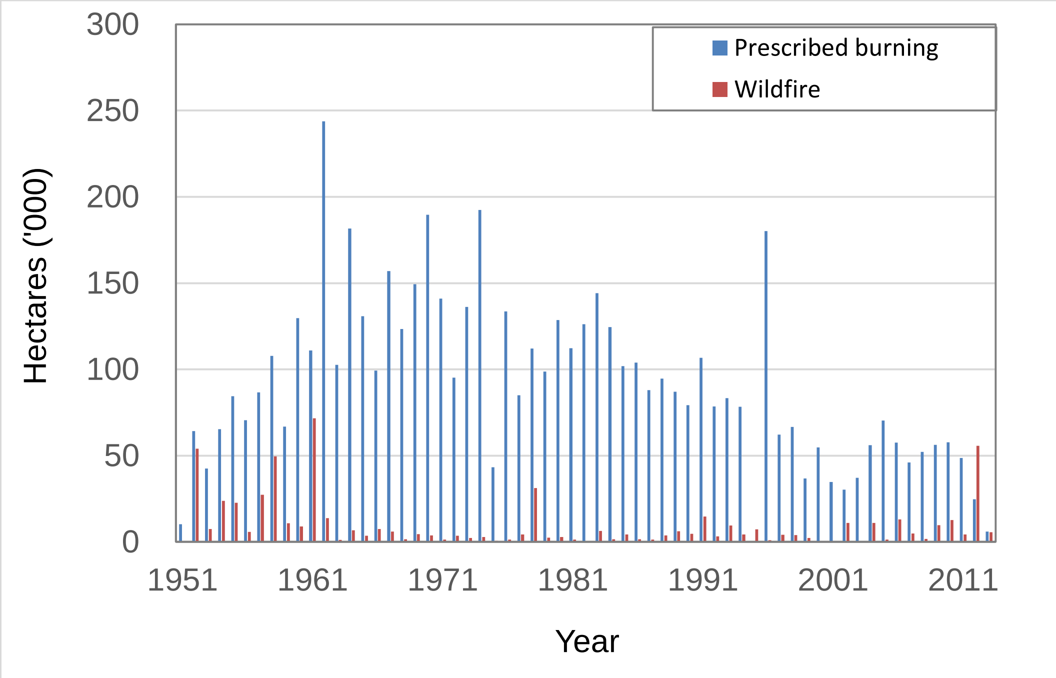

Figure 3.7. Area prescribed burned and area burned by wildfires in the South West forest region 1951-2012

Table 3.4. Average number of hectares prescribed burned per year in each decade since the 1960s

Decade	Average hectares prescribed burned per year
1960s	149,000
1970s	117,000
1980s	106,000
1990s	75,000
2000s	50,000

Source: DPaW fire history database

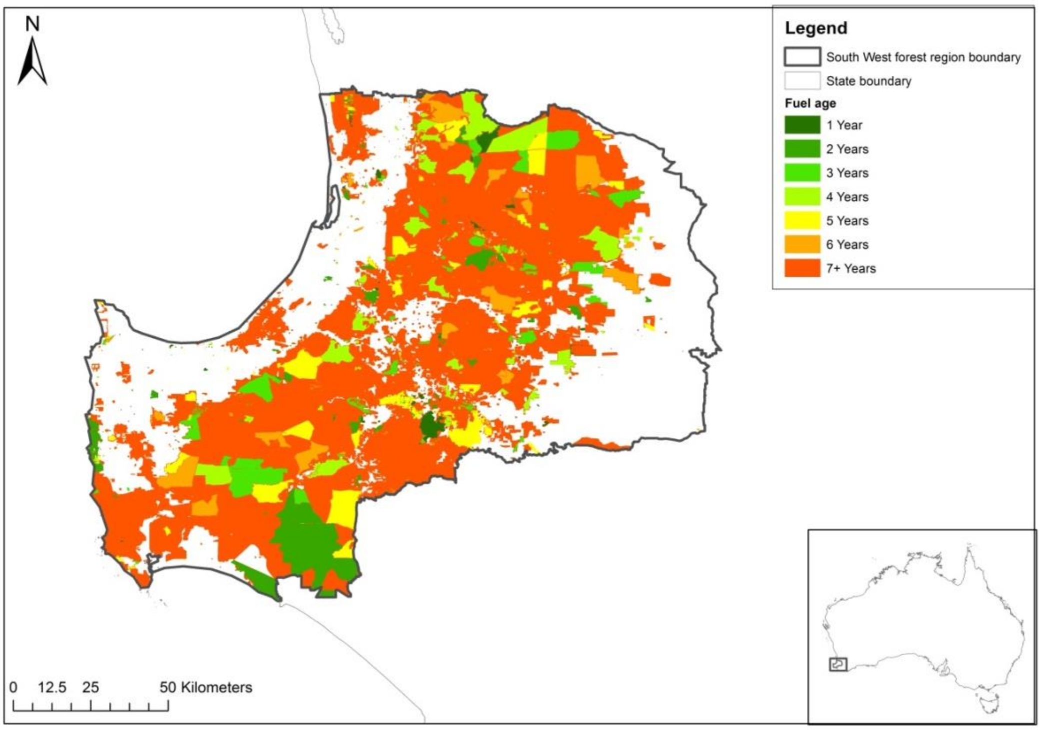

Figure 3.8. Fuel age of forests in the South West forest region in June 2012

3.3.7. Concluding remarks on the case study area

The South West forest region provides a good example of an area where there is an array of fire management challenges. Fire agencies have to use their limited funds to protect different types of assets that compete for protection. The climate and vegetation of the region make it one of the most fire-prone regions in the world. The region presents a complex landscape where large areas of flammable vegetation are intermingled with valuable human assets and where the protection of the natural environment is also a high priority because of the abundance of endemic species. In addition, in 2015, the State government allocated an extra $20 million in funding over four years to DPaW to help meet prescribed burning targets (Government of Western Australia 2015). The complexity of fire management and the substantial increase in funding for prescribed burning makes the South West forest region an interesting case study for the economic analysis of prescribed burning programs in Australia. The following Chapter presents the methodology used in this economic analysis.

Chapter 4.

METHODOLOGY

In order to answer the research questions outlined in the introduction, the methods used here combine economic analysis and simulation. An economic model is used in conjunction with a wildfire simulator to test different prescribed-burning strategies in the South West forest region of Western Australia. The economic model used is a modified version of the cost plus net value change (C+NVC) model which evaluates the welfare changes for varying pre-suppression strategies and is the most widely used model in integrated assessments of fire management programs (Ganewatta 2008; Gebert *et al*. 2008; Chapter 2). A large number of wildfires are simulated under varying climatic conditions for different prescribed-burning strategies using a wildfire simulator developed at the University of Western Australia: the AUSTRALIS wildfire simulator.

This Chapter describes the methodology used in this thesis. Section 4.1 describes the C+NVC model in its usual formulation and the changes that have been made to the model for the analyses conducted in this thesis. Section 4.2 describes the simulator used for the simulations, the modifications done to the simulator for the purpose of this analysis, the inputs it requires and the outputs it produces. Finally, Section 4.3 puts all the pieces together and explains how the outputs of the simulator are used in the economic model and how each economic model parameter is estimated.

4.1. Economic model: the cost plus net value change

The C+NVC is a monetary–based framework that minimises the total sum of pre-suppression costs, suppression costs and net fire damages (Gonzalez-Caban 2007; Venn and Calkin 2011), analogous to a benefit–cost analysis. The underlying assumption in this formulation is that additional investments in pre–suppression result in gains in terms of reduced damages and reduced suppression costs; in other words, damages and suppression costs are assumed to decrease as pre–suppression investment increases. The most efficient level of fire protection is the level at which the total costs of fire management plus net damages are minimised (Donovan and Rideout 2003b; Ganewatta 2008; Rodriguez y Silva and Gonzalez-Caban 2010).

4.1.1. Standard formulation of the C+NVC

In the standard formulation of the C+NVC model, pre-suppression expenditure is the independent variable that determines suppression costs and damages, the latter being outputs of the model. Pre-suppression is variously measured, sometimes including fuel reduction treatments and other times corresponding only to investment in initial attack resources. From Donovan and Rideout (2003b), the C+NVC model is expressed as:

$$\text{Min } C + NVC = W_P P + W_S S(P) + NVC\big(P, S(P)\big) \quad (1)$$

in which W_P is the price of pre-suppression; P is the pre-suppression effort; W_S is the price of suppression; S is the suppression effort, which is dependent on pre-suppression effort; and NVC (net value change) corresponds to net fire damages (fire damages less fire benefits).

In the C+NVC model, damages are assumed to decrease with an increase in suppression expenditure, so the change in NVC relative to a change in S is assumed negative in Equation (1) (i.e. $\partial NVC/\partial S < 0$). Similarly, the change in NVC relative to a change in P is also assumed negative (i.e. $\partial NVC/\partial P < 0$). Since suppression costs are assumed to decrease as pre–suppression investment increases, a marginal increase in pre–suppression denotes a marginal reduction in suppression ($dS/dP < 0$). Figure 4.1 shows a graphical representation of this model (adapted from Gonzalez-Caban 2007 and Donovan and Rideout 2003b).

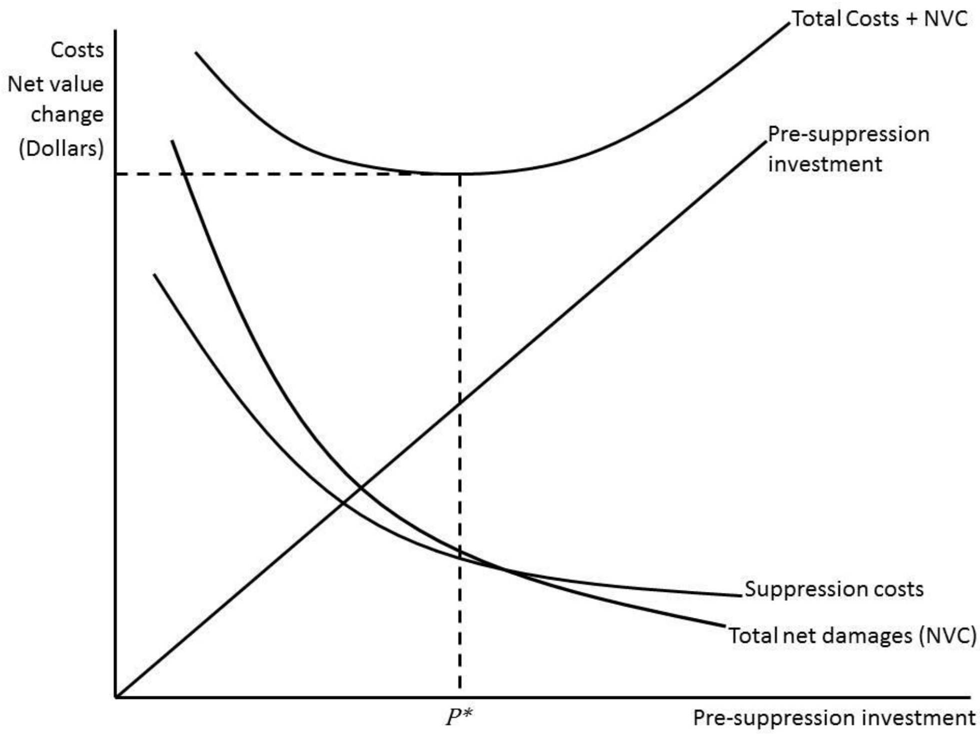

Figure 4.1. The cost plus net value change (C+NVC) model

Generally, the analysis yields a U-shaped function known as the C+NVC curve (Figure 4.1). The minimum point of the C+NVC curve corresponds to the most efficient level of pre-suppression investment (Mills and Bratten 1982; Gonzalez-Caban 2007; Rodriguez y Silva and Gonzalez-Caban 2010). A global extremum may be found when differentiation of $C + NVC$ with respect to P equals zero, that is, when the following first order condition is fulfilled (Donovan and Rideout 2003b):

$$\frac{\partial(C + NVC)}{\partial P} = W_P + W_S\frac{dS}{dP} + \frac{\partial NVC}{\partial P} + \frac{\partial NVC}{\partial S}\frac{dS}{dP} = 0 \qquad (2)$$

The value of P that achieves this corresponds to the level of pre-suppression investment P^*. For the global extremum to be a minimum, the C+NVC function must be convex and satisfy the following second order condition:

$$\frac{\partial^2(C + NVC)}{\partial P^2} = \frac{d^2S}{dP^2}\left(W_S + \frac{\partial NVC}{\partial S}\right) + \frac{\partial^2 NVC}{\partial P^2} > 0 \qquad (3)$$

In the application of the C+NVC model, for each given pre-suppression budget, an average fire season is simulated and suppression expenditures and wildfire damages are calculated (e.g. Rodriguez y Silva and Gonzalez-Caban 2010).

4.1.2. Model modifications

In some studies, pre-suppression corresponds to fuel reduction treatments (Jones *et al*. 1999; Prestemon *et al*. 2001; Mercer *et al*. 2007; Mercer *et al*. 2008; Schaaf *et al*. 2008; Butry 2009; Prestemon *et al*. 2012). In these analyses, initial attack and extended attack are combined into total suppression expenditures and then the trade-offs between fuel reduction treatments and suppression can be quantified (as opposed to trade-offs between initial attack and extended attack). In this thesis, we have adopted this definition of pre-suppression, which considers fuel treatments as the principal pre-suppression management activity and combines initial attack and extended attack into total suppression costs. We have focused on prescribed burning as the pre-suppression strategy to be evaluated.

Instead of evaluating different levels of pre-suppression investment, our modified version of the C+NVC model evaluates varying levels of area prescribed burned. Thus, our model identifies the amount of prescribed burning in hectares that maximises welfare, that is, the amount of area prescribed burned that minimises the sum of management costs (prescribed burning plus suppression) and net damages. The independent variable in our modified C+NVC is the area prescribed burned. Hence, the formulation of the C+NVC presented above has been modified to have the number of hectares prescribed burned as the independent variable and to have prescribed-burning costs, suppression costs (initial attack and extended attack combined) and net damages as the dependent variables.

The reason for using the amount of hectares prescribed burned as the independent variable instead of the amount of dollars invested in prescribed burning is because the area may provide more information to policy makers than the amount of dollars. Since the cost per hectare of prescribed burning varies with the size of the treatments and the distance from the wildland-urban interface (WUI) (Berry and Hesseln 2004; Berry *et al*. 2006; Calkin and Gebert 2006; Chapter 2), a given amount of dollars can be used to prescribe burn many different amounts of area depending on how large the treatments are and how close to towns they are applied. Therefore, the same level of investment in prescribed burning can result in different levels of damages and suppression costs, depending on how large the treatment areas are and where they are located. Thus, the level of investment alone may not provide enough information to policy makers to know which prescribed-burning strategy minimises the sum of management costs and damages. It can be argued that the same logic applies to the amount of hectares prescribed burned. If only the total number of hectares prescribed burned is reported, the same amount of hectares treated may correspond to several different levels of investment for different combinations of treatment size and location. Thus, in this study we report the number of hectares prescribed-burned as well as the details on treatment size and location, so that this can be easily converted to a unique amount of dollars invested in prescribed burning.

In the most recent economic evaluations of prescribed burning, the intensity of wildfires was an important determinant of the level of damages (e.g. Mercer *et al*. 2007; Butry *et al*. 2010). Carrying on with this approach, in our model we also take into account wildfire intensity. As demonstrated by Mercer and colleagues, the results of an economic analysis of prescribed burning depend strongly on whether wildfire damages account for intensity or not. Since the level of damages is determined by both the size and the intensity of the fires, ignoring intensity can result in a suboptimal application of prescribed burning. Hence, in our model the level of damage per hectare depends on the level of intensity with which each hectare burns.

An important development of our study is that the modified version of the C+NVC model is spatially explicit. Wildfires can affect a variety of environmental, social and economic assets in a landscape, and depending on where they occur, they may damage different types of assets. If a wildfire mainly burns forested areas, it is likely to create disturbances in environmental values, biodiversity and recreation demand, but if it occurs close to a town, it would have a greater impact on human assets and social values. Thus the spatial distribution of wildfires is a key element in determining the level of damages. However, previous integrated assessments of fire management have not made the level of damages spatially explicit. Prestemon *et al*. (2001) calculated timber losses in dollars per hectare and applied the same measure of damages homogeneously across the landscape; that is, total damages changed only with the amount of area burned by wildfires. Net value change estimates in the study conducted by Schaaf *et al*. (2008) change with fire size and intensity, but the authors do not specify which resources are included in their net value change estimates and their initial values. Mercer *et al*. (2007) calculated damages in dollars per hectare of area burned based on the findings of Butry *et al*. (2010). Although Mercer and colleagues included other types of damages, such as damages to buildings and losses to the tourism industry, they also used the estimated dollars per hectare homogeneously across the landscape. In their model, the level of damages changes depending on the amount of area burned and the level of intensity, but not depending on the location of the wildfires. In the modified C+NVC model used in this thesis, we have incorporated a spatial dimension and estimated damages (in dollar per hectare) depending on where the fires burn and the type of assets they destroy.

Another important development in the present study is that we account for the change in prescribed burning cost per hectare based on both the size of the treatment and the closeness to towns. These spatial attributes have not been considered in previous studies. The existing integrated economic studies of prescribed burning have generally used a constant average cost per hectare of prescribed burning (e.g. Jones *et al*. 1999; Schaaft *et al*. 2008; Butry *et al*. 2009), that in some cases changes with the total area prescribed burned (e.g. Mercer *et al*. 2007; Butry *et al*. 2010) but none have considered the changes with distance from the WUI. However, the costs of prescribed burning treatments per hectare have been shown to increase when they are located in or close to the WUI (Berry and Hesseln 2004; Berry *et al*. 2006; Calkin and Gebert 2006; Chapter 2) and decrease for larger treatment size (Calkin and Gebert 2006; Chapter 2). In our model, we have included changes in prescribed burning costs per hectare with both treatment size and distance from the WUI.

In addition, we have separated suppression costs and damages to be able to evaluate the trade-offs between prescribed burning, suppression and damages. In wildfire management, trade-offs exist between the level of suppression and fuels management (Mercer *et al*. 2007; Schaaf *et al*. 2008; Butry *et al*. 2009), but few studies have actually investigated these trade-offs. In fact, the consideration of suppression costs in previous integrated economic analyses varies greatly. Some studies have not included suppression costs in their analysis (Prestemon *et al*. 2001), while others have used measures of suppression effort, such as fire-crew response time in hours, for which any changes in the level of effort cannot be easily converted into dollar estimates (Butry 2009). In cases where suppression costs per hectare of wildfire burned have been considered, they have been aggregated with damages per hectare and the trade-offs between fuel management and suppression were not identified (e.g. Mercer *et al*. 2007). Only Schaaf *et al*. (2008) investigated the interactions between fuel-treatment costs, suppression costs and damages by considering suppression costs and damages separately. In our study, we have adopted a similar approach to Schaaf and colleagues' and have estimated the average suppression costs per hectare separately from damages per hectare. In this way, we are able to analyse the trade-offs between prescribed burning, fire suppression and damages.

The formulation of our modified C+NVC is as follows. Let P be the number of hectares prescribed burned and W_P the average prescribed-burning cost per hectare. To account for the changes in the price per hectare with closeness to towns and the size of the treatment, we have multiplied the price per hectare by coefficients that reflect the changes in each attribute. Thus β_i is a coefficient that depends on distance i from any town ($i = 1, \ldots, 4$ where $1 =$ less than 5 km, $2 =$ between 5 and 10 km, $3 =$ between 10 and 20 km, and $4 =$ more than 20km), with $\beta_1 > \beta_2 > \beta_3 > \beta_4$ so that the price per hectare decreases as the distance from any town increases. For each prescribed burning treatment, we estimate the distance from the centre of the treatment to the closest town to decide which coefficient should be applied. Conversely, μ_x is the coefficient that depends on the size of the treatment, with x corresponding to the treatment size class ($x = 1, \ldots, 4$ where $1 =$ less than 100 ha, $2 =$ between 100 and 500 ha, $3 =$ between 500 and 1,500 ha, and $4 =$ more than 1,500 ha). In this case, $x_1 > x_2 > x_3 > x_4$ so that the price per hectare for prescribed burning decreases as the size of the treatment increases. How these coefficients were estimated and the distance from towns or treatment size range that corresponds to each coefficient is explained in Section 4.3.1. Let A be the total number of hectares burned by wildfires.

Total prescribed-burning costs $PB\ costs(P)$ depend on the amount of area prescribed burned, P, and the location and size of the treatments, reflected by the coefficients β_i and μ_x. Let P_{ix} be the number of hectares prescribed burned in treatments located at distance i from towns and of size x.

Since fire size is one of the most important determinant of suppression costs (Calkin *et al*. 2005; Gebert 2007; Donovan *et al*. 2008; Liang *et al*. 2008; Chapter 2), in our model total suppression costs depend on the amount of area burned by the fires. Suppression costs were calculated for each fire season by multiplying the average suppression cost per hectare W_S by the number of hectares burned A_{km}. In Equation (6), A_{km} corresponds to the amount of hectares burned by fire k ($k = 1, \ldots, K$) in fire season m ($m = 1, \ldots, M$), and M is the total number of fire seasons simulated. Thus, total suppression costs for each strategy $SUP\ costs\ (A(P))$ correspond to the average suppression costs for all the fire seasons simulated.

NVC is the net value change (net fire damage) cause by the fires, which depends on the amount of hectares burned by the fires, the location of the fires, and their intensity. Similar to suppression costs, the total net value change for each strategy $NVC(A(P), I(P))$ corresponds to the average net damages for all the fire seasons simulated. In Equation (7), A_{jkm} is the amount of hectares burned in land-use category j by fire k in fire season m; Z_j is the value in dollars of a hectare located in land-use category j; and η is the percentage of value destroyed depending on the level of intensity I $(0 \geq \eta \leq 1)$. The different land-use categories and the base values used for each category are presented in Section 4.3.3.

The model can then be expressed as:

$$\text{Min } C + NVC = PB\ costs(P) + SUP\ costs\ \big(A(P)\big) + NVC\big(A(P), I(P)\big) \tag{4}$$

with

$$PB\ costs(P) = \sum_{i=1}^{4} \sum_{x=1}^{4} \beta_i \mu_x W_P P_{ix} \tag{5}$$

$$SUP\ costs\ \big(A(P)\big) = \frac{1}{M} \sum_{m=1}^{M} \sum_{k=1}^{K} W_S A_{km}(P) \tag{6}$$

$$NVC\big(A(P), I(P)\big) = \frac{1}{M} \sum_{m=1}^{M} \sum_{k=1}^{K} \sum_{j=1}^{J} A_{jkm}(P) Z_j \eta\big(I(P)\big) \tag{7}$$

Advantages of this method

A significant advantage of using this method is that it makes it possible to evaluate the spatial distribution of the treatments. Depending on where the burn treatments are located in the landscape, they help protect different assets. As a result, changing the spatial allocation of the treatments may have a significant effect on wildfire consequences (Bar Massada *et al.* 2011; Gibbons *et al.* 2012). However, this has not been evaluated in previous studies. With our model, it is possible to test different strategies with different locations for the treatments. For instance, it is possible to compare a strategy where the treatments are primarily located in forested areas with a strategy where more treatments are concentrated around towns to protect highly valued human assets. With this method, we can investigate whether there are additional economic benefits from prescribed burning close to towns and whether the benefits outweigh the costs.

Another advantage of this method is that it uses a common unit of measure to evaluate wildfire impacts on different types of assets. In this analysis, wildfire impacts on human infrastructure, ecological values, agricultural land and forestry plantations are appraised in the same currency (i.e. dollar values). By using a common unit of measure, it is possible to estimate total expected wildfire impacts and evaluate the cost and benefits of different management strategies (Finney 2005). A common unit of measure also makes the quantification of trade-offs possible, both between the values protected and between different policy objectives. With improved knowledge of the existing trade-offs between alternative management options, fire managers will have more tools available to make evidence-based decisions and develop sustainable fire management programs.

4.2. AUSTRALIS wildfire simulator

One of the factors that restrict research on wildfire management is the lack of data for some of the parameters of interest (Thompson *et al.* 2013a; Milne *et al.* 2014). Historical data is generally not available for some of the options that we may want to evaluate, such as zero prescribed burning or zero suppression effort. For prescribed burning, we lack the empirical data that would allow a model to test the prescribed-burning strategies that have never been implemented. Hence we need to rely on simulation, using the current available knowledge about wildfire behaviour, to test a broader range of prescribed-burning strategies–beyond those that have been applied in the past–and assess their effects on small and large wildfires. Previous research has used simulation effectively for evaluating different fire management programs, including studies by Jones *et al*. (1999), Omi *et al*. (1999), Prestemon *et al*. (2001), Mercer *et al*. (2007), Schaaft *et al*. (2008), Liu *et al*. (2010), and Stockmann *et al.* (2010a). In this thesis, we also use simulation to evaluate more prescribed-burning strategies than the current available data would allow. In this way, all aspects of physical fire behaviour modelling are integrated into the economic model through the simulator.

The fire simulations performed for this analysis under different scenarios for prescribed burning were done using the AUSTRALIS high-performance wildfire simulator. AUSTRALIS was developed at the School of Computer Science and Software Engineering, University of Western Australia, in cooperation with the Western Australian Land Information Authority (Landgate) and the Department of Fire and Emergency Services (DFES) (Milne *et al.* 2012; Simulation and Modelling Research Group n.d.).

AUSTRALIS is an interactive wildfire simulator that predicts the spread of wildfire across a landscape. It uses a range of information, including topographic, climatic and vegetation variables, to forecast the progression of a discrete fire event and calculate the area burned by the fire and its intensity at different points in time. The simulator was designed to assist fire managers in decision making for incident response by providing them with a rapid forecast of the spread of a fire and the location of future perimeters at different points in time. The simulator generates detailed spread maps that can be used by incident managers to make decisions regarding the deployment and positioning of fire-fighting resources, or for the issuing of alerts. At an operational level, it can be used as a predictive tool for wildfire incident management or as a risk analysis tool to explore the outcomes of different "what-if" scenarios for alternative weather patterns. At a strategic level, it can be used as a cost-effective complement to field experiments with real fire and as a planning tool to evaluate risk assessment strategies and the effectiveness of fuel reduction treatments.

AUSTRALIS is currently used for these purposes by incident managers in DFES. It has an interface that projects the simulated fire on a map as a series of perimeters that show the progression of the fire over time (see Figure 4.2). The interface has different sections or panels where a number of inputs (i.e. weather, duration of the simulation, geographic coordinates of the ignition point, and cell size) can be modified.

AUSTRALIS requires a number of datasets. These include topographic maps, vegetation maps, fuel-load maps, a rate-of-spread model for each vegetation type, weather conditions for the duration of the fire event, and the time and location of the ignition (or current fire perimeter). Some of these datasets, such as topographic data, vegetation type and rate-of-spread model, are already embedded in the simulator, while other datasets have to be provided by the user. Details for each dataset and their respective sources are summarised in Section 4.2.2.

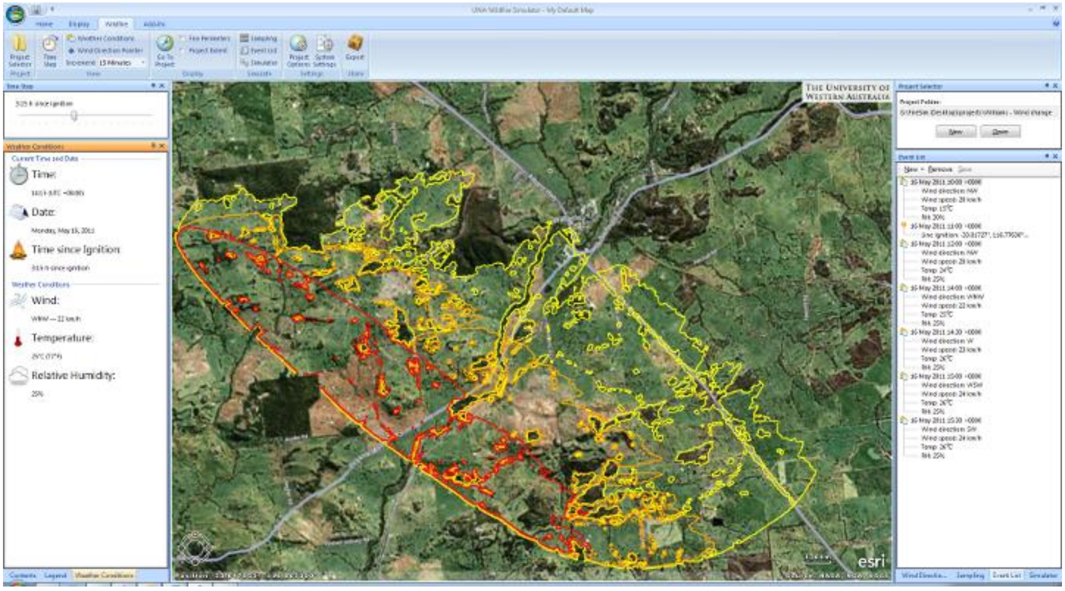

Figure 4.2. The interface of AUSTRALIS wildfire simulator

The lines represent the perimeter of the fire at different times after ignition. Red: 8 hours after ignition; orange: 16 hours after ignition; yellow: 24 hours after ignition.

For the simulation of a fire event, AUSTRALIS first divides the landscape into a collection of cells. While many fire simulators generate regular grids with square cells of equal size, AUSTRALIS generates an irregular grid, where the cells are of similar size, but have different shapes (see Figure 4.3). The size of the cells is defined by the user; the cells can be as small as one square metre or as large as wanted. With every increase in the size of the cells, there is a loss of accuracy in the final shape of the fire. However, the smaller the size of the cells, the longer it takes to simulate each fire. Thus there is a trade-off between the accuracy of the simulation and the computational time required to run the simulations, so a balance has to be found between both.

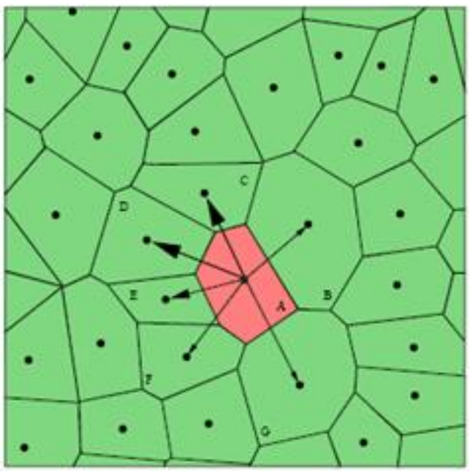

Figure 4.3. AUSTRALIS irregular cell grid

AUSTRALIS uses an irregular grid because previous research has found that a regular grid, despite ease of implementation and increased performance, causes artificial distortions to the shape of the fires generated by some simulators (Kourtz and O'Regan 1971; Sullivan and Knight 2001; Pastor *et al.* 2003). An irregular grid addresses this problem and results in simulations that more closely approximate the shapes of observed fires (Johnston *et al*. 2008).

Each cell in the grid represents an area in the landscape (a polygon). With each simulation, a new grid is created in the form of a GIS layer containing all the information for each cell (polygon) in the layer's attributes; namely slope, elevation, vegetation type and fuel loads. When a cell is ignited and the conditions are favourable for the fire to spread, the fire moves in the direction of the wind to the neighbouring cells. The neighbouring cells catch fire after a time delay that accounts for the time it takes for the fire to travel the distance between the cells. The simulator uses empirically derived fire-spread models for different types of vegetation to estimate the spread time between cells and the intensity with which they burn. Thus, for each simulation we know which cells were burned, where those cells are located in the landscape, their physical attributes, the assets they contain and the intensity with which they are burned.

4.2.1. Modifications made to the simulator

AUSTRALIS was designed to perform discrete-event simulations, where an operator uses the interface to manually input the current and forecast weather conditions (i.e. temperature, wind direction, wind speed and relative humidity) and the time and location of the ignition (or the current fire perimeter) for each fire that needs to be simulated. For the analysis presented in this thesis, however, it was necessary to simulate a large number of fires (in the order of thousands), and it was not practical to manually input the information fire by fire. Furthermore, the simulator had a fuel layer embedded in it with the current fuel levels in Western Australia (up to June 2012), but this layer could not be modified in a rapid and practical manner. In order to test different prescribed-burning strategies, the fuel layer needed to be easily modifiable. Finally, the simulator randomly generated a new cell grid each time a simulation was initiated, and as a consequence, two simulations with the exact same parameters could produce different outcomes; but for the purpose of this analysis we needed to control for this variation and make the changes in outputs depend solely on the changes to the fuel levels achieved through prescribed burning.

In order to adapt the simulator to the analysis here presented, we worked with a computer programmer over the course of six months to address the limitations mentioned above. To be able to simulate a large number of fires, we created a system that automated the simulations and ran all fires consecutively for the indicated prescribed-burning strategies, ignitions and weather conditions. Instead of using the simulator's existing interface, a new interface was developed where the parameters of all the simulations could be specified. In this new interface, it was possible to tell the simulator which fuels layers to use as inputs, which ignitions and which weather conditions. In this manner, the fuel layer used for the simulations could be easily modified. In addition, this new system addressed the variation in the cell grid and created an identical cell grid for each simulation in a set of model runs, so that the simulations could be compared between them.

Figure 4.4 is a graphic representation of this system. In the new interface, first we selected the fuel layers on which the fires were to be simulated (i.e. one fuel layer per prescribed-burning rate: 0, 5, 10, 15, and 20% of DPaW managed land). Then several GIS point layers containing the ignitions for a whole fire season (110 ignitions, the average annual number of ignitions >1 ha) were inputted into the simulator. Each of these layers with 110 ignitions is represented in Figure 4.4 as an 'ignition pattern'. A total of 22 ignition patterns were fed into the simulator (more information on these ignition layers is provided in Section 4.2.3). The fires in each of these ignition patterns were simulated under 10 different 'weather years', corresponding to the weather conditions experienced in the area in the years between 2003 and 2013 (i.e. weather 1 corresponds to the weather in the region in the year 2003-2004, weather 2 corresponds to the year 2004-2005, and so on).

After the simulations were run, the simulator produced two outputs: area burned and intensity (i.e. the cells in the landscape that were burned by the wildfires and the fire intensity of each cell). We worked with the computer programmer to automate the extraction of these outputs in the form of GIS layers that could then be easily overlaid on top of the land use layer that contained the information on the location of the different assets in the landscape. For any selected prescribed-burning rate, a total of 220 'fire seasons' were simulated, given by 22 ignition points, each simulated with 10 'weather years'. A simulation run of 220 fire seasons for one single prescribed-burning rate (i.e. 24,200 fires) took, on average, 2 to 3 days (around 60 hours all together when uninterrupted).

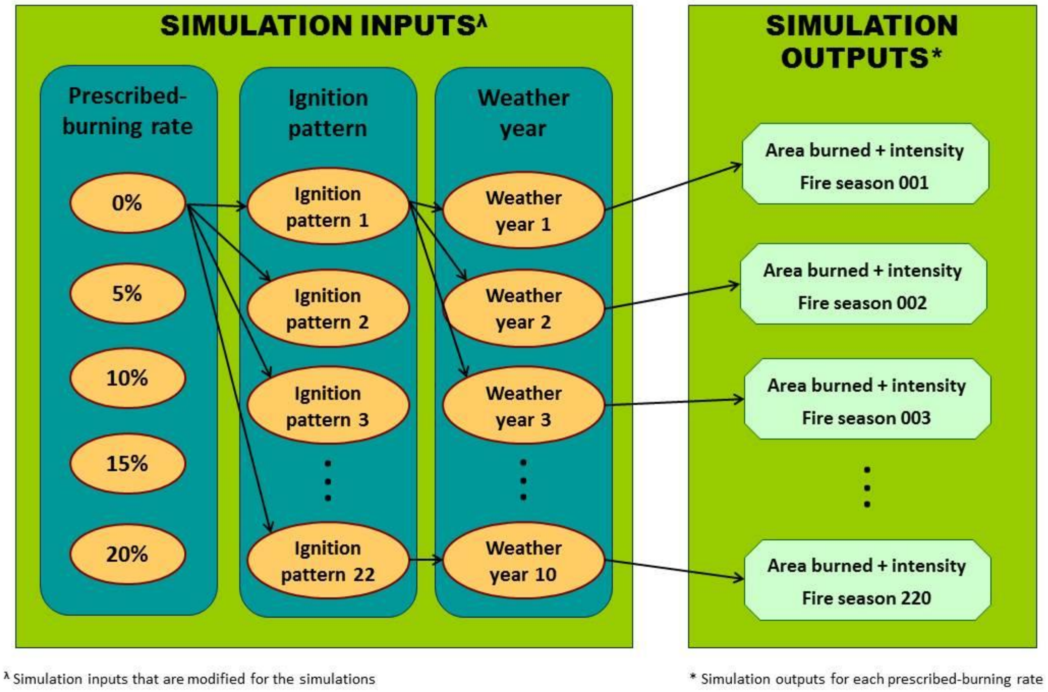

Figure 4.4. Automated simulations system

4.2.2. Datasets embedded in the simulator

The simulator has a number of datasets embedded in it. To run a simulation, the simulator extracts information from these datasets to calculate the rate of spread, direction of spread, and intensity of the fire in each cell. A summary of these datasets is provided below. For more detailed description of the simulator see Johnston *et al*. (2008) and Kelso *et al*. (2015).

Elevation and slope

Elevation was obtained from the STRM digital elevation map of Australia (Joel Kelso, personal communication), a raster dataset with a spatial resolution of approximately 90m x 90m (Farr *et al.* 2007).[11] Slope is calculated from this digital elevation model (DEM) using GIS tools.

Vegetation and fire-behaviour model

For simulating fires on land that is covered by native vegetation, the simulator uses the vegetation data provided by the Australian Government in the National Vegetation Information System (NVIS) (Joel Kelso, personal communication). The NVIS dataset provides

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[11] This Digital Elevation Model is derived from the Shuttle Radar Topography Mission (SRTM).

summary information on Australia's present (extant) native vegetation modelled from data originally supplied by all Australian states and territories. Depending on the type of vegetation, the model uses a different fire-behaviour model to determine the rate of spread. Table 4.1 shows the fire-behaviour model used for each type of vegetation.

Table 4.1. Vegetation type and fire-behaviour model

Vegetation type	Fire-behaviour model
Eucalypt woodland[a]	McArthur Mk 5 (Noble *et al.* 1980)
Grass and land that has been cleared for agriculture	Grassland model (Cheney *et al.* 1998)
Mallee-heath shrublands	Mallee-heath shrubland model (McCaw 1997; Plucinski 2003)
Banksia woodland	Banksia low woodland model (McCaw and Burrows 1989)

[a] The vast majority of the south-west forests are composed of eucalypt forests and woodlands (see Chapter 3, Section 3.3.2).

4.2.3. Other datasets used for the simulations

Fuel loads

In a standard run of the model, current fuel loads are employed. However, for our purposes, prescribed-burning strategies alter current fuel loads, and hence they have to be manipulated. Fuel loads are calculated for each cell depending on how much time has elapsed since the vegetation in the cell was last burned. The simulator uses the fuel accumulation tables in Sneeuwjagt and Peet (1998), which specify fuel load as a function of fuel age for different types of vegetation.

Initially, we use the fuel levels as they were in June 2012 in the case-study region as the base layer to be modified with the different prescribed-burning strategies. The historical information on the time since last burn for this layer was obtained from the Department of Parks and Wildlife (DPaW). DPaW's fuel-age dataset is a detailed and comprehensive GIS layer that contains records of wildfires in the south-west of WA and records of prescribed burning in DPaW's managed land since 1937 (Hamilton *et al.* 2009).

Changes to the time since last burn in this base layer correspond to the different prescribed-burning strategies tested; that is, changing the time to last burn to zero years is equivalent to simulating an area being prescribe burned this season. Five different rates of prescribed burning were tested, some of which have never been applied in the South West forest region. The five rates of prescribed burning correspond to treating 0%, 5%, 10%, 15% and 20% of land managed by DPaW in the South West forest region (approximately 790,000 hectares). Table 4.2 shows the number of hectares treated for each of the prescribed burning rates tested. How each of these treatments rates were distributed across the landscape is explained in Section 4.3.1.

Table 4.2. Hectares treated for each rate of prescribed burning applied

Prescribed burning rate	Hectares treated
5%	39,500
10%	79,000
15%	118,500
20%	158,000

Weather

The simulator requires four climatic parameters for the whole duration of each fire to run the simulations. These four parameters are temperature, relative humidity, wind speed and wind direction. In order to run simulations with average weather seasons, historical weather data was obtained from the Australian Bureau of Meteorology for all weather stations located within or close to the South West forest region. Figure 4.5 shows the location of all weather stations in the region for which data was available for at least 10 consecutive years and Figure 4.6 shows the average minimum and maximum temperature and average rainfall for four selected stations. The weather dataset contains hourly data for all parameters for all weather stations from 2003 to 2013. This dataset was divided into 10 weather years from July to June of the following year, which corresponds to a fire season (winter to winter) in the southern hemisphere. All ignitions were simulated with the 10 different weather years.

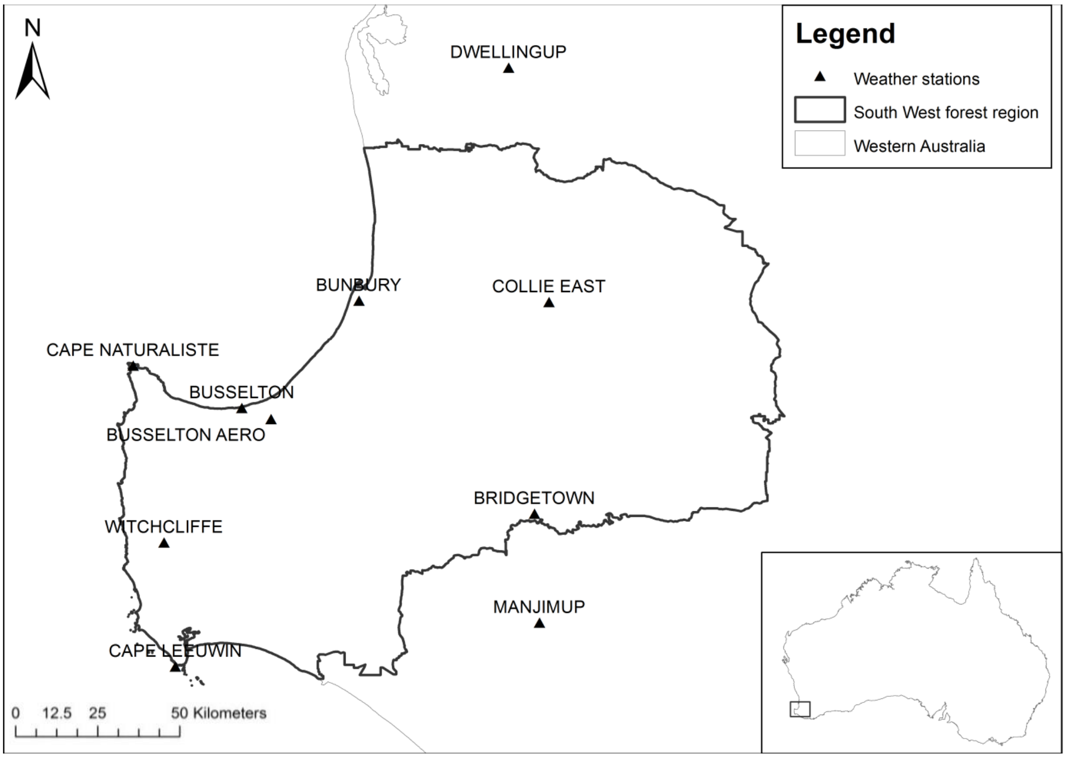

Figure 4.5. Weather stations located within and close to the South West forest region

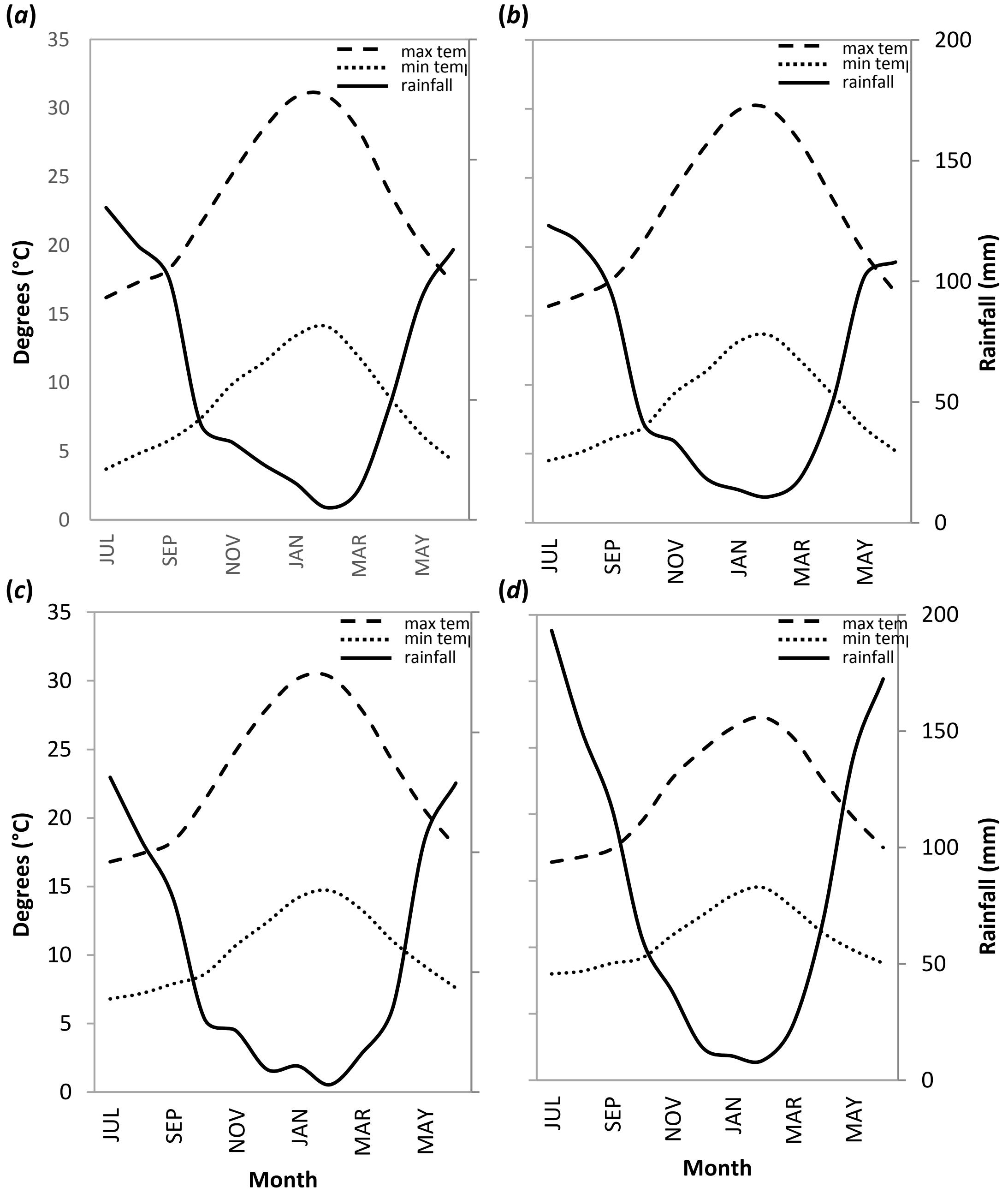

Figure 4.6. Mean maximum temperature, mean minimum temperature and mean rainfall for four weather stations located within the South West Forest Region

The weather stations are (*a*) Collie East, (*b*) Bridgetown, (*c*) Busselton Aero, and (*d*) Witchcliffe

For each ignition it was assumed that the weather at the time of ignition and subsequent hours was equal to the weather of the closest weather station. The extent covered by each weather station is shown in Figure 4.7. The size of the area covered by a weather station ranges between 247 km^2 (area covered by the Cape Naturalist weather station) and 5,483 km^2 (area covered by Collie East).

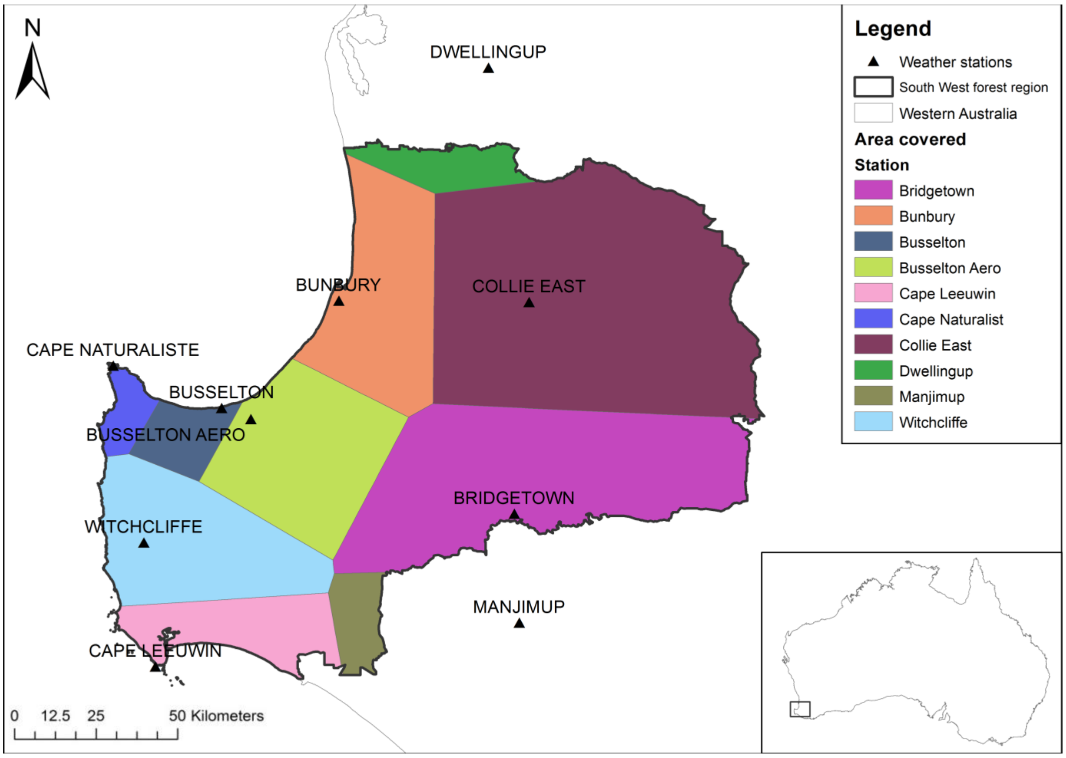

Figure 4.7. Area covered by each weather station

Ignition location

To determine the location of ignition points for an average fire season, we created a probability raster dataset that predicts where ignitions are most likely to occur in the South West forest region. We used a logit model to calculate the values of the cells in this raster dataset, using ignition location data between 2003 and 2013 obtained from DPaW. The model estimates the probability of ignition of a given cell depending on its distance from roads, vegetation type, land-use, topography and fuel loads. Table 4.3 shows the results of the logit model. These are consistent with the existing literature on ignition probability, which shows that the probability of an ignition decreases as the distance from towns or from roads increases (Arndt *et al.* 2013; Plucinski *et al.* 2014; Barreal and Loureiro 2015).

Using this raster dataset, we generated ignition points according to the probability of ignition of each cell in the landscape and the average number of ignitions that occur in a fire season in the region (an average of 110 ignitions per year). A total of 22 ignition patterns were created in separate GIS layers, each ignition pattern containing 110 ignition points (i.e. each ignition pattern corresponds to one fire season).

Table 4.3. Logistic regression for historical ignitions points between 2003 and 2013

Variable	Coefficient	Std. Err.	z	P >|z|	[95% Confidence interval]	
constant	-6.792761	0.283888	-23.93	0.000	-7.349172	-6.236350
clear_land	-3.361737	0.271805	-12.37	0.000	-3.894465	-2.829010
l_roads	-0.562653	0.037522	-15.00	0.000	-0.636195	-0.489111
slope	0.049155	0.006851	7.18	0.000	0.035728	0.062583
year						
2004-2005	0.064544	0.254143	0.25	0.800	-0.433567	0.562655
2005-2006	0.383030	0.236783	1.62	0.106	-0.081055	0.847116
2006-2007	0.262389	0.242858	1.08	0.280	-0.213603	0.738380
2007-2008	-0.033904	0.260426	-0.13	0.896	-0.544329	0.476520
2008-2009	0.530685	0.230101	2.31	0.021	0.079695	0.981675
2009-2010	0.336505	0.239057	1.41	0.159	-0.132038	0.805048
2010-2011	0.659322	0.224900	2.93	0.003	0.218525	1.100118
2011-2012	0.875583	0.217319	4.03	0.000	0.449645	1.301520
2012-2013	1.173695	0.208921	5.62	0.000	0.764217	1.583173

Log likelihood = -5032.14	Number of observations	=	10,111,330 pixels
	LR chi2(12)	=	731.20
	Prob > chi2	=	0.0000
	Pseudo R2	=	0.0677

Variables: clear_land is a dummy variable that denotes if the unit area (the pixel on a 50 metres resolution) has native vegetation or if it has been cleared for agriculture; l_roads is the log of the distance to the nearest road; slope corresponds to the steepest slope of a cell relative to its eight surrounding neighbours; year is a dummy variable for the year in which the ignition occurred used to observe which years had significantly more ignitions, here the fire season 2003-2004 was used as the base.

The fires occurring on average weather conditions and the fires occurring in more extreme weather conditions were simulated separately because they had a different stopping rule (explained in Section 4.2.4 below). In order to combine these two sets of simulations and process the results for a complete fire season that included both types of fires (i.e. occurring under average weather conditions and occurring under extreme weather conditions) we used the historical probabilities of wildfires occurring under extreme weather conditions according to their resulting size. Fires >1,500 ha but ≤5,000 ha occur on average every year with a probability of 0.7; fires >5,000 ha but ≤20,000 ha occur on average every 3 years, with a probability of 0.3; and fires >20,000 occur on average every 10 years with a probability of 0.1. For each fire season, one of the fires simulated under extreme weather conditions was added to the fires simulated under average weather conditions. The fire in extreme weather conditions was selected at random according to its probability of occurrence.[12]

Ignition date and time

In the south-west of WA, the peak fire season extends from October to April, with the majority of the fires occurring between December and February (Burrows and McCaw 2013, Plucinski *et al*. 2014). Figure 4.8 shows the distribution of wildfire ignitions throughout the year for the case-study region between 2003 and 2013. Since the timing of ignitions approximates a normal distribution centred around January, we generated random dates according to this distribution for all ignitions in each fire season. The ignition hour was generated at random between 9am and 5pm from a normal distribution centred at midday.

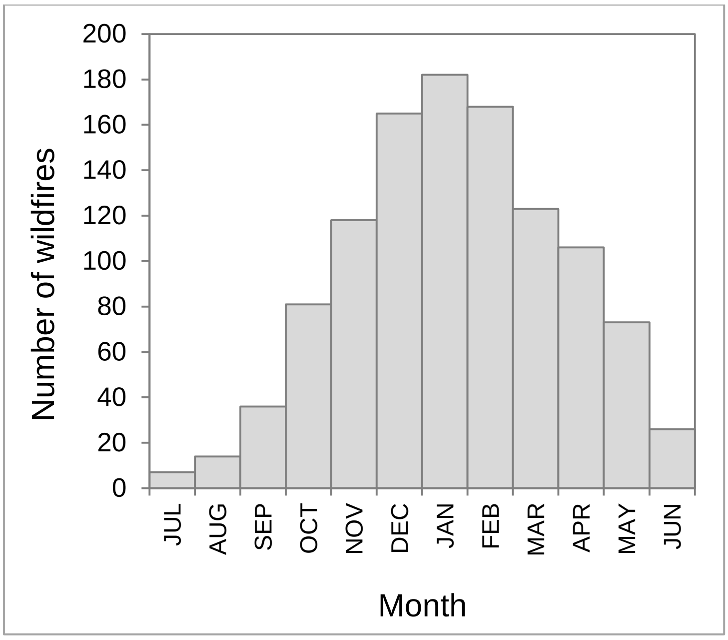

Figure 4.8. Number of wildfires per month between July 2003 and June 2013 (n=1,099)

4.2.4. Stopping rule

The AUSTRALIS simulator does not simulate suppression *per se*; that is, it does not simulate the deployment of fire-fighting forces to suppress a fire event, nor the results of their efforts on the rate of spread of the fire. Hence, if no stopping rule is specified, a simulated fire would

[12] To ensure that the selected fires simulated under extreme weather conditions for each fire season was the same for all the prescribed-burning rates, a seed was used in the processing code to specify the initial value of the suite of random numbers.

keep spreading until it naturally goes out: when all the edges of the fire reach areas where there is nothing to burn and the fire cannot spread any longer (assuming that there are no embers flying ahead of the fire) or the weather conditions change and extinguish the fire (e.g. rain). In order to simulate the effects of suppression on fire spread, we developed a stopping rule for the fires that depends on the time that suppression forces would take to reach the fire event and their capacity to suppress the fire depending on the intensity of the fire front. Since the capacity of fire-fighting forces to suppress the fire also depends on weather conditions (Price and Bradstock 2012; McCaw 2013), we created different rules for fires occurring on average weather conditions (usually resulting in fires ≤1500ha) and for fires occurring in more extreme weather conditions (usually resulting in fires >1500ha).

The stopping rule is composed of two parameters: (1) a delay parameter that tells the simulator to keep spreading the fire until the delay indicated has elapsed. This delay represents the approximate time that it would take fire-fighting forces to become aware of the fire event and to get to the location where the fire is burning. This delay is a function of distance from towns, since the more remote the fire, the longer it would take local fire brigades to become aware of the fire and deploy the resources to it. (2) After the delay has elapsed, a second parameter, fire-line intensity, determines the success of suppression efforts. For this parameter, a threshold is identified to indicate whether fire suppression is possible. Below the threshold, the simulator assumes that the fire can be suppressed and stops the spread of the fire from the cell that is burning with low intensity; above the threshold, the cell is assumed un-suppressible and may spread further into other unburnt cells.

Rule for stopping fires under average weather conditions

In order to create a stopping rule that would closely resemble the incidents in the South West forest region, we simulated all historical fires occurring in the region between 2003 and 2013 that burned under average weather conditions (resulting in fires ≤1500ha) and tested different stopping rules. The fires were simulated with similar weather conditions to those with which they actually burned. We found that the stopping rule that most closely fitted the total area burned and the distribution of fire size comprised:

1. A delay showing an exponential relation to the distance from towns. This relation is expressed in Equation (8)

   $$D = 600e^{0.00009\delta} \tag{8}$$

   where D is the delay in seconds, and δ is the distance from the closest town.

2. An intensity threshold of 650 kW/m

For the eucalypt forest of south-west WA it is usually considered that suppression is possible at a fire-line intensity ≤2,000 kW/m. Beyond a fire-line intensity of 2,000 kW/m, the capacity of suppression forces to stop the spread of the fire is significantly reduced (Gould *et al.* 2007). However, the intensity threshold that best reproduced the historical fires experienced in the case-study region between 2003 and 2013 is much lower (650 kW/m). It seems sensible though that in the simulations of historical data the intensity level would be much lower than in reality because it also aims to capture the fact that when fire-fighting resources get to the fire, they are not necessarily able to contain it if the fire perimeter exceeds the ground that the resources deployed can cover.

Rule for stopping fires under extreme weather conditions

For fires occurring under extreme weather conditions (resulting in fires >1500ha), in order to better fit the total area and fire size distribution of large historical fires occurring between 2003 and 2013 in the case-study region, it was necessary to increase the delay and reduce the intensity threshold. The delay and intensity threshold that more closely fitted the total area burned and the fire size distribution of the larger fires (>1500ha) in the dataset were:

1. A longer delay, ten times larger than for the smaller fires, as shown in Equation (9).

$$D = 6000e^{0.00009\delta} \tag{9}$$

2. A reduced intensity threshold of 200 kW/m

Appendix 1 shows the observed and simulated frequency distributions of wildfire size for the historical ignitions between 2003 and 2013.

4.2.5. Outputs of the simulator

The simulator generates a number of outputs, two of which are used in the economic model. Firstly, for all the ignitions simulated, the simulator indicates which cells were burned in the landscape, hence the total area burned is known for each wildfire. The second output used in the economic model is intensity. The simulator indicates for each cell burned in the landscape the intensity with which it burned. Since it has been shown in the literature that the level of damages depend largely on the level of intensity (Mercer *et al*. 2007), the information obtain from the simulator on the intensity of a cell is used to calculate the level of damage incurred by the cell.

4.3. Putting it all together: estimating the C+NVC curve

The C+NVC curve is the result of the sum of three curves: prescribed burning costs, suppression costs and net damages. Here, a brief description is given on how each of these curves was estimated.

4.3.1. Prescribed burning costs

Data containing prescribed-burning costs and area prescribed burned per year for the South West forest region between 2003 and 2013 was obtained from DPaW. This dataset included all expenditures associated with prescribed burning, such as employee salaries, payroll overheads, staff costs (e.g. travel and accommodation costs), vehicle use/hire and maintenance, aircraft use/hire and maintenance, fuel and oil, fire materials and equipment, communication equipment, and contracts paid to external organisations or salaries paid to staff from a different management region (i.e. Swan or Warren).Table 4.4 shows expenditures on prescribed burning for these 10 years and the average prescribed-burning costs per hectare estimated from this period. This average cost per hectare was used as the base cost per hectare and multiplied by a coefficient depending on the size of the treatment (coefficient μ_x) and its closeness to towns (coefficient β_i).

Table 4.4. Annual expenditure in prescribed burning and area prescribed burned in the South West forest region.

Year	Area prescribed burned (ha)	Prescribed burning expenditure (2013 AU$)	Costs per ha (2013 AU$)
2003-2004	56,083	3,224,183	57
2004-2005	70,291	2,940,409	42
2005-2006	57,537	3,016,870	52
2006-2007	46,020	3,286,727	71
2007-2008	52,189	3,229,351	62
2008-2009	56,252	4,115,029	73
2009-2010	57,699	3,621,646	63
2010-2011	48,567	3,147,440	65
2011-2012	24,677	2,647,463	107
2012-2013	6,026	1,968,646	327
Average			92

The data collected was examined in detail in order to extract per hectare costs that could vary with treatment size and distance from towns. In the DPaW dataset, three different treatment sizes were identified for which different per hectare costs could be estimated: treatments ≤500 hectares in size, treatments >500 ha and ≤1500 ha, and treatments >1500 ha. Although the exact costs per hectare for different treatment sizes was not readily available in DPaW's dataset, from the information contained in the dataset, a cost for each treatment size could be approximated. In this study, there are four categories of treatment size: ≤100 ha, >100 ha and ≤500 ha, >500 ha and ≤1,500 ha, and >1,500 ha. However, from the DPaW dataset it was not possible to separate the costs of treatments ≤100 ha. Consequently, the same coefficient was used for treatments ≤100 ha and for treatments >100 and ≤500 ha. Table 4.5 shows the estimated costs per hectare for different treatment sizes and the multiplication coefficients calculated.

Table 4.5. Average per hectare costs of prescribed burning and per hectare costs for different treatment sizes

Average cost/ha and treatment size	Cost per ha (AU$)	Multiplication coefficient
Average	92	
<500 ha	153	1.66
500-1500 ha	47	0.51
>1500 ha	34	0.37

With the data from DPaW, however, it was not possible to isolate the costs of different treatments depending on their distance from towns. As a result, in order to calculate the changes in prescribed-burning costs per hectare with different distance from towns, we used the data available on annual prescribed-burning expenses and hectares treated for each treatment size and each distance from towns. Four categories with different distances from towns were created: treatments located within 5km from the closest town, treatments located between 5 and 10 km from the closest town, treatments located between 10 and 20 km from the closest town, and treatments located beyond 20 km of the closest town. Coefficient values for the four distance categories were determined by minimising the difference between modelled and actual DPaW values for annual prescribed burning expenses using the Excel Solver add-in. The estimated multiplication coefficients for different distances from towns are shown in Table 4.6.

Table 4.6. Average per hectare costs of prescribed burning and per hectare costs for different distances from the closest town

Average cost/ha and distance from town	Cost per ha (AU$)	Multiplication coefficient
Average	92	
<5 km	465	5.1
5-10 km	119	1.3
10-20 km	119	1.3
>20 km	62	0.7

Size of treatments and distance from towns

The treatments for varying rates of prescribed burning (0%, 5%, 10%, 15% and 20% of DPaW-managed land in the South West forest region) were distributed across the landscape according to two different strategies. One strategy involved applying the treatments in a similar manner to DPaW's current application of prescribed burning in the South West forest region. For this strategy, additional resources are aimed at reducing fuels in areas where there has been a substantial accumulation of surface fuels because they have not been burned for long periods of time. The size of the treatments and their distance from towns are also similar to DPaW's current application. The percentage of area treated for each combination of treatment size and distance from the closest town corresponds to the percentages currently applied by DPaW. These proportions for different categories of size and distance were kept constant for all rates of prescribed burning, as shown in Table 4.7. In order to achieve a similar size distribution to DPaW's current application, an additional treatment size was created for treatments <100 ha.

Table 4.7. Hectares treated per category of treatment size and distance from towns (strategy similar to DPaW's current application of prescribed burning)

Distance from closest town (km)	Size of treatment (ha)	Percentage of area treated[a] (%)	Hectares treated for each prescribed burning rate			
			5%	10%	15%	20%
< 5	< 100	0.20	80	160	240	320
< 5	100-500	1.00	400	790	1,190	1,580
< 5	500-1500	1.90	750	1,500	2,250	3,000
< 5	> 1500	0.00	0	0	0	0
5-10	< 100	0.30	120	240	360	470
5-10	100-500	2.50	990	1,970	2,960	3,950
5-10	500-1500	5.70	2,250	4,500	6,750	9,010
5-10	> 1500	7.00	2,760	5,530	8,300	11,060
10-20	< 100	0.45	180	350	530	710
10-20	100-500	5.50	2,170	4,350	6,520	8,690
10-20	500-1500	8.55	3,380	6,750	10,130	13,510
10-20	> 1500	24.50	9,680	19,360	29,030	38,710
> 20	< 100	0.05	20	40	60	80
> 20	100-500	1.00	400	790	1,180	1,580
> 20	500-1500	2.85	1,120	2,250	3,380	4,500
> 20	> 1500	38.50	15,200	30,420	45,620	60,830
Total		100.00	39,500	79,000	118,500	158,000

[a] Average proportion of distance from towns and treatment size currently applied by DPaW

The second strategy involved intensifying prescribed burning around towns to protect high-value assets. In this case, additional resources are also directed towards reducing fuels in long-unburned areas, but the proportion of treatments applied closer to towns is significantly higher, as shown in Table 4.8.

Table 4.8. Hectares treated per category of treatment size and distance from towns (strategy intensifying prescribed burning close to towns)

Distance from closest town (km)	Size of treatment (ha)	Percentage of area treated (%)	Hectares treated for each prescribed burning rate			
			5%	10%	15%	20%
< 5	< 100	15.00	5,930	6,100[a]	6,100[a]	6,100[a]
< 5	100-500	7.50	2,960	5,930	7,310[a]	7,310[a]
< 5	500-1500	7.50	2,960	5,930	8,890	11,850
< 5	> 1500	0.00	0	0	0	0
5-10	< 100	6.00	2,370	7,240[b]	9,610[b]	12,030[b]
5-10	100-500	5.00	1,970	7,190[b]	10,930[b]	14,900[b]
5-10	500-1500	6.00	2,370	4,740	12,840[b]	22,060[b]
5-10	> 1500	3.00	1,190	2,370	3,550	4,740
10-20	< 100	5.00	1,970	3,950	5,930	7,900
10-20	100-500	7.00	2,770	5,530	8,300	11,060
10-20	500-1500	5.00	1,970	3,950	5,930	7,900
10-20	> 1500	3.00	1,190	2,370	3,560	4,740
> 20	< 100	1.50	590	1,190	1,780	2,370
> 20	100-500	4.50	1,780	3,550	5,330	7,110
> 20	500-1500	6.00	2,370	4,740	7,110	9,480
> 20	> 1500	18.00	7,110	14,220	21,330	28,450
Total		100.00	39,500	79,000	118,500	158,000

[a] The number of hectares prescribed burned do not correspond to the proportion indicated because the maximum amount of area that can be prescribed burned within this category has already been treated.

[b] The number of hectares prescribed burned do not correspond to the proportion indicated because it has been augmented to compensate for the hectares missing in other categories.

The costs of each prescribed-burning strategy for different prescribed-burning rates

From the spatial distribution of the treatments shown in Table 4.7 and Table 4.8, we calculated the following prescribed-burning costs for each of the strategies as shown in Table 4.9.

Table 4.9 Prescribed-burning costs for each strategy for different prescribed burning rates

Prescribed burning rate	Total prescribed burning costs	
	Current strategy (AU$)[a]	Strategy close to towns (AU$)[a]
5%	2,578,000	10,585,000
10%	5,145,000	17,769,000
15%	7,722,000	23,074,000
20%	10,290,000	27,399,000

[a] Rounded to the nearest thousand.

It is important to note that we only analysed prescribed burning treatments applied within DPaW-managed land; we did not simulate prescribed burning being applied in private land. Although prescribed burning does occur in private land in the South West forest region, it is only sporadically applied and its application does not follow a coordinated approach. In private land, the decision to prescribe burn rests with the land owner and DFES provides support to land owners that decide to treat areas of their property. But since these prescribed burns are not part of a land-management strategy and there are no data records on them, we decided not to include private land burns as part of the strategies tested. However, simulated fires can start on and/or spread to private land.

4.3.2. Suppression costs

Suppression costs per hectare were estimated using DPaW data on suppression costs and area burned in the South West forest region between 2003 and 2013 (see Table 4.10). Average suppression costs per hectare were estimated at AU$1170. In the literature examining suppression costs it has been shown that area burned by wildfires is an important determinant of total suppression costs (Calkin *et al*. 2005; Gebert *et al*. 2007; Donovan *et al*. 2008; Liang *et al*. 2008; Preisler *et al*. 2011), thus total suppression costs for the wildfires simulated were calculating by multiplying the average cost per hectare by the total area burned in each fire season.

Table 4.10. Annual expenditure in suppression and area burned by wildfires in the South West forest region

Year	Area burned by wildfires (ha)	Suppression expenditure (2013 AU$)	Cost/ha (2013 AU$)
2003/04	10,948	2,761,266	252
2004/05	1,405	4,717,340	3357
2005/06	12,983	4,076,628	314
2006/07	4,812	5,887,205	1223
2007/08	1,662	4,704,679	2830
2008/09	9,713	6,102,363	628
2009/10	12,662	5,883,350	465
2010/11	4,229	5,878,419	1390
2011/12	55,607	9,564,636	172
2012/13	5,618	5,987,391	1066
Average			1170

4.3.3. Damages

The analysis of damages occurs after all the simulations have been run, using the output from the simulator as an input in the calculations. To calculate the damage caused by the simulated wildfires, the resulting GIS layer with the area burned obtained from the simulator was overlaid on a land-use raster dataset with a 50-metre resolution. This dataset was downloaded from Australian Bureau of Agricultural and Resource Economics and Sciences, Department of Agriculture, Australian Government.[13] Different values were attributed to each land-use category.

For urban areas, the value per hectare was calculated using the National Exposure Information System (NEXIS), a modelling capability designed by Geoscience Australia that uses publicly available information, statistics, spatial data and survey data to model exposure information about residential, commercial and industrial buildings. The database produced by NEXIS contains up-to-date aggregated exposure data across Australia at the Local Government Area level (Dunford *et al.* 2014). Only exposure data for residential buildings was used because in the case-study region, it is often residential properties that are located at the edge of cities and towns and are surrounded by flammable vegetation. In the South West forest region, commercial and industrial buildings are often located well inside cities or towns, a considerable distance away from the edge of the city (where the WUI starts), and therefore have a lower probability of being destroyed by a fire because of the absence of vegetative flammable fuels. In the NEXIS database, the total value of residential buildings for a Local Government Area corresponds to the sum of replacement cost and content value for all residential buildings located in the Area, and the total value of commercial and industrial buildings corresponds to their structural value. The total value for residential buildings for each Local Government Area was divided by the number of hectares within the area categorised as urban in the land-use raster dataset. The most conservative value amongst the Local Government Areas located within the South West forest region was used as the value per hectare for urban areas.

Wildfires affect agriculture through damage to crops, pastures, fences, livestock, and farming structures and equipment. In the south-west of WA, most crops (wheat, barley, canola, and cereal for hay) are harvested just before the start of the fire season, thus the damages to

[13] Product detail: Land Use of Australia, Version 4, 2005/2006 (September 2010 release). Available from http://data.daff.gov.au/anrdl/metadata_files/pa_luav4g9abl07811a00.xml

agricultural crops can be assumed to be zero. In addition, livestock losses are rare. However, the restoration of damaged fences can represent a significant cost to agricultural producers. For instance, about 15,900 kilometres of fencing were destroyed during the Ash Wednesday fires in 1983 (Healey *et al.* 1985). Bureau of Transport Economics (2001) made an estimate of the cost of repairing damaged fences of AU$5,000 per kilometre (AU$6,935 per kilometre in 2013 dollars). Using this cost of fence restoration, the estimated total cost of fence restoration for Ash Wednesday was AU$110,266,500. This amount, divided by the total area burned by the fires of Ash Wednesday was (418,000 ha) gives an average per-hectare cost of fence restoration of AU$264.

To estimate the cost per hectare of farming structures and equipment, the information available on the 2009 Black Saturday wildfires was used. These fires burned more than 450,000 ha and destroyed 2030 houses and 2000 outbuildings and other structures. The Insurance Council of Australia reported claims of approximately AU$1,008 million (AU$1,086 in 2013 dollars) for property or contents (Teague *et al.* 2010). Assuming that the replacement value of an outbuilding and its contents is double the amount of the replacement value of a house and its contents, the total losses for farming structures for the Black Saturday fires are estimated at AU$360 million (in 2013 dollars), which, divided by the number of hectares burned by the fires, corresponds to a cost per-hectare of AU$800. The sum of per-hectare cost of fence restoration and per-hectare value of farming structures and content was used as the replacement value for agricultural land used for grazing and cropping.

In 2004, the cost to bring a vineyard into production in the south-west of WA ranged between AU$40,000 and AU$60,000 (AU$ 51,238 and AU$ 76,856 in 2013 dollars) per hectare (AHA Viticulture 2006). This cost was used as the replacement cost per hectare for agricultural land containing vine fruits. Table 4.11 shows the different categories, the replacement value per hectare of each category, and the sources used to calculate the different values.

Table 4.11. Land-use categories and replacement values (in 2013 dollars)

Land-use category	Value per hectare (AU$/ha)	Value per pixel (AU$/50x50m)	Source
Urban areas	1,156,015	289,004	Dunford *et al*. (2014)
Conservation areas and State forests	1,192	298	Gibson and Pannell (2014)
Plantation forestry	9,080	2,270	Gibson and Pannell (2014)
Agricultural (grazing, cropping)	1,064	266	Bureau of Transport Economics (2001); Teague *et al.* (2010)
Agricultural (vine fruits)	51,238	12,810	AHA Viticulture (2006)
Mining	49,361	12,340	Dunford *et al*. (2014)

In order to take into account the differences in fire-line intensity across the areas burned, our model calculates the damage to each cell in the landscape depending on the level of intensity with which the cell is burned. It is generally agreed that the level of damage is strongly correlated with the intensity of a fire (Mercer *et al*. 2007). However, literature that discusses this assumption is remarkably rare. Using the small amount of information available in the fire behaviour and fire impacts literature, we estimated a relationship between the level of intensity and the level of damages. Here follows an explanation of how this relationship was estimated. The final relationship between intensity level and the level of damage used is presented at the end of this sub-section in Table 4.13.

First, we reviewed literature that contained information on the level of damage to eucalypt forests caused by fires of different intensity levels. The genus *Eucalyptus*, which is the most commonly found genus in the south-west forests of WA (McCaw and Hanstrum 2003; Burrows and McCaw 2013; McCaw 2013), is particularly resistant to fires of low to moderate intensity, in terms of the capacity of individual trees to survive the fires. Some species in the genus are also resistant to damage from high-intensity crown fires, and only a few species are likely to be completely killed by high-intensity fires (McArthur 1962; Chatto and Tolhurst 2004). Table 4.12 shows the expected damage on eucalypt forests caused by different fire intensities, and the approximate relationship between intensity level and the level of damages summarised from the literature.

Table 4.12. Summary of the literature on the level of damage for eucalypt forests from fires of different intensities[a]

Fire intensity (kW/m)	Intensity level	Damage to eucalypt forests[b]	Level of damage (%)
< 500	Low	Flame height < 1.5m. Little tree canopy scorch, understorey species partially damaged.	1
500 – 1,700	Moderate	Flame height between 1.5 and 6m. Defoliation and death of most understorey species, damage to branches of overstorey.	10
1,700 – 3,500	Moderate	Flame height between 6 and 15 metres. Crown fire in low forest types, scorch of complete crown in most forests.	30 – 50
3,500 – 7,000	High	Flame height between 6 and 15 metres. Crown fire in low forest types, scorch of complete crown in most forests with tree canopies defoliated over large areas.	50 – 70
7,000 – 70,000	Very High	Flame height > 15m. Crown fire in most forest types, with firestorm conditions. Death of almost all above-ground foliage of most species.	> 70

[a] This is a modified version of a table in Chatto and Tulhurst (2004).
[b] Based on data examined by Cheney (1981) and Christensen *et al.* (1981)

To estimate the damage caused to houses (urban areas category) we used the information reported by Wilson and Ferguson (1984) on the number of houses affected by the fire at Mount Macedon on 16 February 1983 in Victoria.[14] The houses affected by the Mount Macedon fire were exposed to intensities ranging from 500 to 60,000 kW/m. Of the 450 houses surveyed, 234 were destroyed. Although there is no indication in Wilson and Ferguson's study about the fire intensity to which the destroyed houses were exposed, it is known that about 45 houses were exposed to crown fire in nearby forest canopy (potentially with flame height >15m, which usually occurs at intensities >7,000 kW/m, see Table 4.12), about 225 houses were exposed to surface fire that was intense enough to scorch the surrounding trees (which might occur at intensities ranging between 3,500 and 7,000 kW/m, see Table 4.12) and 180 houses were exposed to less intense surface fire that produced large amounts of airborne embers due to strong winds (which might occur at moderate intensities ranging between 1,700 and 3,500 kW/m, see Table 4.12).

[14] The Mount Macedon fire was part of a series of fires that burned in Victoria and South Australia on 16 February 1983, known in Australia as the Ash Wednesday bushfires. Ash Wednesday is the second deadliest wildfire event in Australian history after the Black Saturday wildfires of 2009.

Knowing the total number of houses destroyed (234) and the number of houses exposed to different intensities, it is possible to infer an approximate estimate of houses destroyed for different levels of intensity. It is assumed that about 15-20% of the houses destroyed were amongst those exposed to crown fire, about 50-60% of the houses destroyed were amongst those exposed to intense surface fire, and 20-25% of the houses destroyed were amongst those exposed to moderate surface fires. This would mean that about 90% of houses exposed to crown fire (intensity between 7,000 – 70,000 kW/m) get destroyed, about 60% of houses exposed to intense surface fire (intensity between 3,500 – 7,000 kW/m) get destroyed, and about 30% of houses exposed to moderate surface fire (intensity between 1,700 – 3,500 kW/m) get destroyed. With these estimates, an approximate relationship between intensity level and the level of damage was produced, as shown in Table 4.13.[15] The analysis of the information available in other reports investigating house losses in fires in the south-west of WA led to the estimation of comparable levels of damage. Table 4.14 shows a summary of house losses as indicated in these reports.

Table 4.13. Level of damage as a function of intensity for urban areas

Intensity level (kW/m)	Level of damage (%)
≤ 1,700	10
> 1,700 and ≤ 5,000	40
> 5,000 and ≤ 10,000	60
> 10,000 and ≤ 20,000	70
> 20,000 and ≤ 50,000	90
> 50,000	95

Table 4.14. House losses and fire-line intensity for three different fires in south-west WA

Intensity level (kW/m)	Houses destroyed	Houses damaged	Houses surveyed	Fire investigated
6,900 – 11,700	72 (51%)	37 (26%)	141	Roleystone-Kelmscott fire, February 2011 (Smith 2011)
1,500 – 3,500	39 (59%)	26 (39%)	66	Margaret River fire, November 2011 (Smith 2012)
500 – 6,500	48 (60%)	7 (9%)	80	Parkerville-Mt Helena and Stoneville fire, January 2014 (Smith 2014)

[15] The expression that best fitted ($R^2 = 0.93$) these data points (2600, 30; 5250, 60; 38500, 90) was $\omega = 20.683\ln(I) - 126.07$, where ω is the level of damage and I is the intensity level in kW/m.

The estimated level of damage in urban areas as a function of the level of intensity is very similar to the estimated level of damage for eucalypt forests. Thus, it is possible to use the relation found in urban areas for eucalypt forests. Due to a lack of information for other categories, the relation shown in Table 4.13 was used to calculate fire damages as a function of intensity for all categories.

4.3.4. The sum of management costs and damages: the C+NVC curve

Suppression costs and damages were calculated for 220 fire seasons for each prescribed-burning rate. For each prescribed-burning rate, the average suppression costs of the 220 fire seasons and the average damages were calculated. These averages plus the costs of prescribed-burning for each prescribed burning rate correspond to each number in the curve of the C+NVC model. The prescribed burning rate that minimises the sum of prescribed burning costs, average suppression costs and average damages indicates the strategy that maximises benefits to society.

Chapter 5 presents the results of a one-year analysis using the modified C+NVC model described in this Chapter.

Chapter 5.

SHORT-TERM ANALYSIS OF PRESCRIBED BURNING STRATEGIES USING THE C+NVC MODEL

5.1. Introduction

In the literature investigating the economics of fire management, one area of study where more research is needed is in the development of stochastic, spatially explicit models (Chapter 2). In particular, there is a need for models that take into account two important spatial attributes: (1) the differences in value of different types of assets, such as conservation areas, agricultural equipment, buildings and infrastructure, etc., and (2) the variation in prescribed-burning costs with the size and the location of the treatments. Previous models have used common values for all units of area in the landscapes (e.g. Mercer *et al.* 2007; Mercer *et al.* 2008; Butry *et al.* 2010) without taking into account the variation in value between units of area containing different types of assets (Chapter 2). This is of concern; recognition of the spatial heterogeneity of values will improve the prioritisation of investments and the optimisation of the spatial distribution of the treatments. Similarly, the costs of prescribed burning vary spatially. But although it has been demonstrated that the size of a treatment and its proximity to the wildland-urban interface (WUI) can significantly influence per- hectare costs of prescribed burning (Chapter 2), only changes with treatment size have been incorporated in previous studies (e.g. Mercer *et al*. 2007). These gaps in research are addressed in this study.

Chapter 2 also identified three areas where theoretical and empirical models of fire management economics need to be more explicit: (1) the management options that they evaluate, (2) the modelling timeframe to be used, and (3) the initial conditions to which changes are applied (i.e. the counterfactual). First, fire management options are in some cases poorly defined. Some articles use the terms “fire management options” or “pre-suppression activities” without further definition. However, these terms are too vague as they encompass a variety of activities that have different effects on wildfire risk mitigation. Often authors refer to investments in pre-suppression or investments in fire management without specifying what each investment includes. Furthermore, the definition of a term may vary from one article to another. Some studies define pre-suppression as investments in initial attack (e.g. Mills 1979; Mills and Bratten 1982; Gonzalez-Caban *et al*. 1986; Donovan *et al*. 1999), while others define it as investments in prevention strategies such as prescribed burning and/or fire prevention education (e.g. Mercer *et al*. 2007; Butry *et al*. 2010). For fire economics frameworks to be useful and interpretable, these terms need to be more clearly defined. Second, the modelling timeframe is rarely specified in these frameworks. As such, there is no indication about what difference it would make to use different timeframes, e.g. the short term vs. the long term. This in turn raises additional questions: (1) what exactly are the short term and the long term in fire management? and (2) what is an appropriate timeframe for the economic evaluation of a particular fire management option and why? Finally, the initial fuel load in the landscape may affect the result, and thus, initial conditions need to be stated clearly.

In this study, all the elements mentioned above have been addressed and clearly specified. First, the management option evaluated is well defined. This economic analysis only evaluates alternative investments in prescribed burning, which is only one option amongst several prevention or pre-suppression options available.[16] Here, prescribed-burning costs include employee salaries, payroll overheads, staff costs, vehicle and aircraft use and maintenance, fire materials and equipment, communication equipment, and contracts paid (see Section 4.3.1). The total amount invested in prescribed burning is determined by the amount of area treated, and the size and the location of the treated areas.

Second, the timeframe of the analysis is well specified. Two different timeframes have been used: a short term and a long term. The short term corresponds to one fire season, from winter to winter in the southern hemisphere, i.e. 12 months between 1st July in year t and 30th June in year $t+1$. The long term in this study corresponds to at least 10 years. In the long-term strategies that are evaluated here, the prescribed burning strategy is sustained for the whole 10-year period (although the areas prescribed burned vary each year). This Chapter presents the results of the short-term analysis. The details and the results of the long-term analysis are presented in the following Chapter (Chapter 6). Although there is evidence that the effects of prescribed burning may last for several years (Fernandes and Botelho 2003), it was decided to explore the short-term effects of investments in prescribed burning because some fire management decisions are made to create changes in the short term and budgetary planning for fire management programs are done annually. In some cases, decision makers

[16] Other prevention or pre-suppression management options may include mechanical fuel reduction treatments, chemical treatments, fire breaks, and fire education programs.

tend to adopt a short-term perspective for the resource allocation for different government activities, particularly after large wildfire events. For instance, political decisions regarding prescribed burning in the south-west of Western Australia (WA) tend to be affected by recent fire events and the performance of prescribed burning (Burrows and McCaw 2013).[17] These short-term decisions do not always align with the long-term strategic targets for the protection of humans, property and the environment. Thus, it is important to evaluate the effects of alternative investments in prescribed burning in the short term and in the long term and compare the results of both approaches. In the past, the C+NVC model was used by the US Forest Service to set annual budget requests to Congress for operational activities (mostly for fire-fighting activities). Benefit:cost analyses are not yet used in WA to set budget requests for fire management activities. But for now, we aim to understand the consequences of modelling the problem in this way for independent annual budget requests, while extending the approach to include prescribed-burning.

Finally, the initial conditions in the case-study area in terms of fuel levels are also clearly specified in this study. For the short-term analysis, the base fuel layer was the landscape as it was in June 2012 in the South West forest region. The fuels of this layer were modified with the application of prescribed burning. And in order to determine how much these initial fuel conditions affect the results, they were evaluated in a sensitivity analysis.

Using the AUSTRALIS wildfire simulator, five different rates of prescribed burning were tested (0, 5, 10, 15 and 20% of lands under DPaW management). For the short-term analysis, two different strategies with different locations for the treatments were analysed: (1) the "current strategy", consisting of prescribed burning treatments of a similar size and distance from towns as they are currently applied by DPaW in the case-study area, and (2) the "close to towns strategy", for which the treatments are intensified around towns and reduced further from towns. With the model developed in Chapter 3, the costs of prescribed burning, the average costs of suppression and the average damages for the different rates of prescribed burning were estimated for the two strategies. 220 fire seasons with an average of 110 fire events were simulated for each prescribed-burning rate. This gives a total of 121,000 fires, that is, 5 prescribed-burning rates with 24,200 fires per prescribed-burning rate (i.e. 0, 5, 10, 15 and

[17] Prescribed burning may be increased following a severe wildfire event, as it happened after the fires of 1961 that destroyed the town of Dwellingup, located about 100 km south of Perth (Burrows and McCaw 2013); or may be significantly reduced following an escaped prescribed burn, as it happened after the escaped prescribed burn of November 2011 next to the town of Margaret River that instigated a Special Inquiry (DEC 2012; Keelty 2012).

20%). The area where these fires were simulated, the South West forest region, is located in the south-west of Western Australia, about 140 km south of Perth. The region has a mixture of forests, plantations, agricultural land, and important tourist towns and destinations, for which there are multiple and sometimes conflicting fire-management objectives.

The Chapter is organised as follows. Section 5.2 first presents the results for different prescribed-burning rates for the current strategy, then for the close-to-towns strategy, and the last part of the Section compares the results of both strategies. Section 5.3 presents a sensitivity analysis of some of the key variables in the model. Section 5.4 discusses the implications of the main findings for fire management through prescribed burning in the South West forest region and for the economic analysis of prescribed burning.

5.2. Results

5.2.1. Current strategy

First, the prescribed burning treatments were distributed in the landscape using DPaW's current strategy. This means that the treatments are of a similar size and distance from towns to those that are currently implemented by DPaW. Under the current strategy, an average of 70% of the total area treated annually by DPaW in the South West forest region is burned in patches larger than 1,500 hectares. About half of these large patches are located more than 20km away from towns, and about 1/3 are located between 10 and 20km from towns. Only a few large treatments >1,500 hectares are located between 5 and 10km from towns and none are located any closer to towns. The remaining 30% of the total area treated annually in the South West forest region is done in patches of <1,500 hectares at different distances from towns. The table presented in Chapter 3 showing the number of hectares treated for each combination of treatment size and distance from towns is reproduced here again for the convenience of the reader (see Table 5.1). These percentages of area treated for different sizes and distances from towns are the realistic combinations applied in the landscape when DPaW prescribe burns 6% of the area it manages within the South West forest region (these percentages were obtained from historical data). It is assumed in this study that these percentages remain equal for other rates of prescribed burning (i.e. 5, 10, 15 and 20% of DPaW-managed land). This distribution of the treatments is what is here referred to as the "current strategy". The number of patches treated in the landscape with the current strategy for the simulations were 102, 204, 295, and 379 for the 5, 10, 15 and 20% prescribed-burning rates respectively. The average size of a treatment (i.e. a patch prescribed burned) is 388 hectares and the average cost per hectares is $65.

If the current strategy is applied, the prescribed-burning rate that minimises the sum of management costs and damages in the South West forest region in the short term is 15% (see Figure 5.1 and Table 5.2). This corresponds to applying the treatment to approximately 118,500 hectares of DPaW-managed land in the South West forest region and investing AU$7,722,000. This rate is similar to the annual rate of prescribed burning that was applied in the area in the 1960s and 1970s (see Chapter 3).

Table 5.1. Hectares treated per category of treatment size and distance from towns (strategy similar to DPaW's current application of prescribed burning)

Distance from closest town (km)	Size of treatment (ha)	Percentage of area treated[a] (%)	Hectares treated for each prescribed-burning rate			
			5%	10%	15%	20%
< 5	< 100	0.20	80	160	240	320
< 5	100-500	1.00	400	790	1,190	1,580
< 5	500-1500	1.90	750	1,500	2,250	3,000
< 5	> 1500	0.00	0	0	0	0
5-10	< 100	0.30	120	240	360	470
5-10	100-500	2.50	990	1,970	2,960	3,950
5-10	500-1500	5.70	2,250	4,500	6,750	9,010
5-10	> 1500	7.00	2,760	5,530	8,300	11,060
10-20	< 100	0.45	180	350	530	710
10-20	100-500	5.50	2,170	4,350	6,520	8,690
10-20	500-1500	8.55	3,380	6,750	10,130	13,510
10-20	> 1500	24.50	9,680	19,360	29,030	38,710
> 20	< 100	0.05	20	40	60	80
> 20	100-500	1.00	400	790	1,180	1,580
> 20	500-1500	2.85	1,120	2,250	3,380	4,500
> 20	> 1500	38.50	15,200	30,420	45,620	60,830
Total		100.00	39,500	79,000	118,500	158,000

[a] Average proportion of distance from towns and treatment size currently applied by DPaW

The benefits correspond to the reduction in suppression costs and damages compared to a no-prevention strategy (i.e. 0% prescribed burning). As shown in Table 5.2, additional investments in prescribed burning always result in reduced suppression costs and damages, but the reduction varies for different levels of prescribed burning. The prescribed-burning rate that minimises the C+NVC is the rate that maximises the net benefits to society (Table 5.2).

In terms of prescribed-burning investment in the South West forest region, these results indicate that this investment should be substantially increased. Between 2003 and 2013, DPaW prescribed burned an average of 6% per year of the land it manages in the South West forest region. This corresponds to an average of about 47,500 ha prescribed burned per year and an average annual expenditure of AU$3.1 million. If DPaW were to apply the optimal rate for prescribed burning (15%) using the current strategy as indicated by the results above, DPaW would require significant additional resources. To achieve a rate of 15%, both the area prescribed burned per year and the investment in prescribed burning would need to be increased by 150%.

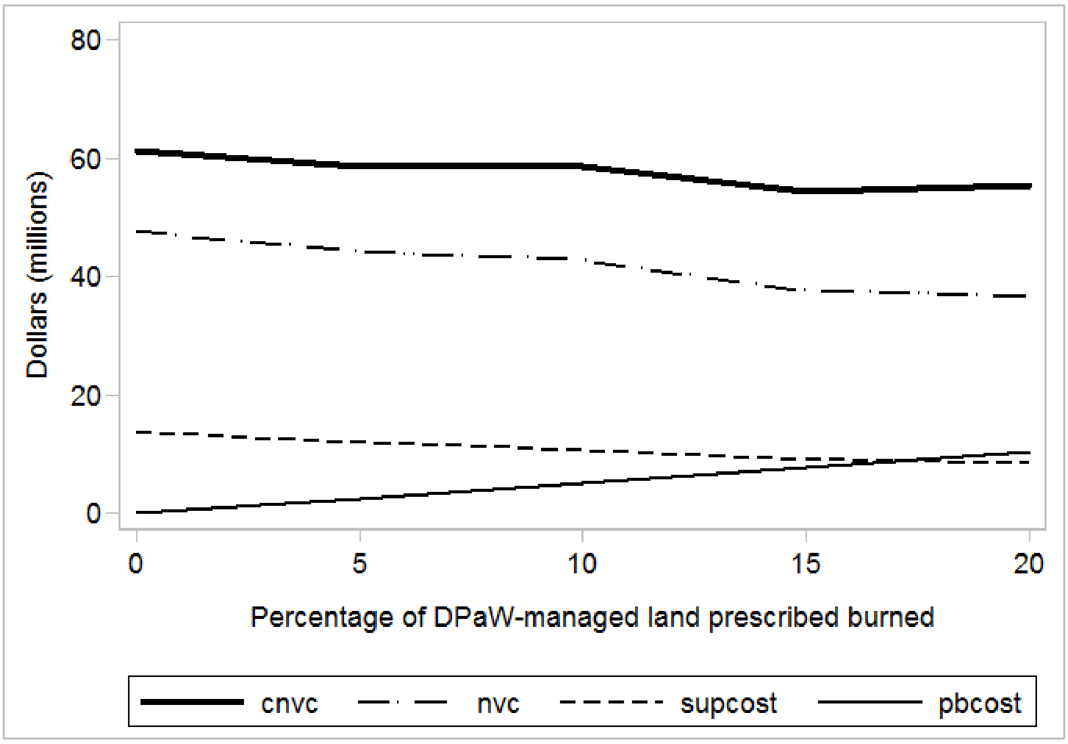

Figure 5.1. Cost plus net value change for different prescribed-burning rates applying the current strategy (short-term analysis)

Table 5.2. Results for different prescribed-burning rates applying the current strategy (millions of AU$, for a one-year time frame)

Rate	Prescribed-burning costs	Suppression costs	Damages	C+NVC	Suppression costs plus damages	Benefits (compared to 0%)[a]	Net benefits (compared to 0%)
0%	0.00	13.71	47.60	61.32	61.32		
5%	2.58	11.92	44.13	58.63	56.05	5.27	2.69
10%	5.15	10.63	42.76	58.54	53.39	7.93	2.78
15%	7.72	9.06	37.50	54.28	46.56	14.76	7.03
20%	10.29	8.49	36.69	55.47	45.18	16.13	5.84

[a] The benefits correspond to the reduction in suppression costs and damages compared to a no-intervention scenario (i.e. 0% prescribed burning)

Short-term investments in prescribed-burning programs in the South West forest region appear to be a favourable investment. Compared to a no-prevention strategy (0% prescribed burning), short-term investments in prescribed burning in the region generate between AU$2.7 and AU$7.0 million in net benefits, which corresponds to a benefit of AU$1.5 to AU$2 per dollar invested. Although the highest net benefits arise from a 15% rate of prescribed burning (Table 5.2), overall the C+NVC curve looks relatively flat (see Figure 5.1), so the superiority of 15% over other prescribed-burning rates is not marked. Up to 15% the increase in prescribed burning costs is approximately offset by the increase in benefits. Beyond 15% the increase in prescribed burning costs is larger than the reduction in damages and suppression costs, but only by a small amount. In terms of decision making for fire management, any prescribed-burning rate could be applied in the short term and yield a comparatively similar result.

Nevertheless, the proportion of management costs and the proportion of damages are very different for different levels of prescribed burning. These proportions are more clearly seen in Figure 5.2. At low levels of prescribed burning, the vast majority of the costs arise from the damages sustained and the amount of money that has to be spent in suppression. When 10% or less of DPaW's managed land is prescribed burned, damages represent about 3/4 of the C+NVC, and management costs (suppression costs and prescribed burning costs) account for about 1/4 of the C+NVC. But at higher levels of prescribed burning (>10%), expenditure in prescribed burning and in suppression costs are approximately the same, and the proportion of damages is reduced. Damages then account for 2/3 of the C+NVC, while management costs correspond to 1/3 of the total C+NVC. In all cases, damages are always much larger than management costs, being twice or three times larger than suppression and prescribed burning expenditures together. The different proportions of damages and management costs are important to fire managers, even for short-term investments.

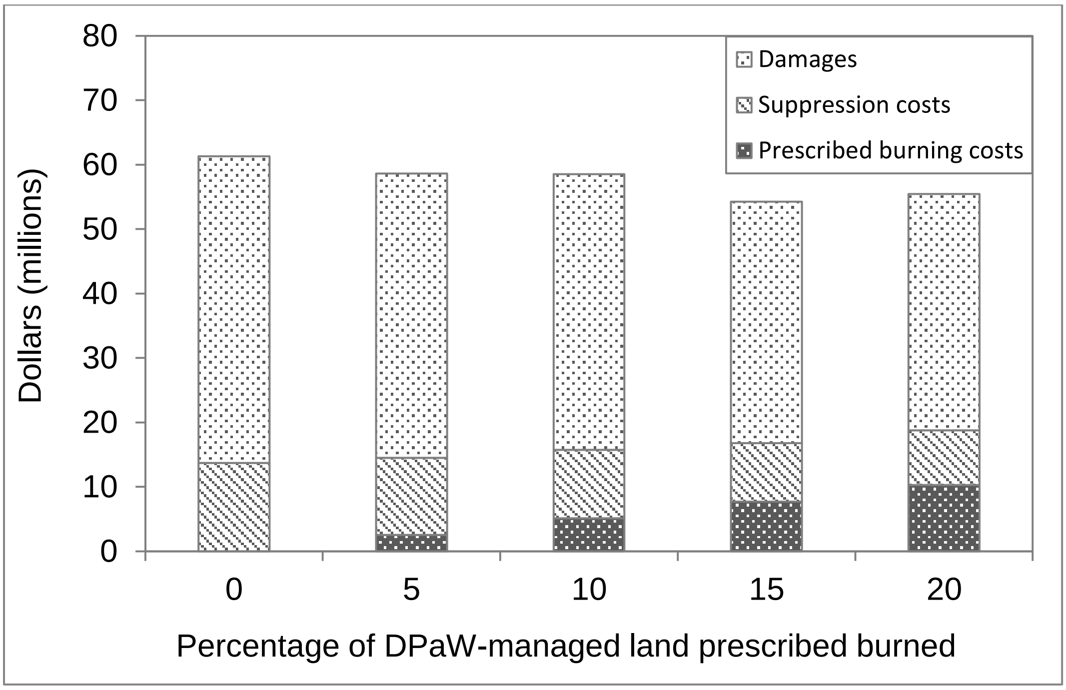

Figure 5.2. Damages, suppression costs and prescribed burning costs for each prescribed-burning rate using the current strategy

From a fire-management perspective, although there is not a substantial difference in the C+NVC between 0% and 20% DPaW-managed land prescribed burned in the short term, some choices may not be politically feasible. In the analysis all dollar values are assumed to be fungible, but this may not be the case since the gains and losses are experienced by different stakeholders. If a very low level of prescribed burning is applied in the short term, society would have to be willing to accept high levels of damages and suppression expenditures. However, society is generally more willing to tolerate high management costs than high levels of damages. Indeed, people are generally willing to pay significant amounts of money for reducing fire risk even when they are insured against property loss. This is because there are substantial non-market losses experienced when a house is destroyed by fire (Chapter 2). Conversely, high levels of prescribed burning may cause smoke issues and increase the chances of escaped fires, which in turn may generate resistance from the community towards the application of prescribed burning. In any case, to achieve a substantial reduction in damages, the amount of area that needs to be treated in the short term is remarkably large, which results in a considerable part of the landscape being subjected to fire.

In the short term, the application of prescribed burning with the current strategy actually increases the total amount of area in the landscape that is subjected to fire of any kind (prescribed or wildfire). In fact, the total area that experiences fire is much smaller if no prescribed burning is applied at the start of the fire season (Figure 5.3). When the optimal rate of prescribed burning is applied with the current strategy, a total of 126,241 hectares are burned by both wildfires and prescribed burns. Results from the simulations show that in the short term large amounts of area need to be prescribed burned in order to reduce wildfire area. On average, to reduce wildfire area by one hectare in the South West forest region, between 26 and 35 hectares need to be prescribed burned before the start of the fire season.

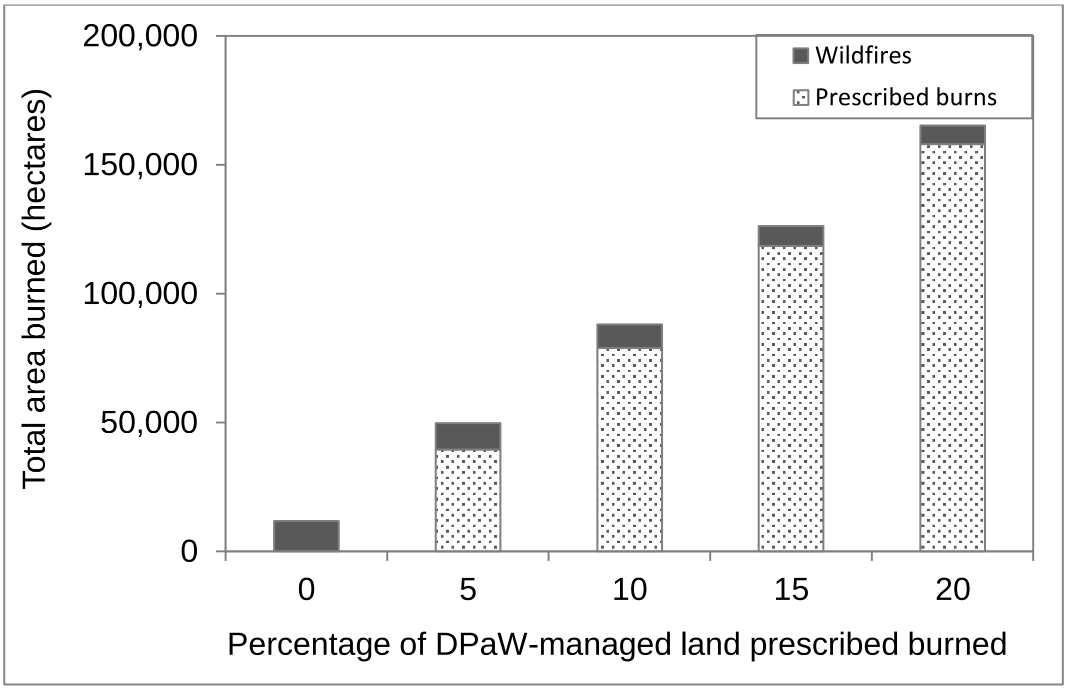

Figure 5.3. Total area subjected to fire of any kind (prescribed or wildfire)

It is important to remember, however, that for this short-term analysis, the fuel levels of June 2012 were used as the starting point. The treatments were applied over the oldest fuels according to the prescribed-burning rate applied (5, 10, 15 or 20% of DPaW-managed land) and all other non-treated fuels were left at the level they were in June 2012 in the region. The fires were then simulated over a landscape containing fuels burned in 2012 or earlier. This means that for the no-prevention strategy (0% prescribed burning), the fuels of June 2012 were left unchanged. Evidently, in June 2012 the South West forest region had a history of prescribed burning and wildfires, thus the landscape had a mosaic of patches of varying fuel ages throughout. Where there is a record of fuel age, the time since last burn varied between 0 and 80 years.[18] The prescribed burning regime carried out before 2012 and the historical

[18] In the fuel-load dataset of the simulator (which combines the datasets of the Department of Parks and Wildlife–public land–and the Department of Fire and Emergency services–private land), 60% of the South West Forest region has a record of fuel age. Nearly all of the forested areas, which are the areas

wildfires in the area have an influence on the behaviour of the fires simulated. Hence, in the short term the results also depend on what has happened before in the area and the level of fuels in the year the simulations were performed. The effect that initial fuel levels have on a shot-term analysis is explored in the sensitivity analysis (Section 5.3).

Another important question for fire managers is what assets are most frequently damaged by the fires. In other words, what do wildfires usually burn? The results of the 121,000 fires simulated for this study show that in the South West forest region, the type of assets that are most frequently burned are Conservation areas and State forests (around 70% of the average area burned by wildfires) (Figure 5.4), compared to 4% in plantations, 25% in agricultural land, and 0.4% in urban areas. But what is most frequently burned in the landscape, is not necessarily what generates the greatest damages.

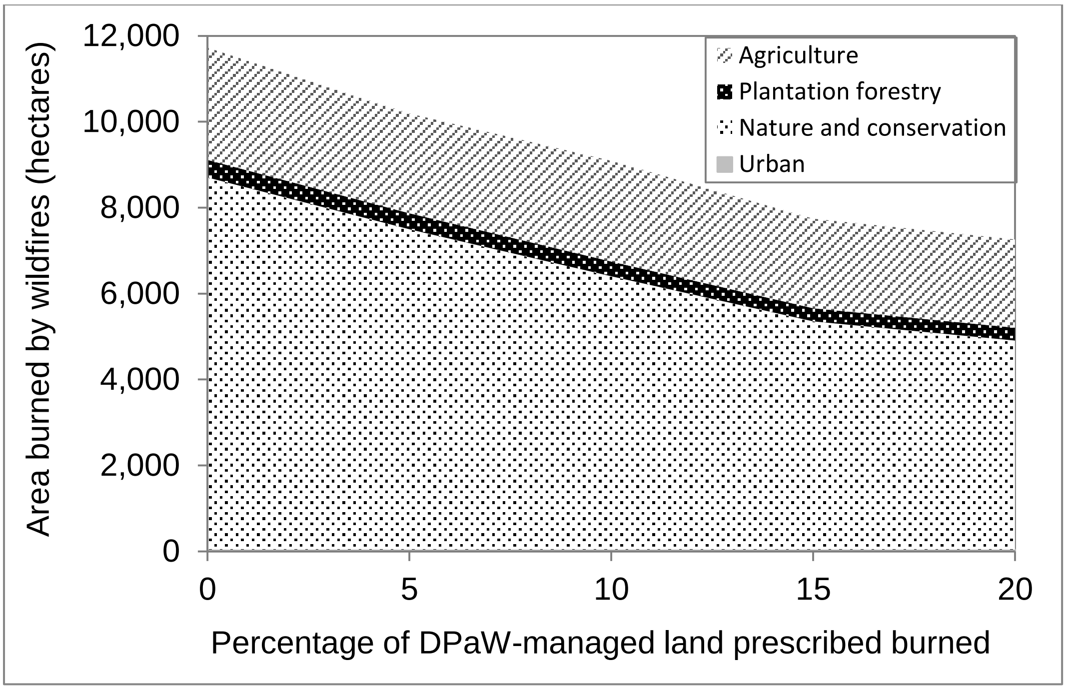

Figure 5.4. Area burned by wildfires per land-use type for different prescribed-burning rates

Because different land uses have different values per hectare, the level of damages (in dollars) vary greatly depending on where the fires burn. For instance, the value of a hectare in urban areas in the South West forest region was estimated at AU$1,160,000 (Chapter 4), whereas the value of a hectare located in any other type of land-use is <AU$51,000. For the purpose of

relevant for prescribed burning, are recorded. Most of the areas for which a record is missing correspond to grassland and cropping areas located in private land where there is not a coordinated approach to prescribed burning. For forested areas where no record of fuel age exists, a high level of fuels is assumed (20 years since last burn).

comparison, the value of one single hectare in an urban area is equivalent to the value of 970 hectares of forests. As a result, although the amount of urban areas burned is relatively small (0.3% to 0.5% of the total area burned by wildfires), the largest proportion of damages in dollars arises from the destruction of assets in urban areas (see Figure 5.5).

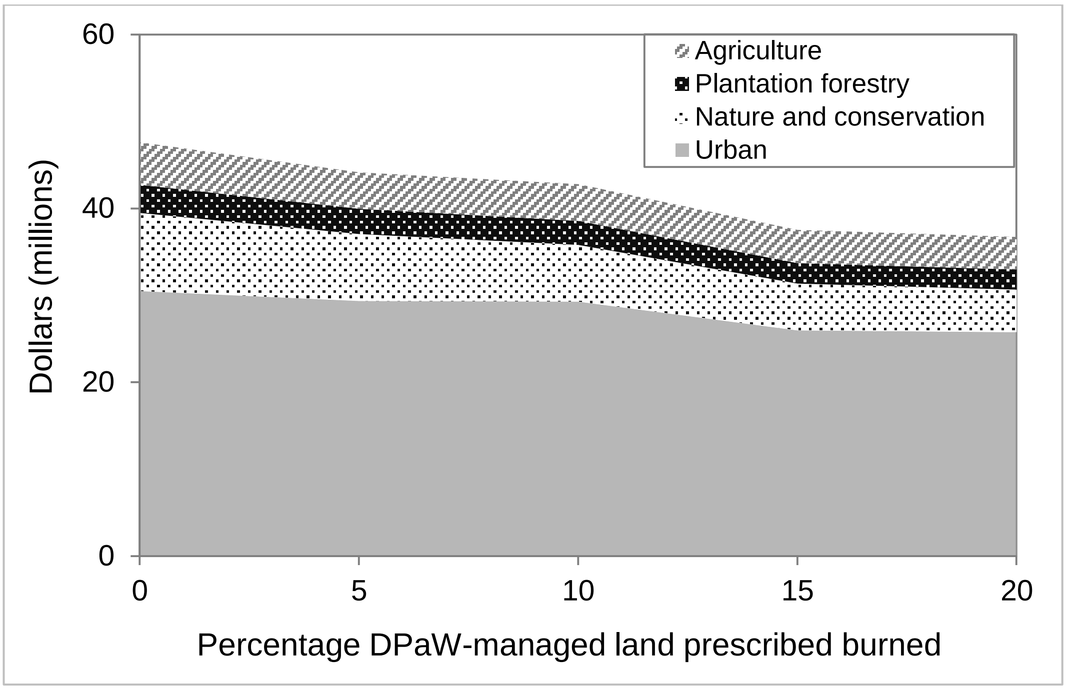

Figure 5.5. Wildfire damages per land-use type for different prescribed-burning rates

Thus, prescribed burning strategies that reduce wildfire area within the WUI, where there is a concentration of high value human assets, may result in substantial reductions in damages. However, in the short term with the current strategy, the reduction in area burned occurs primarily in conservation areas and State forests (Figure 5.4) and the reduction in wildfire area in the WUI is minor. Because wildfire area is significantly reduced in conservation areas and State forests with the current strategy, the reduction in damages in those areas is proportionally large. If the optimal strategy is applied (i.e. 15% of DPaW-managed land is prescribed burned), damages in conservation areas and State forests would be reduced on average by 40% compared to a no-prevention scenario (0% prescribed burning). However, the reduction in total wildfire damages for all land-use categories combined is only 10%. This is explained by the fact that conservation areas and State forests have a considerably smaller value per hectare compared to urban areas. Thus, despite the substantial reduction in wildfire damages in conservation areas and State forests areas with increases in prescribed burning, the total reduction in wildfire damages for all land-use categories combined is relatively small in the short term.

Because of the differences in value between the different land-use categories, prescribed burning generates different levels of benefits for each land-use type in terms of damages avoided. For every dollar invested in prescribed burning using the current strategy, on average a damage of AU$0.43 is avoided in urban areas, of AU$0.46 in conservation areas and State forests , AU$0.16 in agricultural areas, and AU$0.11 in plantations. Thus the land-use categories that benefit the most from prescribed burning under the current strategy and the scenarios here tested are the WUI and conservation areas.

But could more damages be avoided in the WUI by placing the treatments closer to the WUI? What if the spatial arrangement of the prescribed burns was modified and intensified around the WUI to protect high value human assets? Would there be a difference in the results in the short term? And how substantial is the difference, if any? These questions are explored in the following section.

5.2.2. Close-to-towns strategy

In order to evaluate how a different arrangement of the treatments would change the results, a second strategy was tested. In this strategy, hereafter referred to as the “close to towns” strategy, the majority of the treatments are located close to or around the WUI. The same prescribed-burning rates were applied (i.e. 0, 5, 10, 15 and 20% of DPaW-managed land), but with a different distribution of the burns in the landscape. The purpose of testing a different arrangement of the treatments intensifying them around the WUI is to evaluate how much additional protection is gained for high-value human assets, such as residential buildings and infrastructure. This strategy, where different fire regimes are applied around the WUI and in more remote areas has been already suggested in the literature. Burrows and McCaw (2013) proposed a zoning approach based on values at risk for the south-west of Western Australia which divides the resource allocation for prescribed burning in three zones: (1) A community protection zone within 5 km radius of towns where fuels are maintained at < 4 years old, (2) a wildfire modification zone within 20 km radius of towns where fuels are maintained at 5-7 years old to modify wildfire behaviour and increase the likelihood of suppression of fires approaching towns, and (3) a biodiversity management zone where a mosaic of fuel ages is maintained. However, this zoning approach has not had an economic evaluation. Prescribed-burning costs per hectare close to the WUI have been shown to be more expensive than in remote areas (Berry and Hesseln, 2004; Berry *et al*. 2006; Calkin and Gebert 2006; Chapter 2) because of the spatial interactions between housing and fuels, which increases the complexity of the application of the treatment. Hence, the costs of such a strategy are likely to be much higher than the costs of the current distribution of the treatments. Although it appears to be a valuable risk-management approach, it is important to evaluate the economic impacts of the strategy to see whether the additional benefits compensate for the increase in costs. This analysis has been conducted for this study and this Section presents the results of the application of this strategy in the short term. For convenience, the table presented in Chapter 3 with the number of hectares treated is shown again (see Table 5.3).

Table 5.3. Hectares treated per category of treatment size and distance from towns (strategy intensifying prescribed burning close to towns)

Distance from closest town (km)	Size of treatment (ha)	Percentage of area treated (%)	Hectares treated for each prescribed-burning rate			
			5%	10%	15%	20%
< 5	< 100	15.00	5,930	6,100[a]	6,100[a]	6,100[a]
< 5	100-500	7.50	2,960	5,930	7,310[a]	7,310[a]
< 5	500-1500	7.50	2,960	5,930	8,890	11,850
< 5	> 1500	0.00	0	0	0	0
5-10	< 100	6.00	2,370	7,240[b]	9,610[b]	12,030[b]
5-10	100-500	5.00	1,970	7,190[b]	10,930[b]	14,900[b]
5-10	500-1500	6.00	2,370	4,740	12,840[b]	22,060[b]
5-10	> 1500	3.00	1,190	2,370	3,550	4,740
10-20	< 100	5.00	1,970	3,950	5,930	7,900
10-20	100-500	7.00	2,770	5,530	8,300	11,060
10-20	500-1500	5.00	1,970	3,950	5,930	7,900
10-20	> 1500	3.00	1,190	2,370	3,560	4,740
> 20	< 100	1.50	590	1,190	1,780	2,370
> 20	100-500	4.50	1,780	3,550	5,330	7,110
> 20	500-1500	6.00	2,370	4,740	7,110	9,480
> 20	> 1500	18.00	7,110	14,220	21,330	28,450
Total		100.00	39,500	79,000	118,500	158,000

[a] The number of hectares prescribed burned do not correspond to the proportion indicated because the maximum amount of area that can be prescribed burned within this category has already been treated.

[b] The number of hectares prescribed burned do not correspond to the proportion indicated because it has been augmented to compensate for the hectares missing in other categories.

Even though the distribution of the treatments for the close-to-towns strategy is very different from the current strategy, the optimal prescribed-burning rate is the same for both strategies. If the treatments are intensified around the WUI, the prescribed-burning rate that minimises the sum of management costs and damages in the short term is also 15% of DPaW-managed land (see Figure 5.6 and Table 5.4). This also corresponds to prescribe burning to a total of 118,500 hectares in the South West forest region, but the number of treatments, their size and their costs are different to the current strategy. With the close-to-towns strategy, 30% of the total area treated with any prescribed-burning rate is located within 5 km from the closest town–compared to 3.1% for the current strategy–and 27.5% of the total area treated is done in treatments <100 ha in size–compared to 1.45% for the current strategy. This results in a strategy where many more patches of smaller size are selected for the application of the treatment. The number of patches treated in the landscape with the close-to-towns strategy are 1263, 2030, 2540, and 3039 for the 5, 10, 15 and 20% prescribed-burning rates respectively, between 8 and 12 times more treatments than with the current strategy. The average size of a treatment (i.e. a patch prescribed-burned) for the close-to-towns strategy is 42 hectares, which is about 9 times smaller than the average treatment size for the current strategy. The average cost per hectare is AU$215, much more than the current strategy, which costs on average AU$65 per hectare. Although the benefits of prescribed burning are larger for the close-to-town strategy, the costs are larger again, such that, when the optimal rate is applied, the net benefits are considerably lower than those generated with the current strategy (compare Table 5.2 and Table 5.4). The net benefits of treatments with the current strategy are twice as large as those of the close-to-towns strategy when the optimal rate is applied. Note also that the net benefits of the close-to-towns strategy are only positive for a prescribed-burning rate of 15% or higher; below that level, the net benefits are negative. Thus, for the benefits of applying this strategy to compensate the increase in costs in the short term, at least 15% of DPaW-managed land has to be prescribed burned.

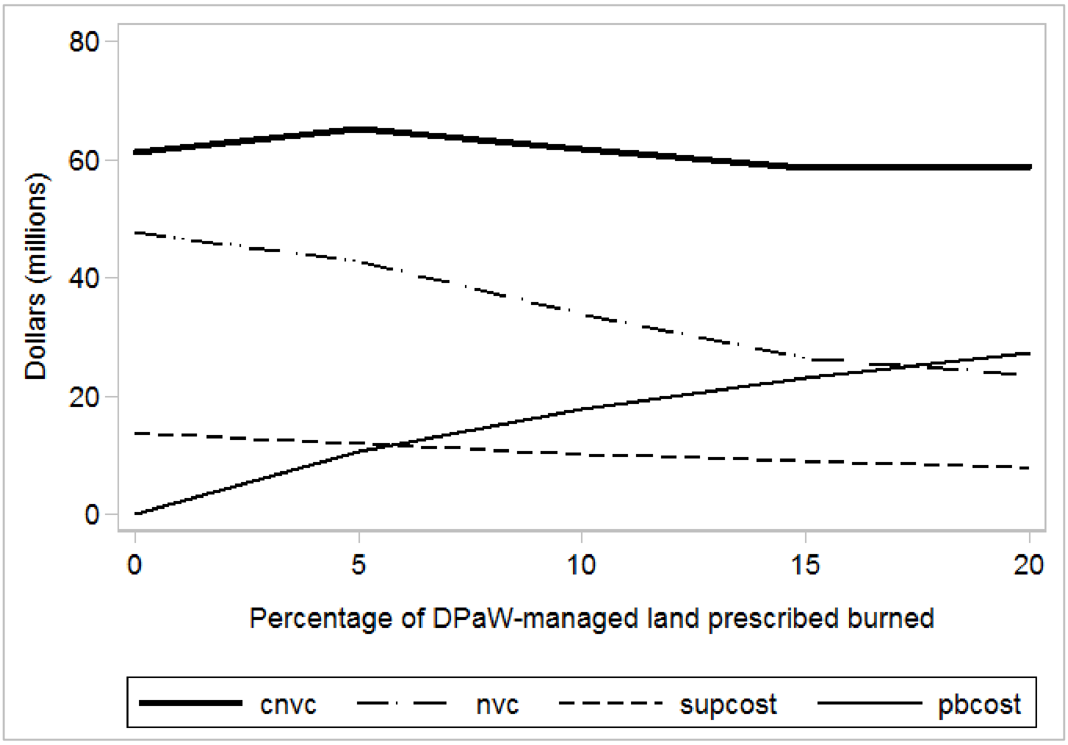

Figure 5.6. Cost plus net value change for different prescribed-burning rates applying the close-to-towns strategy (short-term analysis)

Table 5.4. Results for different prescribed-burning rates applying the close-to-towns strategy (millions of AU$, for a one-year time frame)

Rate	Prescribed-burning costs	Suppression costs	Damages	C+NVC	Suppression costs plus damages	Benefits (compared to 0%)[a]	Net benefits (compared to 0%)
0%	0.00	13.71	47.60	61.32	61.32		
5%	10.59	11.91	42.74	65.24	54.65	6.66	-3.92
10%	17.77	10.07	33.89	61.74	43.97	17.35	-0.42
15%	23.07	9.00	26.36	58.43	35.36	25.96	2.88
20%	27.40	7.79	23.56	58.75	31.35	29.96	2.57

[a] The benefits correspond to the reduction in suppression costs and damages compared to a no-intervention scenario (i.e. 0% prescribed burning)

If DPaW were to modify their strategy and intensify the amount of prescribed burning around towns, the application of this strategy to 15% of DPaW-managed land would require the Department to increase the current prescribed-burning investment in the South West forest region by 736%, an unrealistic increase in the investment.

As for the current strategy, the results of the C+NVC with the close-to-towns strategy for the different prescribed-burning rates are not substantially different. In the short term, damages decrease more rapidly when prescribed burning is concentrated close to towns because of improved protection to highly valued human assets. However, prescribed burning close to towns is considerably more expensive and the costs of applying the treatment increase more rapidly than the reduction in damages and in suppression costs. For the close-to-towns strategy the C+NVC curve does not exhibit the expected U shape of the theoretical model. Instead, the curve increases at first, then decreases between 5 and 15%, and finally increases again. For this strategy the C+NVC curve also looks relatively flat (Figure 5.6). The total sum of management costs and damages for the different prescribed-burning rates ranges between AU$58 and AU$65 million, corresponding to a maximum percentage change of 11%, which is similar to the variation for the current strategy. As a result, for the close-to-towns strategy there is also a wide range of near-optimal strategies and the analysis does not provide a clear-cut answer.

Although the C+NVC curve looks broadly similar for both strategies, the proportions of management costs and damages are very different. An increase in the amount of prescribed burning in the WUI results in a more marked reduction in damages in the short term, compared to the reduction achieved by the current strategy. But prescribed burning close to towns is a lot more expensive per hectare. Consequently, total prescribed burning costs increase more rapidly with this strategy and represent a higher proportion of the total sum of costs and damages compared to the current strategy, particularly for high levels of prescribed burning. When 15% or more of DPaW-managed land is treated with the close-to-towns strategy, management costs (prescribed burning plus suppression costs) are much higher than the total damages (see Figure 5.7). In fact, management costs are 15% higher than damages if the optimal prescribed-burning rate is applied and 60% higher than damages if a rate of 20% is applied. In contrast, for the current strategy the sum of management costs never exceeded the damages.

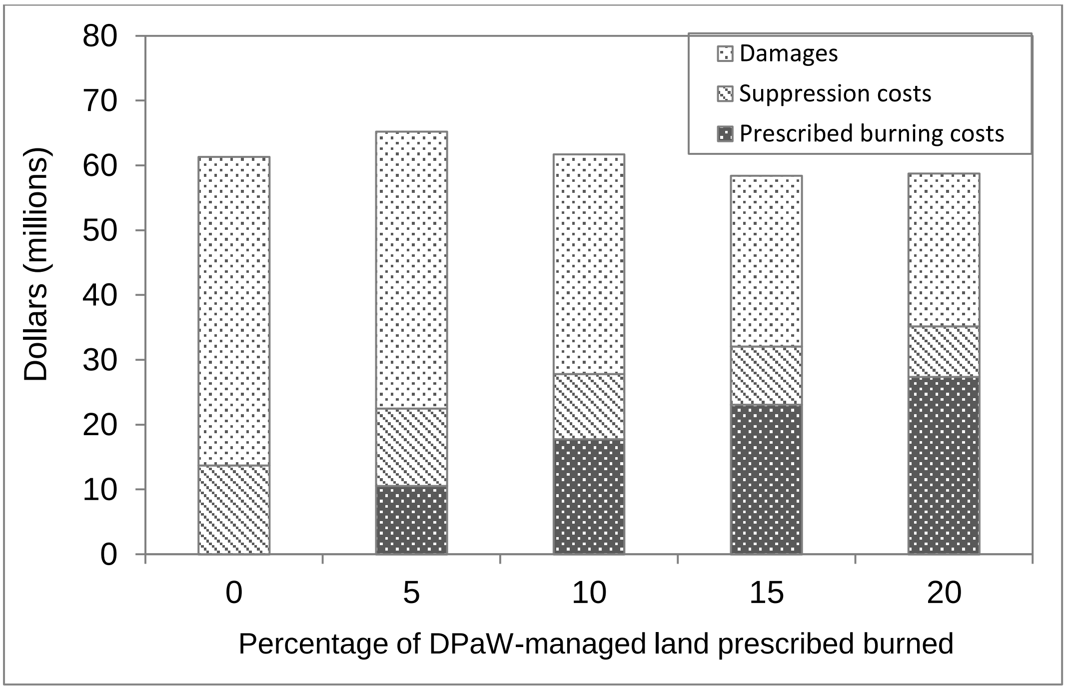

Figure 5.7. Damages, suppression costs and prescribed burning costs for each prescribed-burning rate using the close-to-towns strategy

With the close-to-towns strategy, management costs are dominated by prescribed burning. With additional area treated, while prescribed-burning costs increase rapidly, the reduction in suppression costs is slow. This is because in this model suppression costs are calculated as a function of area burned by wildfires and with the close-to-towns strategy the reduction in wildfire area in the short term is slight. Consequently, prescribed-burning costs rapidly exceed suppression costs and at high rates of prescribed burning, prescribed-burning expenditure is nearly 4 times higher than suppression expenditure.

Considering a one-year time frame, the proportional reduction in average wildfire area resulting from additional prescribed burning in the short term with the close-to-towns strategy is small, and in fact similar to the reduction in wildfire area achieved with the current strategy. This is despite large areas being prescribed burned. As a result, the total area subjected to fire of any type is substantially increased with prescribed burning (Figure 5.8). When the optimal rate of prescribed burning is applied with the close-to-towns strategy (15% of DPaW-managed land), 126,190 hectares are burned by both wildfires and prescribed burns, which is a substantially larger amount than the area that would be subjected to wildfires alone if there was no prescribed burning before the fire season (i.e. a total of 11,720 hectares). On the other hand, prescribed burns are generally lower in intensity, and are usually well under control.

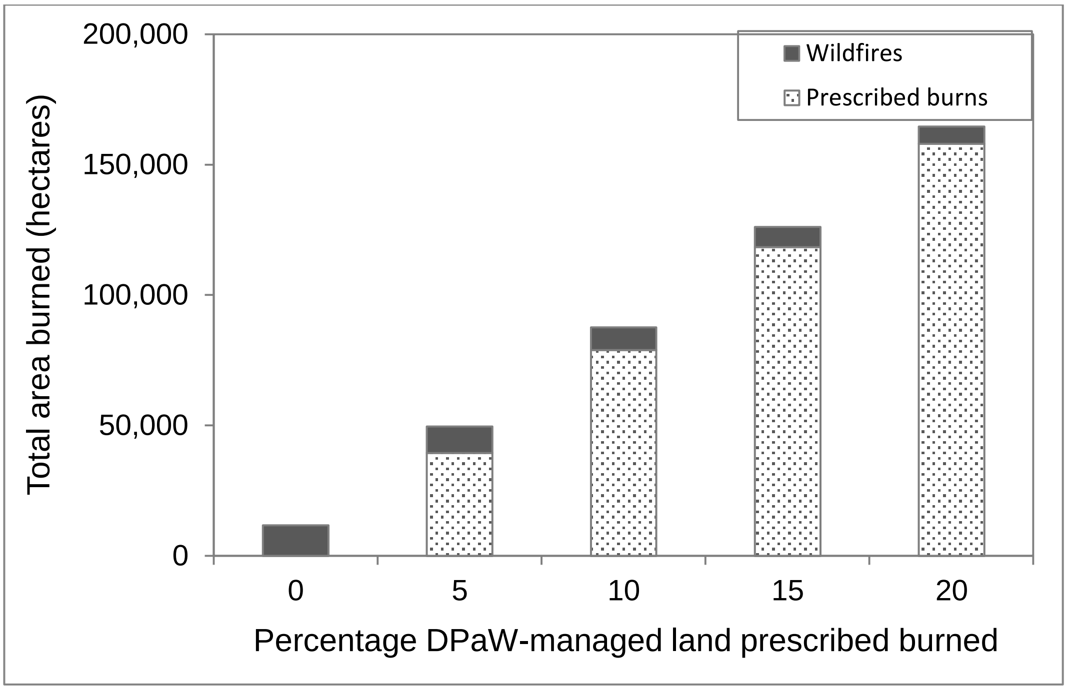

Figure 5.8. Total area subjected to fire of any kind (prescribed or wildfire)

Even in the short term, the spatial arrangement of the treatments matters. The reduction in average wildfire area between the two strategies is very similar (see Figure 5.9 and compare it with Figure 5.4), but the reduction in total damages is very different (Figure 5.10). The main difference is in the average area burned in urban areas. On average, the close-to-towns strategy results in less urban areas lost to wildfires. Even though the difference in the areas burned in urban areas between the two strategies is minor (of the order of 2 to 16 hectares difference), it is enough to make a considerable difference in the damages avoided, largely because urban areas have the highest value per hectare of all land-use categories.

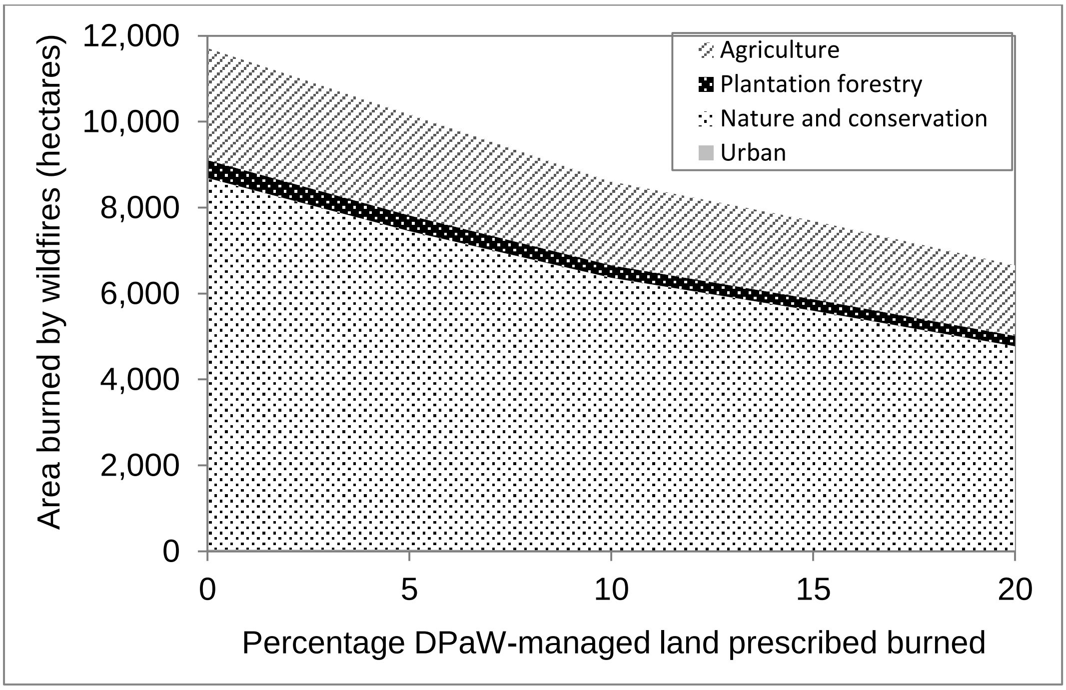

Figure 5.9. Area burned by wildfires per land-use type for different prescribed-burning rates

Thus, in the short term there is a trade-off between management costs and potential damages: on the one hand, management costs are high when a large number of the treatments are located close to high value assets in the WUI, but this may significantly reduce the amount of potential damages; on the other hand management costs are lower when the majority of the area treated is further away from towns, but the reduction in potential damages is less important.

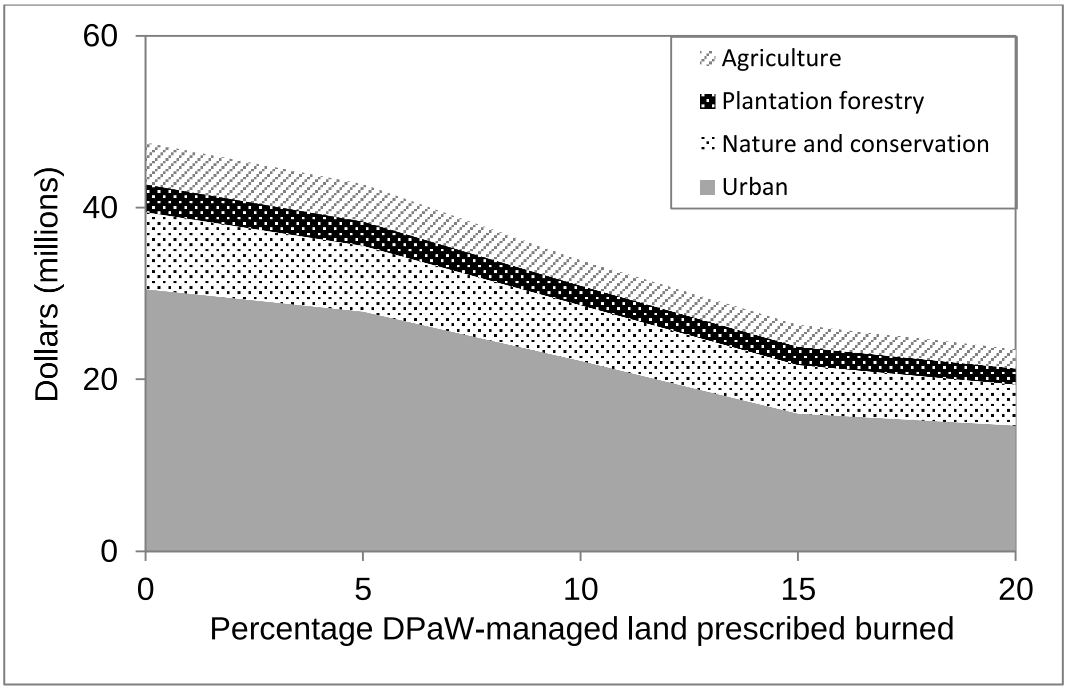

Figure 5.10. Wildfires damages per land-use type for different prescribed-burning rates

5.2.3. Comparing the two strategies

Figure 5.11 shows the average reduction in wildfire area resulting from each strategy in the short term. It can be seen that when prescribed burning is increased with either strategy the results are very similar in terms of reduced wildfire area. However, if instead of the reduction in wildfire area, both strategies are compared according to the reduction in damages and suppression costs achieved, then the results from both strategies differ considerably (see Figure 5.12). With the close-to-towns strategy it is possible to achieve a much higher reduction in damages and suppression costs for the same amount of area prescribed burned. This shows that optimising the spatial distribution to protect high-value assets results in substantial gains in terms of the reduction in damages to those assets. It also emphasises the fact that concentrating on minimising wildfire area without giving consideration to the differences in value between the different assets protected is an incomplete approach that provides only part of the information needed to make sound management decisions.

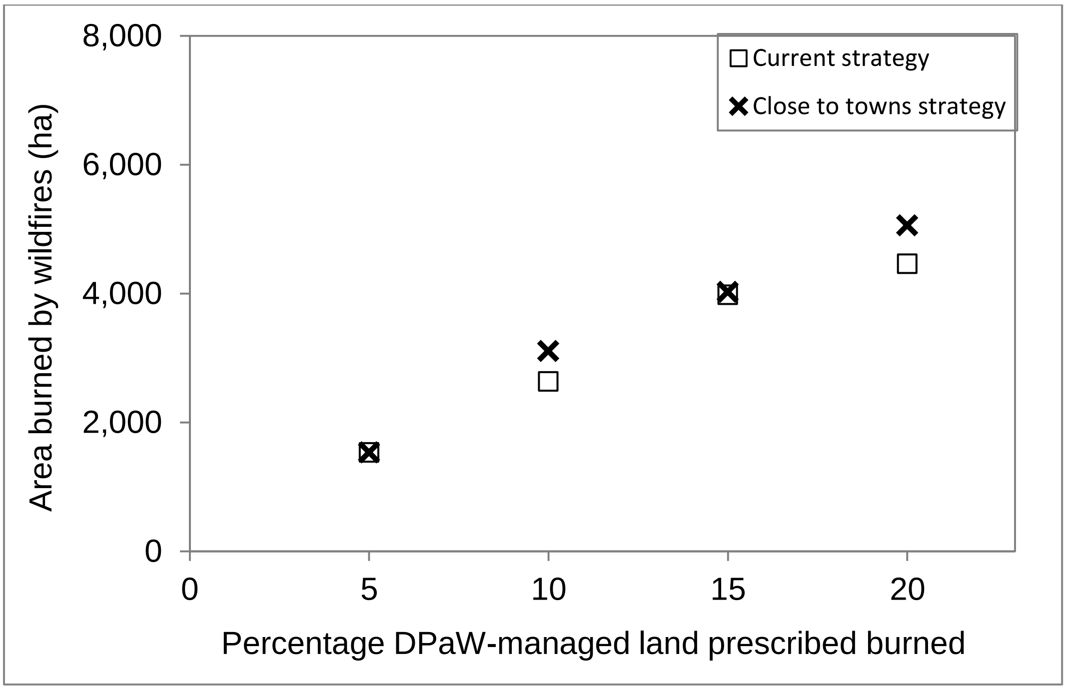

Figure 5.11. Reduction in area burned by wildfires for each strategy for different prescribed-burning rates

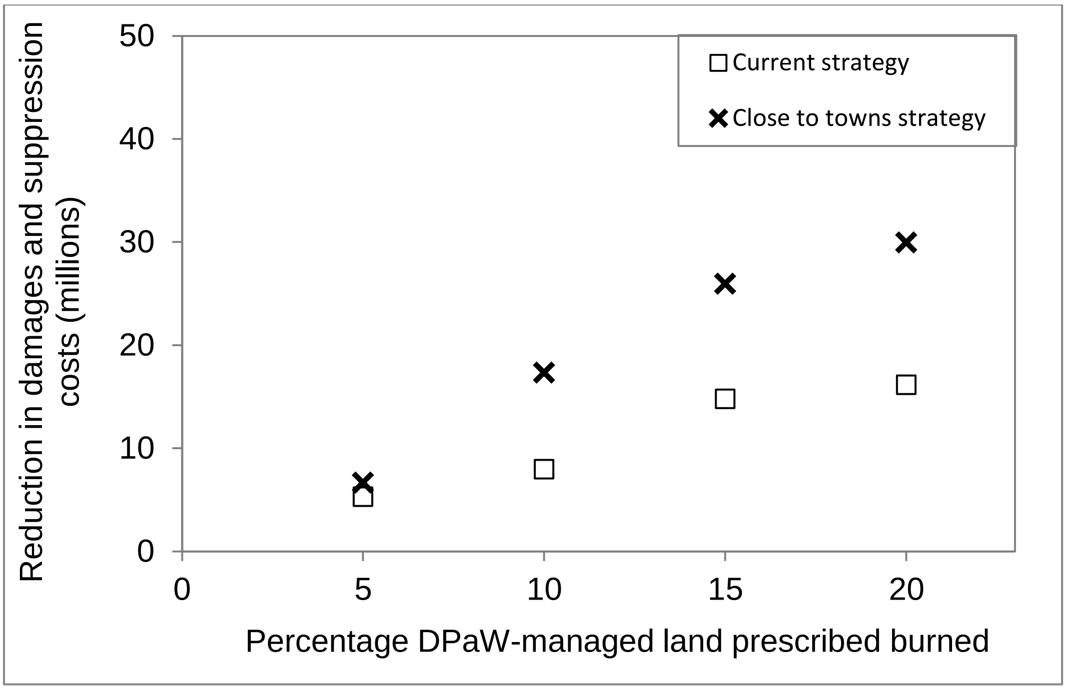

Figure 5.12. Reduction in damages and suppression costs for each strategy for different prescribed-burning rates

However, the information shown in Figure 5.12 is also incomplete. In this comparison the costs of prescribed burning are not included. If policy makers were to draw on the information provided in Figure 5.12 to select a strategy for the short term, it would seem that the strategy close to towns would be the most favourable strategy. But this may change if prescribed burning costs are accounted for.

In order to include prescribed burning costs in the comparison of both strategies, a new figure has been generated with a modified x axis (Figure 5.13). Instead of displaying the percentage of DPaW-managed land prescribed burned, the x axis has been replaced by the amount invested in prescribed burning. The y axis still displays the total reduction in wildfire damages and suppression costs. In Figure 5.13 it is possible to observe the level of benefits generated for different levels of investment in prescribed burning for each strategy. With the results presented in this manner, it is noticeable that the close-to-towns strategy is considerably more expensive. To achieve the same level of reduction in damages and suppression costs in the short term, the close-to-towns strategy requires a greater investment in prescribed burning. For instance, in order to achieve a reduction of AU$16 million in damages and suppression costs in the short term, an investment of approximately AU$10.3 million would be necessary with the current strategy, whereas for the close-to-towns strategy to achieve the same reduction in damages and suppression costs, an investment of AU$17.8 million in prescribed burning would be required. The option that generates the highest benefits is the rate/strategy that is further from the diagonal line (line where benefits = costs), in this case a rate of 15% with the current strategy (labelled with the letter ‘A’ in Figure 5.13).

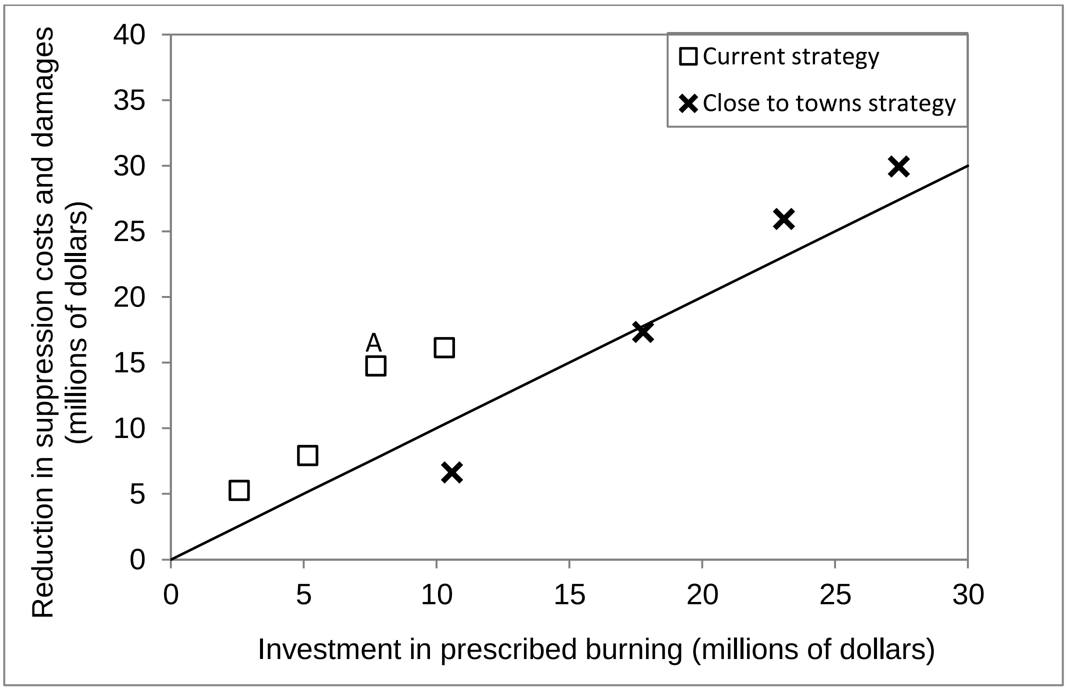

Figure 5.13. Reduction in suppression costs and damages for different levels of investment in prescribed burning

The diagonal line indicates a benefit : cost ratio of 1:1

From a fire management perspective the current strategy may be favoured given that for any prescribed burning budget it generates more benefits per dollar invested in the short term. Since fire management agencies have limited budgets for fire prevention activities, if they were to choose the strategy that generates the highest reduction in damages and suppression costs for a given budget, the current strategy is likely to be always preferred. Figure 5.13 shows that regardless of what the budget for prescribed burning is, it is more beneficial to society to implement the current strategy in the short term. The average budget for prescribed burning in the South West forest region is approximately AU$3 million per annum; with this budget, the current strategy would be preferred over the close-to-towns strategy because it generates more benefits for the same amount of money invested in prescribed burning.

5.3. Sensitivity analysis

A sensitivity analysis was conducted to assess the robustness of the results and assess how much the results change with changes to the parameters. Because there is a high level of uncertainty regarding the range of values that the different parameters can take, all parameters were increased and decreased by 50%. Each parameter was changed independently, while all the others were left unchanged. Although there can be an interaction between the different parameters and a situation may arise where the increase (or decrease) in one parameter causes the increase (or decrease) in another parameter, for ease of analysis, these interactions are not explored here. The values of the following parameters were changed for both of the strategies tested: the value per hectare of different land uses, prescribed burning costs, suppression costs, the relationship between intensity and damages, and the probability of large fires.

This Section is organised as follows: first, the results of the sensitivity analysis for the current strategy are presented. Then, we look at the results of the sensitivity analysis for the close-to-towns strategy. And finally, a different approach is used in which all hectares in the landscape have the same value, in order to investigate the difference it makes to account for the differences in value of various assets.

5.3.1. Sensitivity analysis: current strategy

The first sensitivity analysis shows that the results of the short-term analysis for the current strategy are fairly robust, in terms of the optimal rate of prescribed burning (see Table 5.5).

When the value of different parameters is increased or decreased by 50%, the optimal prescribed-burning rate remains the same (15%) in nearly all cases. The only change in the optimal rate occurs when prescribed burning costs are reduced by 50%, in which case the optimal rate is 20% of DPaW-managed land prescribed burned. The C+NVC curve remains quite flat and none of these changes make the optimal prescribed-burning rate more clear-cut (see Figure 5.14).

However, what changes significantly is the total sum of management costs and damages, that is, the C+NVC. With changes of +50% and -50% to the parameters, the minimum of the C+NVC varies between AU$35.5 million and AU$66.8 million (Table 5.5 and Figure 5.14) corresponding to a change of approximately −35% and +23% from the baseline results presented in Section 5.2.1 (the baseline results are also reported at the top of Table 5.5).

Table 5.5. Sensitivity analysis for the current strategy

	Optimal rate (% DPaW-managed land prescribed burned)		C+NVC (AU$ millions)	
Baseline results	15		54.28	
	Change in parameter			
	-50%		+50%	
Parameter	Optimal rate	C+NVC (AU$ millions)	Optimal rate	C+NVC (AU$ millions)
Urban areas	15	41.75	15	66.81
Conservation areas and State forests	15	51.38	15	57.18
Plantation forestry	15	53.01	15	55.56
Agricultural (grazing, cropping)	15	53.31	15	55.25
Agricultural (vine fruits)	15	53.20	15	55.36
Prescribed burning costs	20	50.33	15	58.14
Suppression costs	15	49.75	15	58.81
Level of damage (depending on the level of intensity)	15	35.54	15	56.27
Probability of large fires	15	50.37	15	55.63

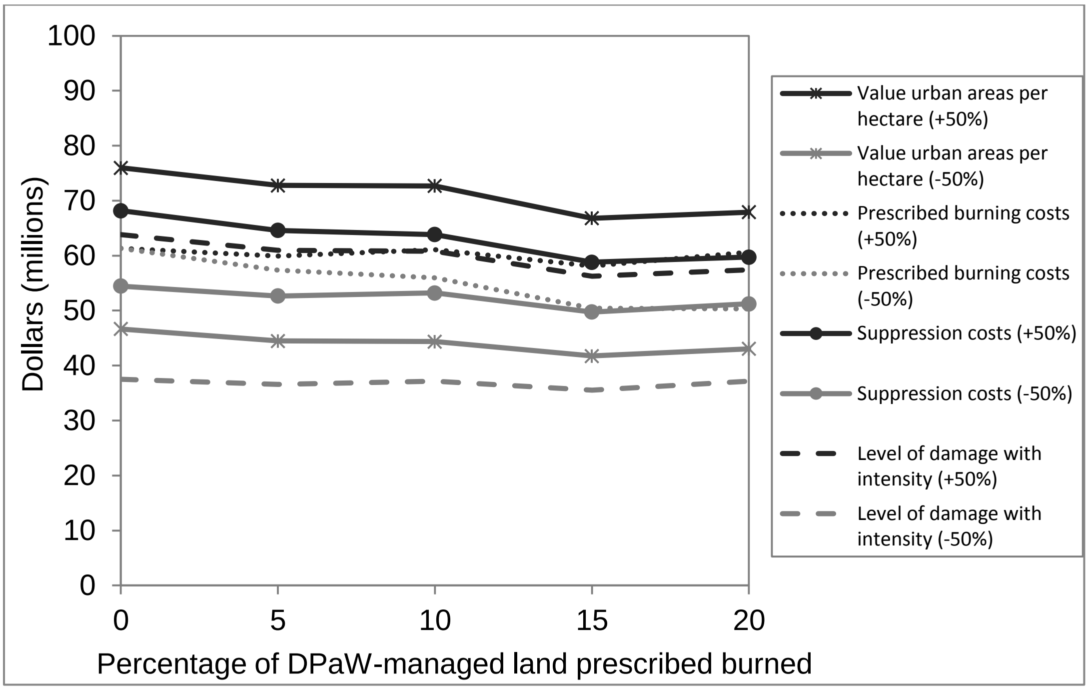

Figure 5.14. Changes in the C+NVC curve with changes in selected parameters

A more significant increase in prescribed burning costs was also considered in this sensitivity analysis. Prescribed burning costs may increase by more than 50% because there are only a small number of days on which the climatic conditions (temperature, humidity and wind speed) are mild enough for a fire to burn safely without escaping, but also warm and dry enough for the burns to spread to their intended boundaries instead of extinguishing too early. All the prescribed burns have to be carried out on these few days. If the area to prescribe burn is to be increased, a larger number of treatments would have to be applied on the same number of days. As a result, the high demand on resources on the few suitable climatic days may increase the costs of prescribed burning by more than the increase in area. In addition, as the chances of an escaped prescribed burn increases with a larger number of treatments, the resulting damages would need to be taken into account. For these reasons, an exponential increase in prescribed burning costs has been also considered in this sensitivity analysis.

In order to estimate an exponential increase for prescribed burning costs, the subject was discussed with the Regional Fire Coordinator for the South West forest region. From this discussion, it was established that if the higher demand on resources and additional escapes were taken into account, the increase in prescribed burning costs could follow an exponential increase such as the one depicted in Figure 5.15 (Peter Gibson, personal communication 2015). For the function shown in Figure 5.15, the rate of growth for prescribed burning costs increases by a factor of about 0.0016 with additional area treated. In other words, the second derivative of the prescribed burning costs function with respect to area treated is equal to approximately 0.0016. The exponential increase in prescribed burning costs shown in Figure 5.15 corresponds to the upper bound for the increase in costs (Peter Gibson, personal communication 2015).

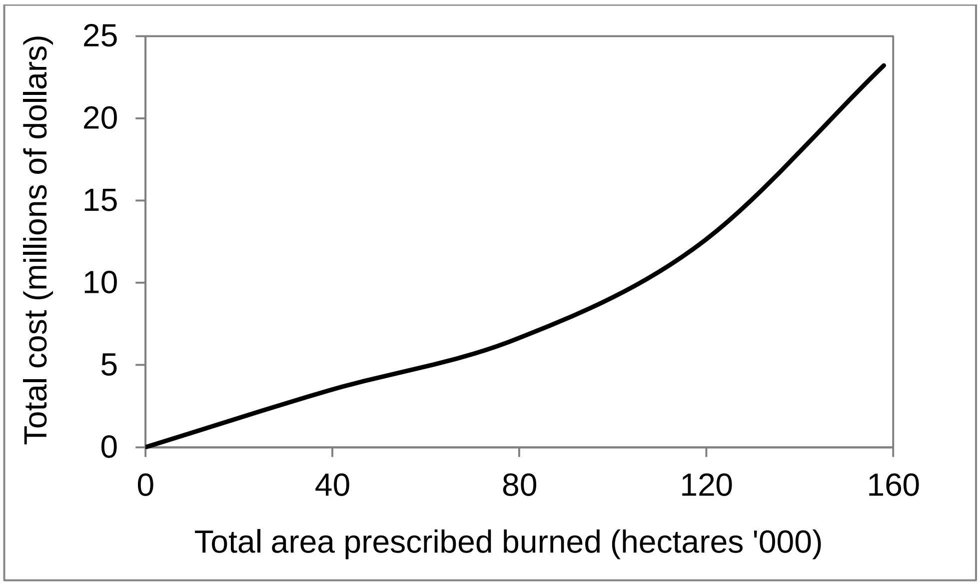

Figure 5.15. Exponential increase in prescribed burning costs

When prescribed burning costs are increased exponentially for the current strategy, the optimal rate of prescribed burning in the short term does not change. It is still 15% of DPaW-managed land in the South West forest region. The C+NVC curve remains flat between 0 and 15% (see Figure 5.16), so there is still a wide range of near-optimal solutions and there is not a clear-cut answer. However, beyond 15% there is a more noticeable increase in the C+NVC due to the more rapid increase in prescribed burning costs. In this case, investments in prescribed burning in the region would generate only between AU$0.7 and AU$1.5 benefits per dollar invested compared to a no-prevention strategy. But overall, even with an exponential increase in prescribed burning costs, the results do not change significantly compared to the base-case scenario presented in Section 5.2.1. If prescribed burning costs are assumed to increase exponentially in the short term, DPaW investment in the South West forest region would need to increase by nearly 295% to achieve the optimal rate of prescribed burning of 15%.

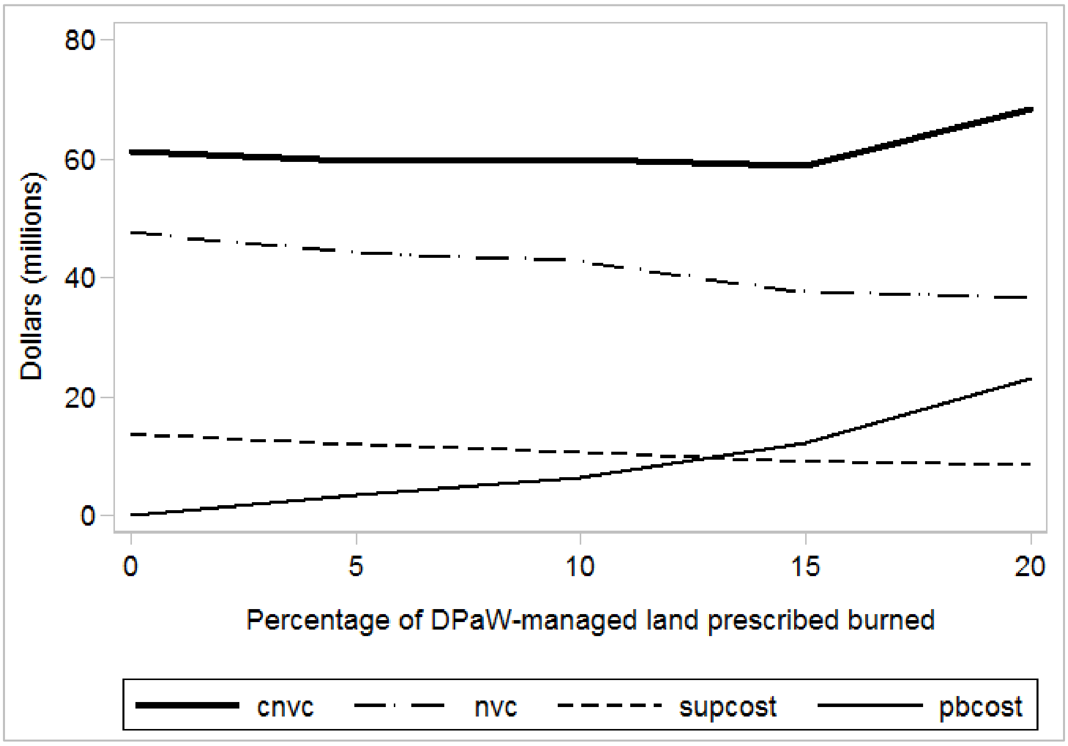

Figure 5.16. Cost plus net value change for different prescribed-burning rates with exponential increase in prescribed burning costs (current strategy)

5.3.2. Sensitivity analysis: close-to-towns strategy

The sensitivity analysis for the close-to-towns strategy shows that the results from this strategy are less robust; they are considerably more sensitive to changes in the different parameters. With changes to plantation forestry or agricultural values, the optimal prescribed-burning rate remains the same, that is, 15% of DPaW-managed land (see Table 5.6). However, with increases in the values of urban areas, conservation areas and State forests and suppression costs, the optimal prescribed-burning rate increases to 20%. The results of this strategy are also very sensitive to changes in prescribed burning costs. If prescribed burning costs are 50% lower, the optimal rate is 20%, but when prescribed burning costs are 50% higher, the optimal rate is 0%. Furthermore, when the level of damage caused by different levels of wildfire intensity is reduced by 50%, the low benefits generated by the application of prescribed burning are not enough to outweigh the costs of the treatment. In this case, the model indicates that no prescribe burning is the optimal solution.

Table 5.6. Sensitivity analysis for the close-to-towns strategy

	Optimal rate (% DPaW-managed land prescribed burned)		C+NVC (AU$ millions)	
Baseline results	15		58.43	
	Change in parameter			
	-50%		+50%	
Parameter	Optimal rate	C+NVC (AU$ millions)	Optimal rate	C+NVC (AU$ millions)
Urban areas	15	50.82	20	65.70
Conservation areas and State forests	15	55.38	20	61.34
Plantation forestry	15	57.29	15	59.58
Agricultural (grazing, cropping)	15	57.58	15	59.29
Agricultural (vine fruits)	15	57.91	15	58.96
Prescribed burning costs	20	45.05	0	61.29
Suppression costs	15	53.94	20	62.65
Level of damage (depending on the level of intensity)	0	37.51	15	59.83
Probability of large fires	10	55.42	15	60.48

The total sum of management costs and damages also changes substantially with changes to the different parameters. The fluctuation of the C+NVC curve for the close-to-towns strategy is similar to the fluctuation of the C+NVC curve for the current strategy. The C+NVC varies between AU$37 and AU$65 million. But the difference here is that in some cases the model shows a more clear-cut answer. For instance, when prescribed burning costs are increased or decreased by 50%, the model clearly shows that the optimal rate in the short term is either 0% (when costs are increased) or 20% (when costs are decreased) (see Figure 5.17). Similarly, when the level of damage cause by the fires depending on the level of intensity is reduced by 50%, the optimal rate is distinctively 0%. An increase in the value per hectare of urban areas also increases the benefits of applying more treatments close to high value human assets, when the value of urban areas per hectare is increased by 50% the optimal rate is 20%.

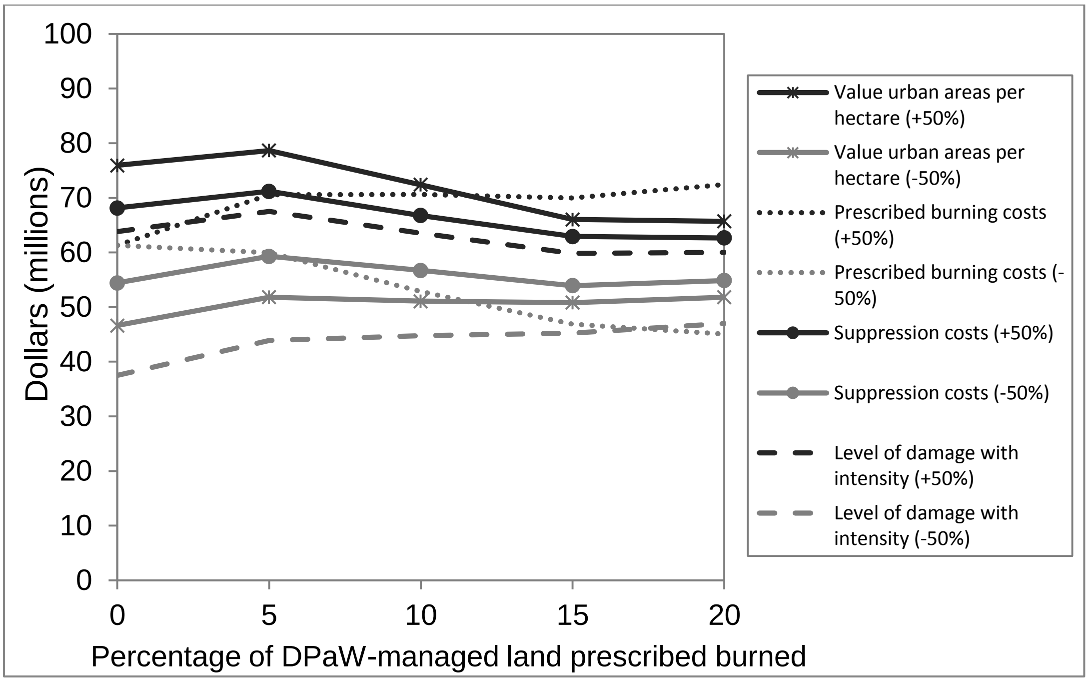

Figure 5.17. Changes in the C+NVC curve with changes in selected parameters

If prescribed burning costs are assumed to increase exponentially for the close-to-towns strategy, the benefits generated by the treatments in the short term do not compensate the increase in costs. In this case, prescribed burning costs are prohibitively expensive and the optimal prescribed-burning rate is 0%. In addition, the C+NVC curve is not as flat, and the optimal rate of prescribed burning is more evident, the model clearly points to 0% (see Figure 5.18).

This shows that the results from the close-to-towns strategy are highly sensitive to prescribed burning costs and any change in them leads to a change in the optimal prescribed-burning rate for the short term. The expected benefits that the strategy may generate in the short term are potentially large, but depending on what is assumed about prescribed burning costs and how they change depending on their location with respect to the WUI, the results may be very different.

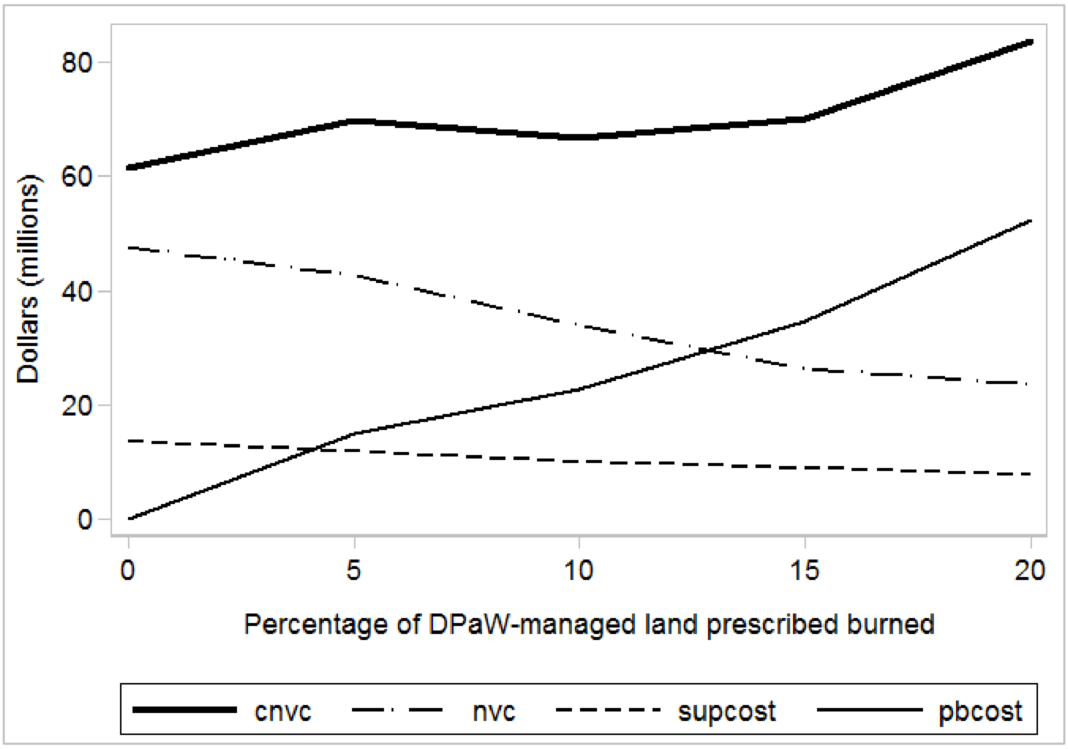

Figure 5.18. Cost plus net value change for different prescribed-burning rates with exponential increase in prescribed burning costs (close-to-towns strategy)

5.3.3. Sensitivity analysis on initial fuel levels

A short-term analysis of prescribed-burning investments, whether it is for the current strategy or the close-to-towns strategy, does not provide a clear-cut answer. In both cases the sum of management costs and damages does not change substantially when different rates of prescribed burning are applied and there is a wide range of near-optimal prescribed-burning rates. It seems that, in the short term, there is not a considerable difference in the sum of management costs and damages for different levels of prescribed burning. However, it is important to remember that for the short-term analysis the fuel layer on which prescribed burning is applied corresponds to the fuel levels as they were in June 2012. If a different year had been used, would the results change significantly? What if DPaW had not done any prescribed burning for several years before?

Indeed, the fuels of June 2012 already had a history of wildfires and prescribed burning. Between 2002 and 2012, the region had an average of 6% of DPaW-managed land prescribed burned. As a consequence, even if no prescribed burning is applied in the area for the following fire season, there are some patches in the landscape with very low fuels that would reduce the intensity of the fires simulated. Thus the 0% prescribed burning does not represent a landscape where a decision to not treat the fuels has been implemented for a long period of time. Since fuel levels and fire scars may significantly affect fire behaviour, it is necessary to evaluate the impact of changing the initial fuel levels on the results of the short-term analysis.

To achieve this, the same fires were simulated under the same weather conditions using a different initial fuel layer. For this new initial fuel layer, the fuel levels of June 2012 were artificially increased by 5 years and no fuel reduction was allowed to occur during that time (for simplicity purposes, it is assumed that no wildfires occurred during that period). The fuels in this new layer are between five and 79 years old. And the same treatments in the same locations under the current strategy were applied for the different prescribed-burning rates.

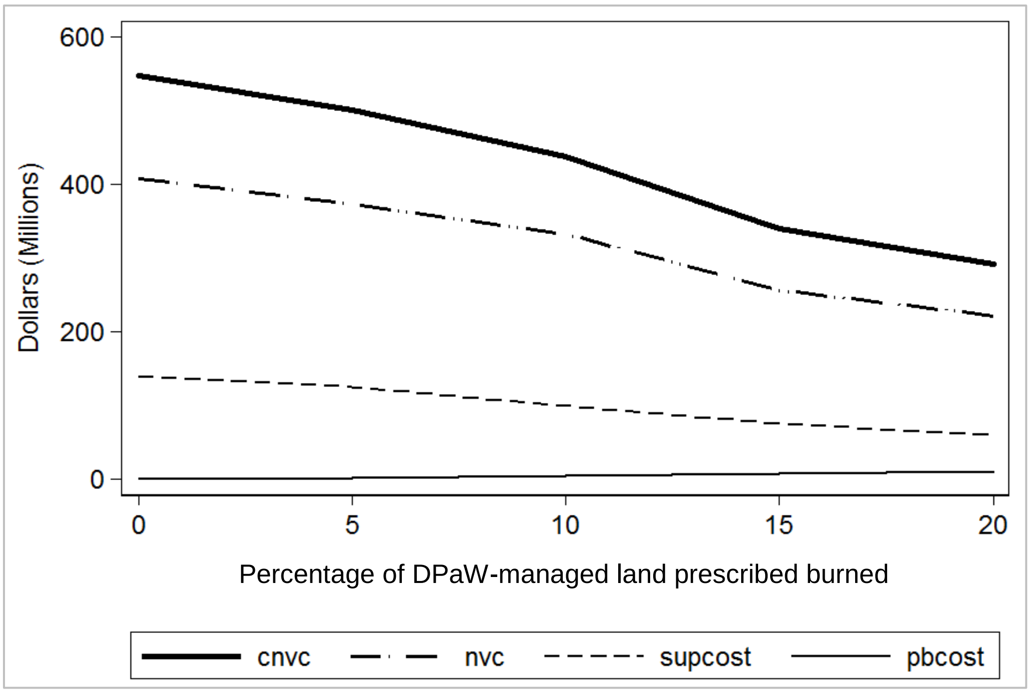

Figure 5.19. Cost plus net value change for different prescribed-burning rates with fuels ≥5 years old (current strategy)

Figure 5.19 shows the C+NVC results of the simulations for the current strategy using the fuel layer with fuels at least 5 years old. The results indicate that if the initial fuel levels are higher, the optimal prescribed-burning rate with the current strategy is at least 20%. Note that the C+NVC curve is no longer approximately flat and it clearly shows that there are substantial benefits to be gained from prescribed burning a large area. The potential benefits outweigh the costs of prescribed burning by a considerable amount. Because the slope of the C+NVC curve between 15% and 20% is relatively steep, the optimal prescribed-burning rate may be much higher than 20%. This shows that the initial fuel levels in the landscape on which prescribed burning is applied significantly affect the results of the short-term analysis.

The results presented earlier in this Chapter largely depend on what happened to the fuels of the South West forest region before June 2012. In the base-case scenario with the current strategy, the level of damages and suppression expenditure for 0% prescribed burning is much lower than what it would be if there had not been any prescribed burning between 2007 and 2012 (compare Figure 5.1 with Figure 5.19). This means that the results of a short-term economic analysis of prescribed burning cannot be extrapolated to other landscapes or other years. Here, the results of the simulations run in the fire season 2012-2013 are only valid for that particular fire season and only apply to the case-study area. Therefore, to be able to evaluate the economic consequences of investing in a particular prescribed-burning strategy for other fire seasons, it is necessary to simulate fires on a fuel layer that reflects the application of that prescribed-burning strategy for several years. This is explored in Chapter 6.

5.3.4. Comparison with the existing approach for the economic analysis of prescribed burning programs

The previous sections showed the results of a spatially explicit model that was developed to evaluate the economic consequences of changing the prescribed-burning rate and strategy in the short term. The purpose of developing a spatially explicit model was to enable us to evaluate the consequences to different types of assets in the landscape. This could not be done with the models previously developed in the literature, which use a single value for all hectares in the landscape (e.g. Mercer *et al*. 2007; Liang *et al*. 2010). The question is, what difference does it make for short-term decisions on prescribed-burning investments to have a spatially explicit model that differentiates the value of different assets in the landscape instead of a model that uses a common value for all hectares? In order to answer this question, the model was run again but with a common value for all hectares in the South West forest region. The common value used for this run was the weighted average of the values per hectare of different types of land uses (i.e. the sum of the values reported in the Methodology Chapter, Table 4.11, multiplied by the proportion of area for each land use). The value obtained in this manner is AU$8,600 per hectare.

The results of using a common value for all hectares in the landscape for the current strategy and for the close-to-towns strategy are shown in Figure 5.20 and Figure 5.21 respectively. When a common value is used for all hectares in the landscape, the level of damages is increased and the benefits of prescribed burning in the short term using the current strategy are overestimated, and the prescribed-burning rate that appears optimal increases to 20% or more. The C+NVC curve is less flat and clearly points to high levels of prescribed burning. Although the C+NVC is similar for 15% and 20%, the difference between 0% and 20% is more pronounced than previously estimated (compare Figure 5.1 in the Results Section 5.2.1 and Figure 5.20 below). The C+NVC varies between AU$107 and AU$76 million, a greater change (29%) than in the results presented in Section 5.2.1, where heterogeneity in values was accounted for.

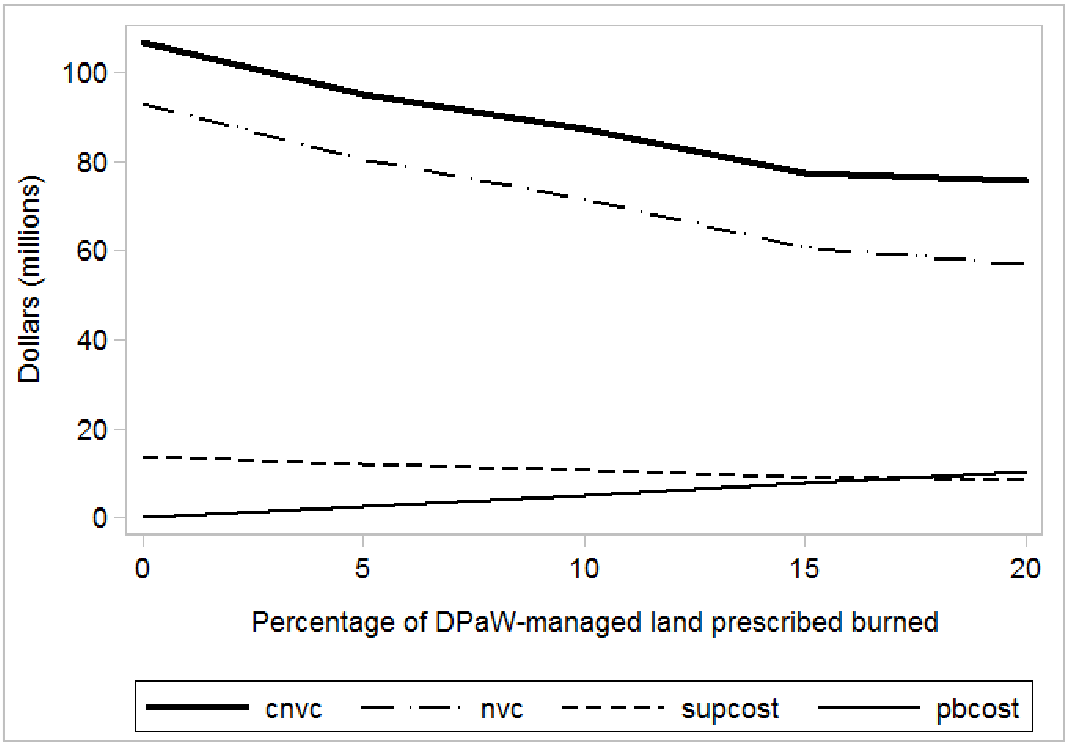

Figure 5.20. Cost plus net value change for different prescribed-burning rates using a single value per hectare for all hectares (current strategy)

For the close-to-towns strategy, when all hectares have the same value the optimal rate of prescribed burning is also 20% (or more) and the level of damages is also increased (Figure 5.21). But in this case, the C+NVC curve is less flat compared to the results presented in Section 5.2.2. The C+NVC ranges between AU$107 and AU$87 million, a change of 19%.

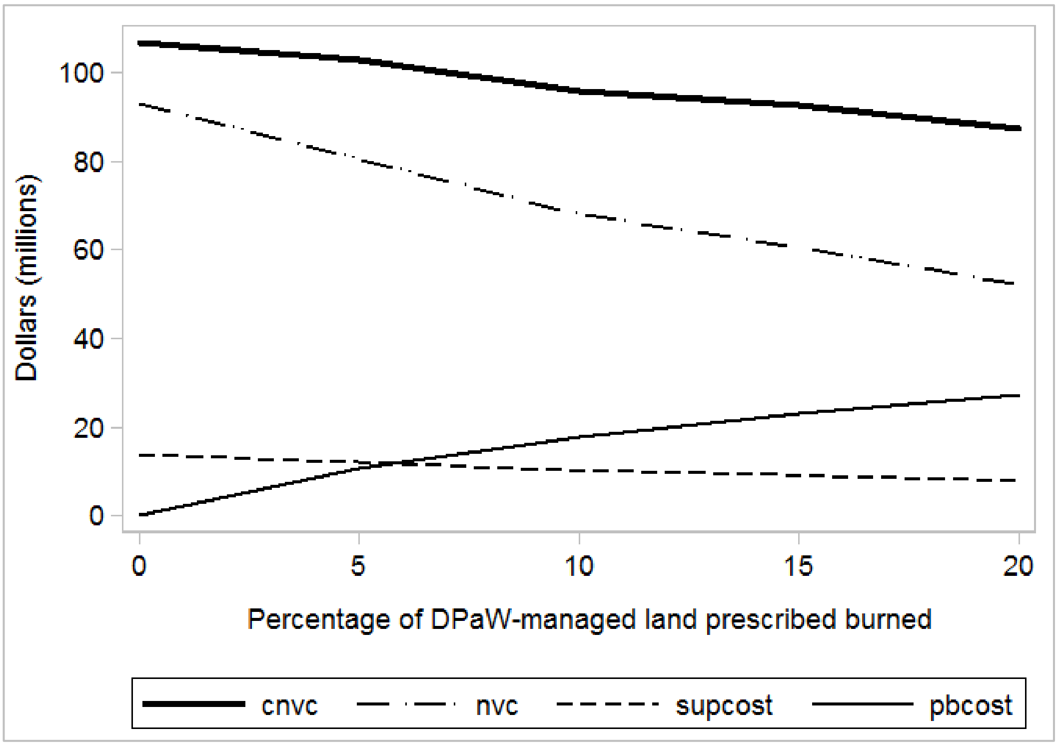

Figure 5.21. Cost plus net value change for different prescribed-burning rates using a single value per hectare for all hectares (close-to-towns strategy)

The reduction in total damages is very similar for both strategies. This is because, in this scenario, a hectare in a forested area is assumed to have the same value as a hectare in the WUI, and the simulated fires burned mostly forested areas. In addition, the reduction in total damages appears to be more rapid when all hectares have the same value than when they have different values. This is explained by the fact that wildfires mostly burn forested areas and in this run the value per hectare of forested areas is higher than in the base-case scenario.

If all types of assets have the same value, the most significant reduction in damages occurs in forested areas for both the current strategy and the close-to-towns strategy (see Figure 5.22 and Figure 5.23). In this case, damages in the WUI appear to be negligible. In contrast, when the difference in value for different assets is taken into account, the most significant reduction in damages arises from damages avoided in the WUI. If a model using a common value for all hectares in the landscape is used to evaluate different arrangements and rates of prescribed burning treatments, it may be concluded that treatments in the WUI do not generate substantial additional benefits. This is a valuable insight, because it shows that accounting or not accounting for the difference in asset values may lead to very different decisions in fire management in the short term. If the results showed from Figure 5.20 to Figure 5.23 are used to decide on the most beneficial prescribed-burning rate and the most beneficial arrangement of the treatments in the short term, the benefits expected from the strategy and rate implemented may not be realised.

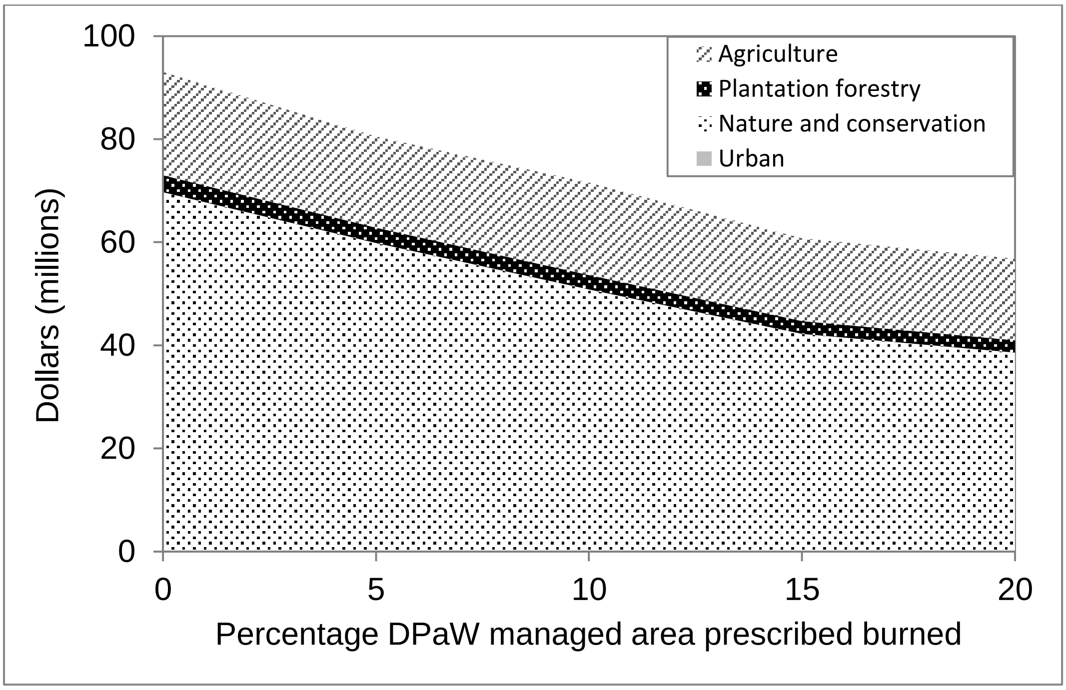

Figure 5.22. Wildfires damages per land-use type for different prescribed-burning rates using a single value per hectare for all hectares (current strategy)

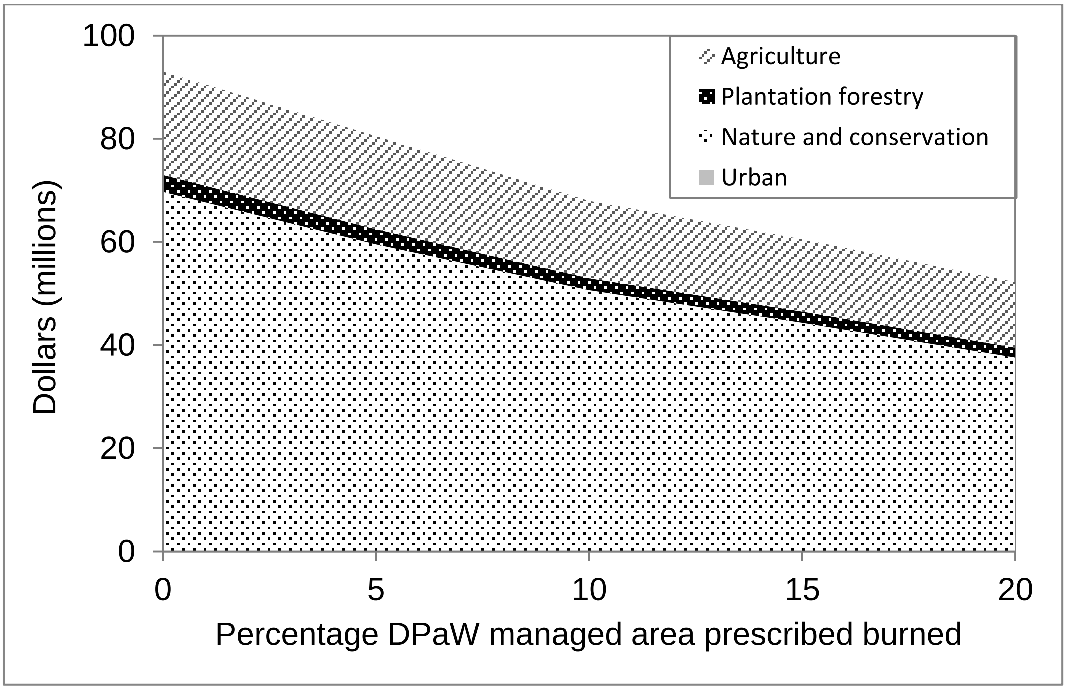

Figure 5.23. Wildfires damages per land-use type for different prescribed-burning rates using a single value per hectare for all hectares (close-to-towns strategy)

5.4. Discussion

Simulation modelling of the economic impacts of prescribed burning shows that wildfire damages in the South West forest region are likely to be reduced in the short term by the use of prescribed burning. However, the level of damages is only partially mitigated, it is not eliminated in the short term even when large areas are treated (i.e. when 20% of DPaW-managed land is treated). In addition, this research shows that the location of the treatments matters and wildfire damages reduce more rapidly when the treatments are primarily located around the WUI to protect high value human assets. Even though the reduction in wildfire area is very similar between both strategies, the difference in damages avoided is comparatively large.

To date, the majority of studies that have looked at the optimal location for prescribed burning treatments have concentrated on minimising wildfire area instead of wildfire damages (e.g. Liang *et al.* 2008; Bradstock *et al.* 2012b). Because there are numerous studies that focus on optimising prescribed-burning treatments to minimise wildfire area, there is an abundant discussion in the literature on prescribed-burning leverage. Prescribed-burning leverage indicates the number of hectares that need to be prescribed burned in order to reduce wildfire area by one hectare.

In the results presented in this Chapter, regardless of the strategy applied (current strategy or close-to-towns strategy), between 25 and 81 additional hectares need to be prescribed burned to reduce wildfire area by one hectare. This number is high compared to other studies. For instance, Boer *et al*. (2009) estimated a leverage of 4:1 in some parts of the south-west of Western Australia (i.e. four hectares of prescribed burning are needed to reduce wildfire area by one hectare). And in a study of the forested areas east of Sydney, Price and Bradstock (2011) estimated a leverage of 3:1. The high number of hectares that need to be prescribed burned in order to reduce wildfire area in the short term can be explained by the previous history of prescribed burning that occurred in the area. The landscape on which the simulations were performed had a history of prescribed burning and wildfires, so any additional hectares treated made a relatively small difference. If the fuels in the landscape had not been previously reduced through the application of prescribed burning and the occurrence of wildfires, a smaller number of hectares treated may have been needed to achieve a similar reduction in wildfire area.

However, the problem with concentrating on treatment leverage is that it does not tell us anything about the capacity of prescribed burning to reduce wildfire damages, or if the leverage can be increased by placing the treatments in different locations. There are other questions that may be more relevant to fire managers and policy makers such as: How much wildfire damage can be avoided by investing one additional dollar in prescribed burning? How would this change when the treatments are concentrated around the WUI? These questions have been answered throughout this Chapter for short-term decisions about prescribed burning.

In the results presented in this Chapter, it has been shown that increasing the area treated close to the WUI results in a more substantial reduction in potential damages in the short term compared with following the current spatial pattern. However, because prescribed burning close to the WUI is more expensive, in the short term one dollar invested in prescribed burning with the current strategy generates more benefits than one dollar invested in prescribed burning close to the WUI. This highlights the importance of using spatially explicit models for the economic analysis of prescribed burning. The spatial nature of the issue calls for a model that accounts for the difference in value of the assets protected as well as the changes in prescribed burning costs per hectare with the size and the location of the treatments. Because the model developed for this study includes these factors, it is possible to evaluate the economic impacts of changing the location of the treatments in the landscape. It is also possible to discern the different proportions of management costs and potential damages resulting from the strategies implemented in the short term, and ascertain where the highest benefits are generated in the landscape.

This analysis shows that focusing on reductions in wildfire area alone can result in sub-optimal decisions about strategies. Reductions in wildfire area are important, but they are only part of a bigger picture that needs to include potential damages and management costs and how these vary depending on the strategy implemented. A more comprehensive approach for policy making that may be used to make decisions in the short term has been shown in Figure 5.13. This figure reported the reduction in damages and suppression costs for different levels of investment in prescribed burning and for different arrangement of the treatments.

5.4.1. Implications for wildfire management in the short term in the South West forest region

These results suggest that it would be worthwhile to increase current prescribed-burning investment in the South West forest region to very high levels. Although a rate of 15% was applied in the 1960s and 1970s, for today's context in the South West forest region, this would be an exceptionally high level of prescribed burning. First, with the recent and future changes to the global climate, applying such a high rate may not be feasible. Indeed, the number of suitable days during which the treatment can be applied has been reduced in the past 60 years (Burrows and McCaw 2013). Further, fuels have accumulated in the landscape in the south-western forests of WA in the past 50 years (Boer *et al*. 2009) due to increases in suppression capability, higher rates of suppression success and lower rates of prescribed burning (Burrows and McCaw 2013; McCaw 2013). With higher levels of fuels, the climatic conditions need to be milder to be able to apply the treatment and minimise the risk of escape. Lastly, the WUI has continuously grown in the last few decades and will most likely continue to grow, as it has been the case in other forested landscapes, despite the presence of wildfires (Alexandre *et al.* 2015). This increases the complexity of the problem and makes the application of prescribed burning even more challenging. Hence achieving a rate of 15% would likely be impracticable and could potentially cost a lot more than what is indicated in these results, including those in the sensitivity analysis.

Based on a single-year assessment, an initial fuel load equal to that in 2012, and the values estimated for the different land uses in this model, the current strategy appears to be the most beneficial strategy for society in the South West forest region. The current strategy would most likely be preferred in the short term, regardless of the prescribed-burning rate applied. However, there is a high level of uncertainty associated with the results of the close-to-towns strategy. In order to increase the confidence in the results of the close-to-towns strategy, more information is needed on prescribed burning costs per hectare in the WUI. However, this information is not currently available from the agencies responsible for fire management in Western Australia. To obtain this information, government departments would need to expand on the information they collect and keep a certain level of disaggregation in the data collected. At present, the information on management costs collected is aggregated by region or district and the disaggregation of these costs to obtain prescribed burning costs per hectare as a function of size and location would be a lengthy and complex process. This process could be accelerated and the quality of per-hectare cost estimates improved if a larger amount of the information collected on the ground (at the moment the treatment is applied) was kept disaggregated in the appropriate databases (e.g. expenditures on different types of resources, human resources needed, equipment, size of the treatment, type and age of fuel, timing, etc.). This is now underway with the introduction of a Bushfire Risk Management System, a system that will be adopted by all Local Government Areas in Western Australia where all the information on wildfire risk reduction will be collected.

For both strategies, the results show that in the short term there is a wide range of near-optimal prescribed-burning rates. But even though the C+NVC curve is very flat and the short-term analysis does not provide a clear-cut answer, the proportion of management costs and damages within the C+NVC is very different between the different rates of prescribed burning. For fire managers this is also a very valuable insight.

There may be substantial non-market benefits gained from fire management programs (Venn and Calkin 2011) that are not considered in this study. If included, these non-market benefits may increase the benefits per dollar invested in the close-to-towns strategy and shift the preference towards prescribed burning close to the WUI.

5.4.2. Implications for the design of economic analyses of fire management programs

The C+NVC model has received a great deal of attention in the theoretical literature on fire economics. Since Sparhawk first formulated the Least-Cost-Plus-Loss model, the predecessor to the C+NVC model, numerous authors have put effort into improving, modifying and reformulating the model. Today, the C+NVC is the most widely accepted model for evaluating fire management programs. But one of the main issues with some of the studies using the C+NVC is the lack of clarity regarding the activities evaluated. For instance, studies that evaluate the economic impacts of varying pre-suppression investments, fail to clearly define the type of activities actually included in the term pre-suppression (Chapter 2).
Pre-suppression is broadly defined as everything that happens before a fire event, but this definition may correspond to a number of different activities such as prescribed burning, the construction and maintenance of fire breaks, fire education programs, detection systems, pre-positioning of resources, and in some cases it may also be equated to initial attack. Some authors define pre-suppression as “preventative activities”, but this definition is much too vague. Very few studies provide a clear explanation of what they mean by pre-suppression.

Some exceptions include Rodriguez y Silva and Gonzalez-Caban (2010), who define it as initial attack expenditures, Prestemon *et al*. (2012), who define it as fuel reduction through mechanical treatments, and Butry *et al*. (2010), who define it as a combination of prescribed burning and wildfire education programs.

Another aspect of the model where there is often lack of clarity is in the benefits. There is often ambiguity with regards to how the benefits are calculated. Do the benefits correspond to reduced damages, or reduced suppression costs, or both? And are other benefits included such as social benefits or environmental benefits? In this Chapter, the sources of benefits have been clearly specified and quantified.

This Chapter has shown that for prescribed-burning strategies in the South West forest region, a short-term analysis of one year does not provide a clear-cut answer and there are numerous near-optimal levels of funding. In addition, the results are very sensitive to what has been done before with the fuels in the landscape. Thus, a short-term economic evaluation of prescribed-burning programs in the South West forest region does not provide a full picture of the costs and benefits of the practice, since it is an inherently long-term issue.

For economic analyses of fire management to provide fire managers with tools that can help them make better decisions, it is essential that these analyses be clear about all these elements. Otherwise, the conclusions and recommendations derived from economic analyses may have little applicability because managers are unclear about what they relate to.

Chapter 6.

LONG-TERM ANALYSIS OF PRESCRIBED-BURNING STRATEGIES USING THE C+NVC MODEL

6.1. Introduction

The short-term analysis presented in the previous Chapter could not be used to answer a key question for fire managers: what would be the cost to society of not doing any prescribed burning for several years? And also: Given that loads respond dynamically to prescribed burning, what is the long run optimal level of prescribed burning? The previous Chapter showed that in the short term there is not much difference in the total sum of management costs and damages between prescribed burning a large area of the South West forest region or not doing any prescribed burning at all. What changes is the distribution of management costs and damages. But if a decision was made to minimise prescribed burning and utilise the fire management funds for fire suppression instead, what would be the long-term economic consequences of such a decision?

If the fuels in the landscape were not reduced through prescribed burning for several years, wildfire damages could increase substantially. As the sensitivity analysis of the short-term study showed, if the initial fuel levels of the simulation model are left to cumulate for five years in the region without any treatment, the costs to society of wildfires would increase significantly. However, this test does not entirely reflect what would happen in reality because wildfires are an inevitable disturbance of the South West of Western Australia, and it is unlikely that the fuels in the area would be able to increase undisturbed for several years. If there was no fuel reduction through prescribed burning, less frequent but larger, more intense fires would reduce the fuels in the landscape. This needs to be incorporated in the analysis in order to avoid an overestimation of the damages in a 0% prescribed-burning scenario. This Chapter reports a method developed to capture the long-term dynamics of fuel accumulation and reduction that occur in an untreated landscape. This method integrates wildfires and prescribed burning (where applicable) every year for several years, while allowing the fuels to accumulate in unburned/untreated areas. With this method it is possible to evaluate the long-term effects of prescribed burning and its economic consequences.

The main questions that this long-term analysis seeks to answer are: what is the optimal prescribed-burning rate in the long term for the South West forest region? And can a clear-cut optimal level be identified? With a long-term analysis it is also possible to get a better understanding of what could be expected if DPaW's prescribed-burning budget was reduced and used for suppression activities instead. In the fire management literature, it has been shown that the dramatic increase in suppression expenditures and fire severity in the last few decades has been partly the product of significant fuels build-up in the landscape due to an emphasis on rapid fire suppression and a limited focus on fuel reductions (Chapter 2). Greater suppression capacity and success rate have not improved the management of fire in the landscape (Morgan *et al*. 2007). But what are the full economic consequences of focusing on fire suppression in the South West forest region and how can prescribed burning help? To answer these questions, the C+NVC has been used to estimate the management costs and damages of different prescribed-burning rates that simulate the long-term application of the treatment.

This Chapter is organised as follows. Section 6.2 explains the methods used to incorporate the long-term dynamics of prescribed burning and wildfires in the economic model presented in Chapter 3. The same economic model is used, but the fuels on which the simulations were run are different to reflect the long-term application of prescribed burning, or the lack thereof. Section 6.3 present the results of the long-term analysis. The sensitivity analysis is presented in Section 6.4. Then Section 6.5 compares the results of the short-term and the long-term analyses. Section 6.6 discusses the results, comparing this with other economic studies of long-term fuel reduction programs in the literature.

6.2. Method used for generating the fuel layers for each prescribed-burning rate

Unlike the short-term analysis, in which the fuel levels of June 2012 were used as the basis for the simulations, the fuels used in the simulations for the long-term analysis are not assigned to any particular year. The fires were simulated on five entirely different fuel bases, each fuel base representing the long-term application of a particular prescribed-burning rate.

Prescribed burning, if applied for long periods of time, ultimately changes the fire regime of a region. Instead of having large, intense and infrequent fires, the application of prescribed burning transforms the fire regime into smaller, less intense and more frequent fires (Gill and Allan 2008). Each prescribed-burning rate characterises a different fire regime with different frequencies for the application of the treatments. In order to simulate the effects of different fire regimes on wildfire spread and intensity, and their economic consequences, the fuels in the case study area were modified to represent each of the fire regimes evaluated (i.e. prescribed-burning rates 0, 5, 10, 15 and 20% of DPaW-managed land). The same fires simulated in the short-term analysis (i.e. the same ignition points, with the same dates, times and variations of temperature for the duration of the fire) were simulated here again, but on fuels at an “equilibrium” level for the fire regime they represent. This “equilibrium” level means that the simulations were run on what would likely be a representative year of the prescribed-burning rate evaluated, if that rate was applied for a long period of time. The economic model in itself was not modified. Here follows an explanation of how each “equilibrium” or representative fuel layer for each of the prescribed-burning rates was created.

6.2.1. Fuel layer for 0% prescribed burning (only wildfires reduce the fuels in the landscape)

To create a fuel layer that represents a 0% prescribed-burning regime, and arrive at an “equilibrium” load were the fuels would be at a representative level, the following process was employed: First, the fuel levels of June 2012 were homogeneously increased by five years. This was done because it was deemed more efficient to start with a high level of fuels and simulated wildfires every year for several years until the fuels reached an “equilibrium” level, instead of starting with treated fuels. Second, an average wildfire season (i.e. 110 ignitions distributed across the landscape according to the probability of ignition of each pixel) was simulated on this fuel layer. Third, the areas burned by the wildfires were updated on the fuel layer and a value of zero was given to the fuel-age attribute of the pixels burned. In the rest of the landscape where no wildfires occurred, the fuels were allowed to accumulate for one year. Another average wildfire season with different ignition points was simulated on the new fuel layer and the area burned was again updated on the layer. This process was continued for 15 years, until the area burned by the wildfires each year was reasonably stable (see Appendix 2). Figure 6.1 illustrates this process. The resulting fuel layer at the end of 15 years was used as the “equilibrium” fuel layer for the 0% prescribed-burning. Finally, 121,000 fires (i.e. 220 fire seasons) were simulated on the “equilibrium” layer to generate the results of the C+NVC. It is assumed that the simulations produce what could potentially happen in any one year in the future if a decision was made to not prescribe burn and this decision was maintained for at least 15 years prior.

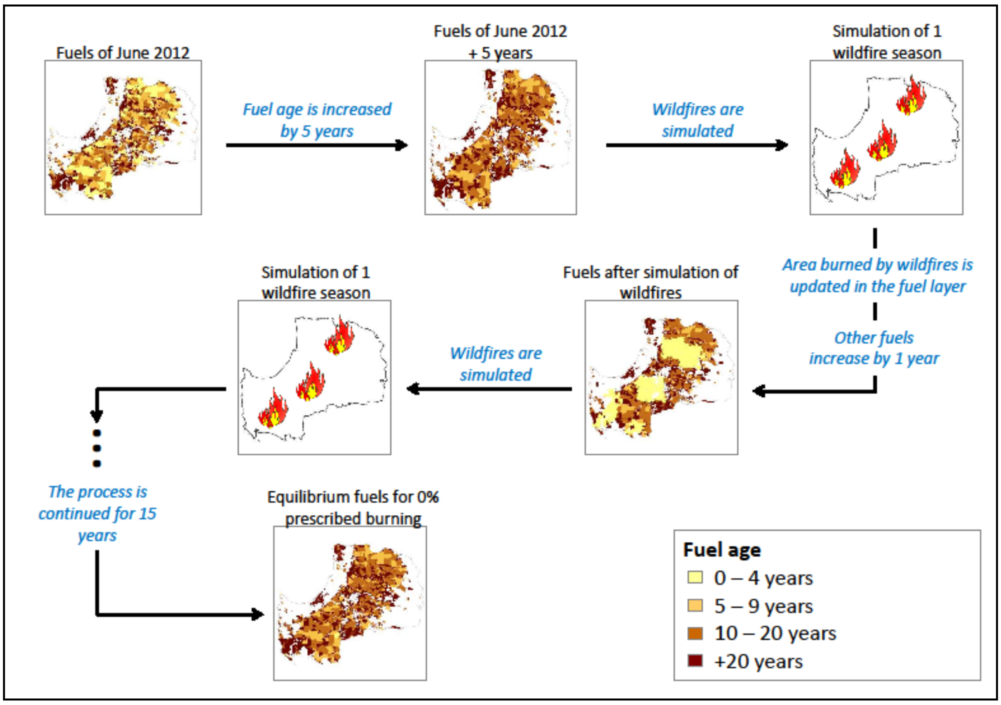

Figure 6.1. Creating an "equilibrium" fuel layer for a 0% prescribed-burning rate

6.2.2. Fuel layer for 5% prescribed burning

For the fuel layer that represents the long-term application of prescribed burning at a rate of 5% of DPaW-managed land per year, a slightly different process was followed to generate the "equilibrium" fuel layer. In this case, the starting fuel layer is different and prescribed burning is incorporated every year between fire seasons. If 5% of DPaW-managed land is prescribed burned every year and no area is treated twice in the space of 20 years, all of DPaW-managed land would be treated every 20 years. Thus, as a first step in the creation of the 5% "equilibrium" layer, the fuel-age attribute was modified to be between 0 and 20 years old for all fuels in DPaW-managed land in the region. A wildfire season was then simulated and the area burned by the wildfires updated on the initial layer. On this new layer, 5% of DPaW-managed land was prescribed burned, starting with the patches with the oldest fuel age. In the rest of the landscape the fuels were allowed to accumulate for one year. Then a new wildfire season was simulated and the fuel-age attribute was updated for the pixels burned with a value of zero. This process was repeated for 10 years. It was considered unnecessary to continue for more than 10 years because the starting point of this process was a landscape in which a 5% rate had been applied for 20 years (but without wildfires in each fire season). Since the effects of prescribed burning on wildfire behaviour are estimated to last between 4 to 7 years after the application of the treatment (Fernandes and Bothelo 2003; Boer *et al*. 2009; McCaw 2013), a 10 year simulation of fire seasons was considered sufficient to arrive at an "equilibrium" fuel layer. After the last wildfire season was simulated on year 10 and 5% of the area was again prescribed burned, the resulting layer was used to run the simulations of the 121,000 fires that generate the C+NVC results. Figure 6.2 shows the process used for generating the base "equilibrium" fuel layer for a 5% prescribed-burning rate in the long term. It is assumed that these simulations are a representation of what could happen in any one year in the future if 5% of DPaW-managed land is prescribed burned for at least 10 years prior.

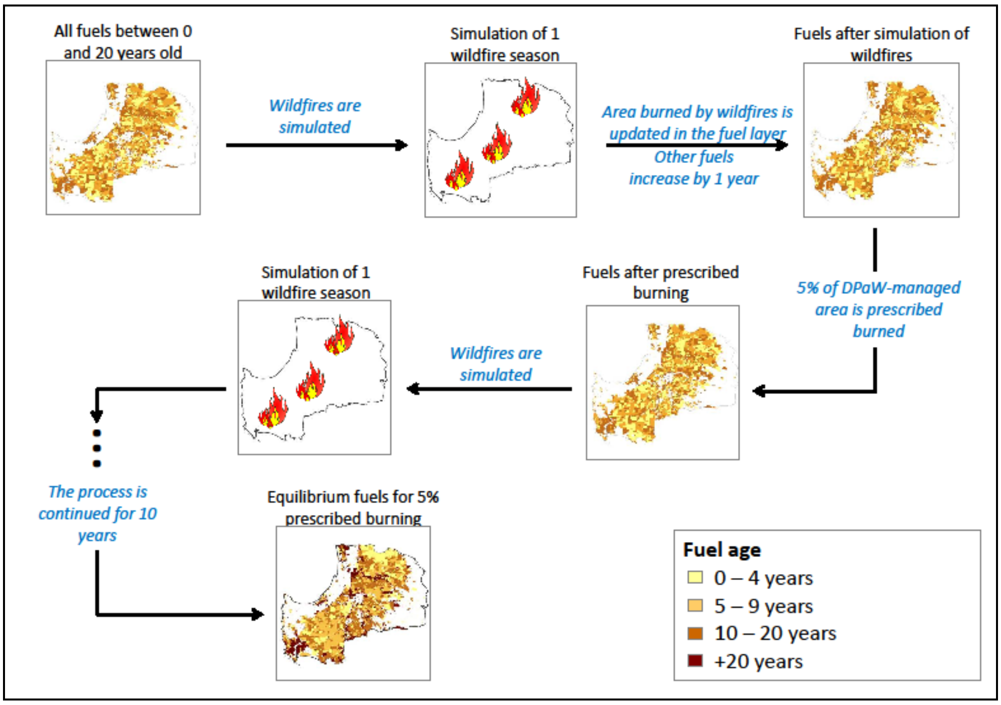

Figure 6.2. Creating an "equilibrium" fuel layer for a 5% prescribed-burning rate

6.2.3. Fuel layers for 10%, 15% and 20% prescribed burning

For the "equilibrium" fuel layers representing 10%, 15% and 20% prescribed-burning rates, the same process used for 5% was repeated here, but with a different starting point for each rate. For a 10% rate, the fuel-age attribute for all the fuels in DPaW-managed land was modified to be between 0 and 10 years old. For a 15% rate, it was modified to be between 0 and 6.7 years old and for a 20% rate between 0 and 5 years old. The same process of simulating an average wildfire season with different ignition points each year followed by prescribe burning was repeated for 10 years. The resulting "equilibrium" layers after the last application of prescribed burning in year 10 was used to simulate the 121,000 fires.

Prescribed-burning costs per year are calculated using the same parameter values that were used for the short-term analysis. It is acknowledged, however, that other factors could influence prescribed-burning costs per hectare in the long-term. For instance, lower fuel levels across a landscape would reduce the cost per hectare of prescribed burning. When the area to be treated has relatively low fuel levels and the neighbouring areas also have limited amounts of fuels, less resources are needed to apply the treatment and the risk of escape is reduced (Peter Gibson, personal communication 2015). However, for simplicity purposes this reduction in prescribed-burning costs per hectare is not included in this long-term analysis. Other costs, such as the additional resources that would be required in order to get from the current prescribed-burning rate to a higher one are not included in this analysis.

6.3. Results

The optimal prescribed-burning rate for the South West forest region in the long term is 15% of DPaW-managed land prescribed burned per year (see Table 6.1). Although the optimal rate is identical to the short-term optimal rate, the results differ significantly from the short-term analysis in two ways: (1) the annual sum of damages and suppression costs at 0% prescribed burning is much higher in the long term (adding up to nearly AU$257 million in the long term, compared to AU$61 million in the short term); and (2) the C+NVC curve is no longer flat between 0% and 20% and there is a substantial difference between 0%, 5% and 10% prescribed-burning rates (see Figure 6.3). The long-term analysis results in a C+NVC curve with the shape of an inverse exponential function. At low levels of prescribed burning, additional area prescribed burned produces a substantial reduction in the sum of costs and damages, and thus the curve has a steep negative slope. The slope of the curve diminishes with additional proportions of area prescribed burned, until the curve is visibly flat.

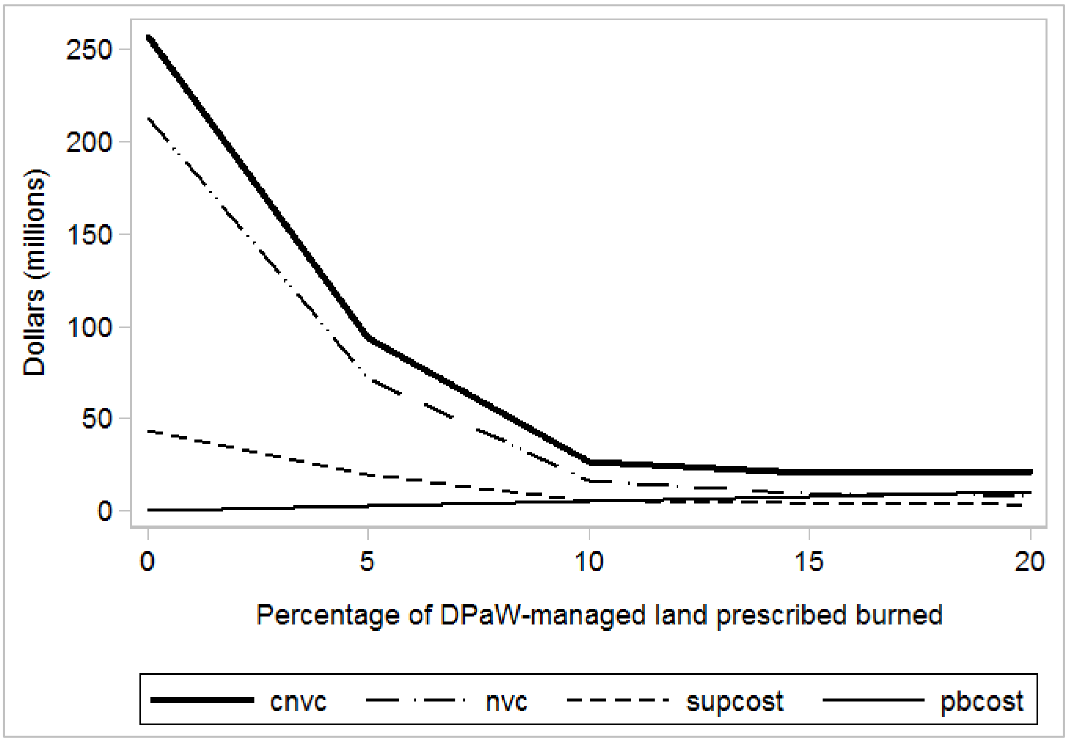

Figure 6.3. Cost plus net value change for different prescribed-burning rates applying the current strategy (long-term analysis)

Table 6.1. Results for different prescribed-burning rates applying the current strategy (millions of AU$ per year after the long-term application of each rate)

Rate	Prescribed-burning costs	Suppression costs	Damages	C+NVC	Suppression costs plus damages	Benefits (compared to 0%)	Net benefits
0%	0.00	43.88	213.05	256.93	256.93		
5%	2.58	19.78	71.60	93.96	91.38	165.56	162.98
10%	5.15	5.22	16.17	26.53	21.39	235.54	230.40
15%	7.72	3.95	9.18	20.86	13.13	243.80	236.08
20%	10.29	3.09	8.00	21.37	11.08	245.85	235.56

If prescribed burning is applied for long periods of time, it becomes a highly beneficial fire management activity. Compared to a no-prescribed-burning strategy (0% prescribed burning), long-term investments in prescribed burning in the region generate between AU$163 and AU$235 million in net benefits per year, 40 to 60 times more net benefits than indicated by the short-term analysis.[19] Investments in prescribed burning in the long run generate large benefits per dollar invested. At low levels of prescribed burning, additional treatments generate approximately AU$63 benefits per dollar invested every year. As the area treated increases, the benefits per dollar invested in prescribed burning are smaller and smaller, but even at a 20% rate, the benefits are still substantial (AU$ 23 per dollar invested per year). When the optimal rate is applied (15% of DPaW-managed land is prescribed burned), every dollar invested generates AU$31 in benefits per year. Thus the marginal returns from investments in prescribed burning follow the law of diminishing returns.

Compared to the short-term analysis, the long-term analysis shows a more clear-cut answer, suggesting that at a 0% or at a 5% prescribed-burning rate there are still substantial benefits that can be gained from increasing the amount of area prescribed burned per year. However, there are still several near-optimal options when a rate of 10% or higher is applied in the long term. The C+NVC results between 10% and 20% are relatively similar, and the superiority of a 15% rate over 10% or 20% is not marked. But the proportion of management costs vs. the proportion of damages between the different prescribed-burning rates is somewhat different, shifting from approximately 2/3 damages and 1/3 management costs at a rate of 10%, to about 1/3 damages and 2/3 management costs at a 20% rate.

Thus, in the long term there are also trade-offs between management costs and damages, but these trade-offs are considerably more apparent in the long term than in the short term for the current strategy. At low levels of prescribed burning, the vast majority of the costs arise from damages (see Table 6.2 and Figure 6.4). With additional area treated, the proportion of damages is progressively reduced until it is lower than the proportion of management costs. This is another valuable insight for fire managers in the region; it shows that on average damages can be substituted by management costs in the South West forest region as long as

[19] In all of the analysis that follows, we report the values at equilibrium. There is an additional issue of the discounted net present value associated with the transition to the equilibrium, and the optimal rate of transition to the equilibrium, but this would require a full dynamic simulation of prescribed burning under alternative weather seasons/ignition patterns to identify the expected C+NVC in each year in the transition period. The amount of computer simulation time required to achieve this meant that this is not considered here.

15% or more of DPaW-managed land is treated every year for a long period of time. Notice also that the proportion of damages is lower than the proportion of management costs when the investment in prescribed burning is higher than in suppression. This also highlights the importance of investigating the long-term dynamics of prescribed-burning investments, because the short-term analysis showed that this substitution was only minimal between the different prescribed-burning rates for the current strategy and that the proportion of damages was always higher than the proportion of management costs. In contrast, the long-term analysis shows that these proportions can be reversed.

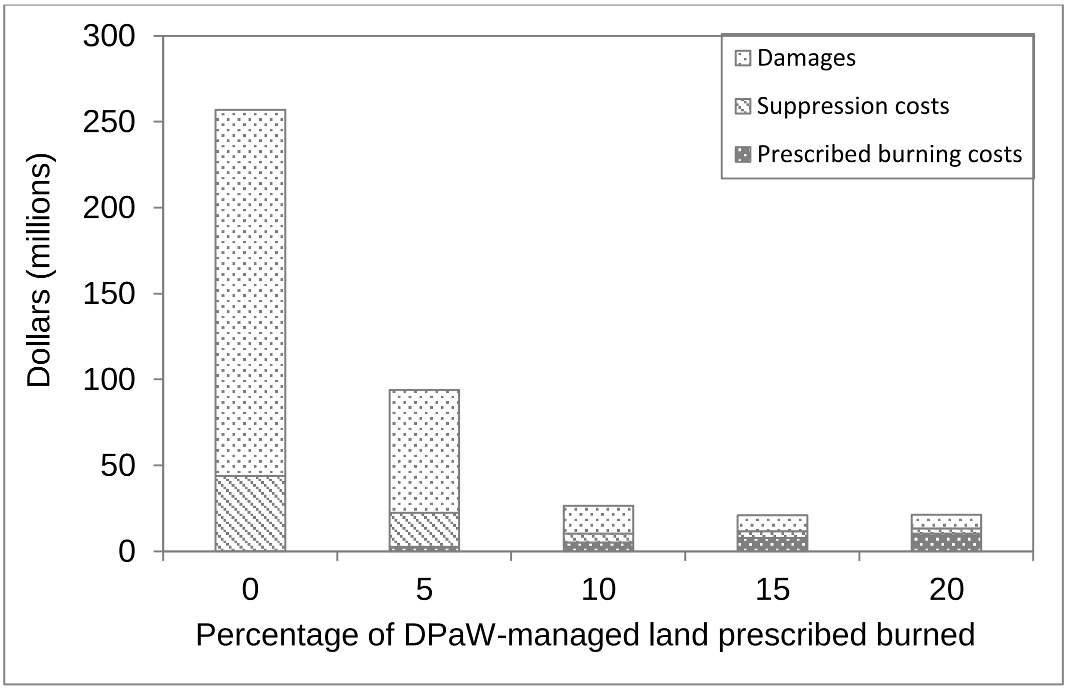

Figure 6.4. Damages, suppression costs and prescribed-burning costs for each prescribed-burning rate applying the current strategy: long-term analysis

Table 6.2. Proportion of damages, suppression costs and prescribed-burning costs for each prescribed-burning rate (% of total C+NVC for each prescribed-burning rate)

Rate	Prescribed-burning costs	Suppression costs	Damages	Management costs[a]
0%	0%	17%	83%	17%
5%	3%	21%	76%	24%
10%	19%	20%	61%	39%
15%	37%	19%	44%	56%
20%	48%	14%	37%	63%

[a] Prescribed-burning costs plus suppression costs

In the long term, prescribed burning significantly reduces wildfire area. However, the application of prescribed burning at any rate drastically increases the total area that is subjected to fire of any type (prescribed burning or wildfires) (see Table 6.3 and Figure 6.5). With no prescribed burning, wildfires would burn around 37,000 hectares on average per year in the South West forest region. With the application of prescribe burning at a rate of 10% for instance, average wildfire area is reduced to less than 4,500 hectares, but the total area subjected to fire is slightly more than 83,400 hectares, which is more than double the amount compared to a 0% prescribed-burning rate. For any application of prescribed burning ≥5% of DPaW-managed land in the South West forest region, the total area prescribed burned is always larger than the average area burned by wildfires (Figure 6.5).

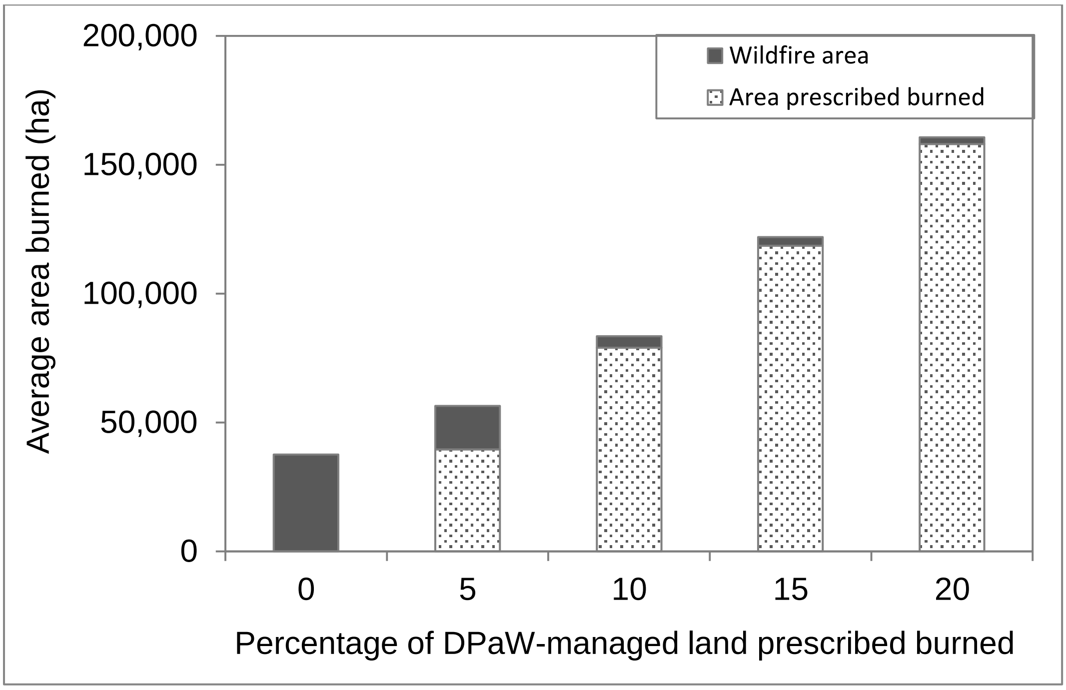

Figure 6.5. Total area subjected to fire of any type

As with the benefits per dollar invested, prescribed burning exhibits diminishing returns in area burned; that is, each additional hectare prescribed burned results in a smaller reduction in wildfire area. The results of the simulation model indicate that to reduce wildfire area by one additional hectare, between two and 60 hectares need to be prescribed burned (Table 6.3). When 5% of DPaW-managed land is prescribed burned, treatment leverage is 2:1, so to reduce wildfire area by one additional hectare, two more hectares need to be prescribed burned. At higher levels of prescribed burning, more and more hectares need to be prescribed burned in order to reduce wildfire area (Table 6.3) and the reduction in wildfire area achieved with more prescribed burning is very small. This decrease in the marginal reduction in wildfire area with additional area prescribed burned is clearly illustrated in Figure 6.6.

Table 6.3. Wildfire area, prescribed-burned area and leverage

Rate	Wildfire area (ha)	Area prescribed burned (ha)	Total area subjected to fire (ha)	Leverage[a]
0%	37,506	0	37,506	
5%	16,902	39,500	56,402	2
10%	4,459	79,000	83,459	18
15%	3,378	118,500	121,878	35
20%	2,637	158,000	160,637	60

[a] Leverage is calculated as the ratio of wildfire area over prescribed-burned area

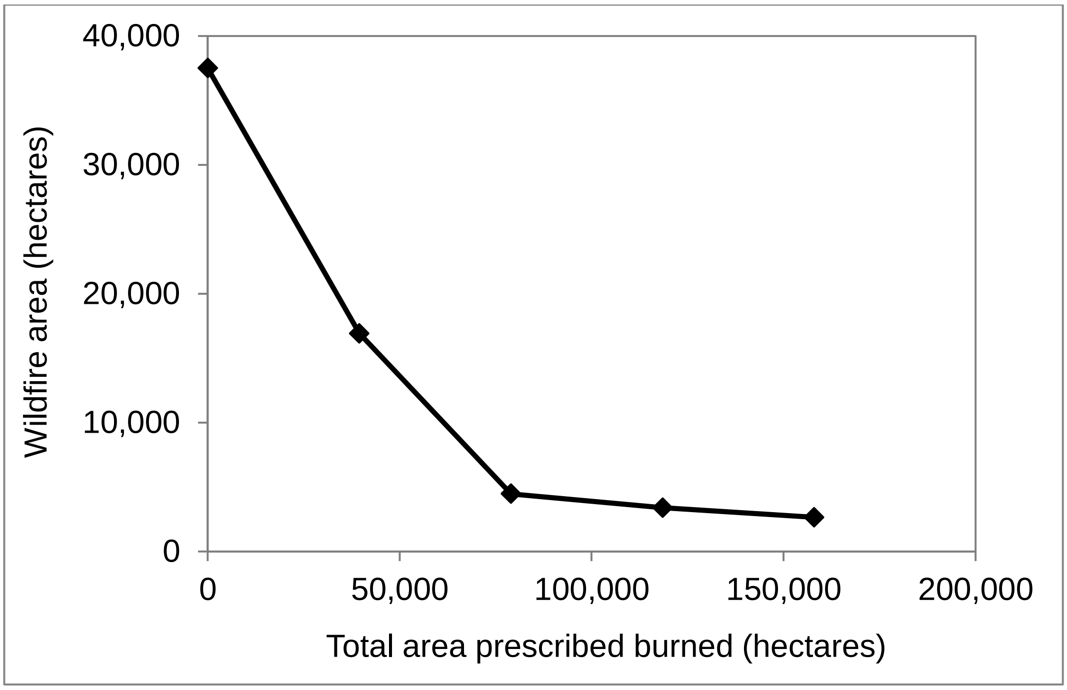

Figure 6.6. Treatment leverage

In the long term there is also a dichotomy between what is most frequently burned in the landscape and what generates the greatest economic damages. Similar to the short-term analysis, the most frequently burned areas are conservation areas and State forests (Figure 6.7). But because urban areas have the highest value per hectare, the vast majority of damages in dollars are the result of destruction of assets in urban areas. As a consequence, any reduction in area burned in the WUI significantly reduces the level of damages (Figure 6.8). With 0% prescribed burning, on average, 213 hectares are burned by wildfires in urban areas. This is reduced to 69 hectares when 5% of DPaW-managed land is prescribed burned, to 16 hectares when 10% is prescribed burned and to 7 hectares when 15% or more is prescribed burned. This reduction in the average area burned by wildfires in urban areas results in a significant reduction in urban area damages from AU$155.5 million for a 0% prescribed-burning rate, to AU$48.7 million for 5%, to AU$10.1 million for 10%, and to AU$4.7 million for 15% or more. Thus the reduction in damages in urban areas drives the substantial reduction in total damages.

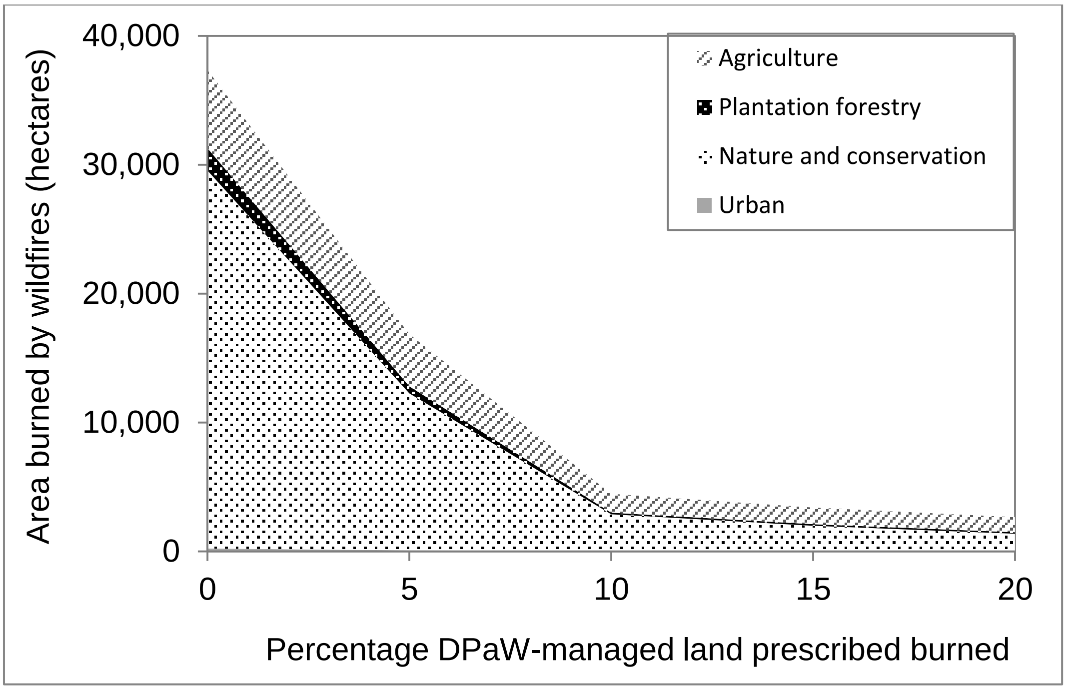

Figure 6.7. Area burned by wildfires per land-use type for different prescribed-burning rates

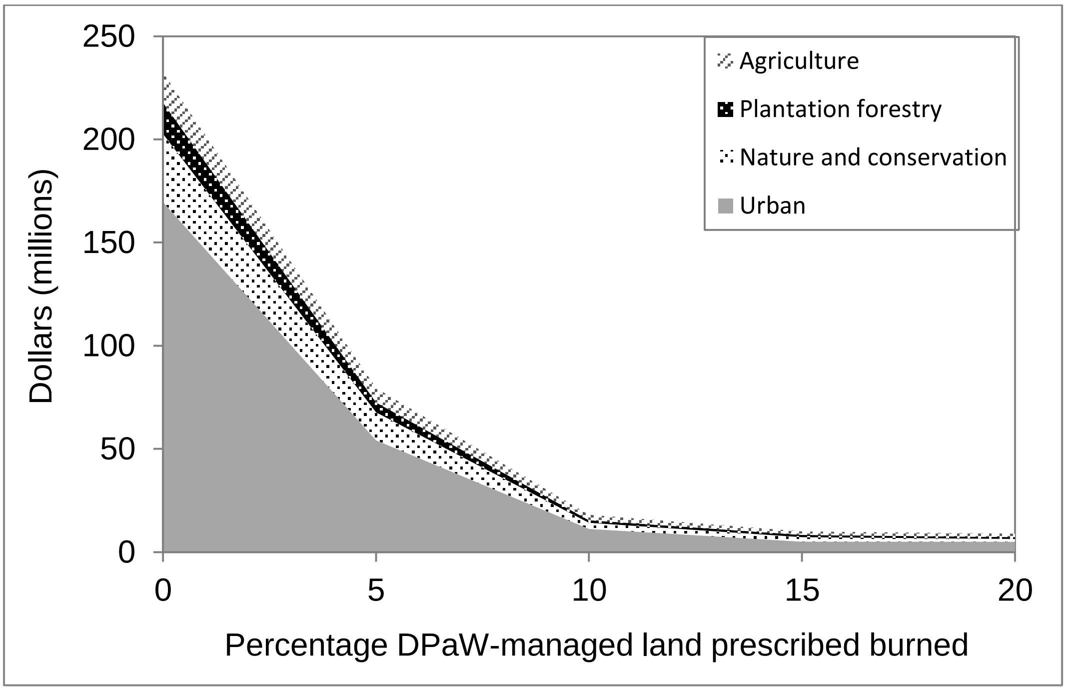

Figure 6.8. Wildfire damages per land-use type for different prescribed-burning rates

This shows the importance of maintaining a minimum level of prescribed burning to protect human assets in the South West forest region, even when the majority of the prescribed-burning treatments are located in forested areas and only a minority are placed close to high-value human assets (as is the case with the current strategy). The results of the long-term analysis clearly show that prescribed burning in forests far away from towns also protects human assets in urban areas. Even though most of the treatments are located more than 10km away from any town in the South West forest region (60% of the total area treated for any prescribed-burning rate is located more than 10km away from towns), this strategy still generates a significant reduction in damages in urban areas.

6.4. Sensitivity analysis

A sensitivity analysis was conducted for the long-term model using the same approach that was used for the short-term analysis (i.e. each parameter was changed independently while all others were left unchanged and all parameter values were increased and decreased by 50%).

The sensitivity analysis shows that the results of the long-term analysis are robust. The optimal prescribed-burning rate does not change with changes to the different parameters of +50% and -50% (Table 6.4). The only exception is when prescribed-burning costs are reduced by 50%, in which case the optimal rate increases to 20%. The C+NVC at the optimal rate changes with different parameter values, but the change is relatively small. The C+NVC ranges between AU$16 million to AU$23 million, which corresponds to a change of -20% and +15% respectively from the baseline result.

Table 6.4. Sensitivity analysis of the current strategy (long-term analysis)

	Optimal rate (% DPaW-managed land prescribed burned)		C+NVC (million AU$)	
Baseline results	15		20.86	
	Change in parameter			
	-50%		+50%	
Parameter	Optimal rate	C+NVC (AU$ millions)	Optimal rate	C+NVC (AU$ millions)
Urban areas	15	18.68	15	23.04
Conservation areas and State forests	15	19.84	15	21.87
Plantation forestry	15	20.35	15	21.37
Agricultural (grazing, cropping)	15	20.26	15	21.45
Agricultural (vine fruits)	15	20.56	15	21.15
Prescribed-burning costs	20	16.23	15	24.72
Suppression costs	15	18.88	15	22.83
Level of damage (depending on the level of intensity)	15	16.27	15	21.35
Probability of large fires occurring	15	19,75	15	25,49

In the same way that the exponential increase in prescribed-burning costs was explored in the short-term analysis, here too an exponential increase is considered for the long-term model.[20] When prescribed-burning costs are increased exponentially, the optimal long-term prescribed-burning rate remains the same (15%). Even with an exponential increase in prescribed-burning costs, there are still substantial benefits generated per dollar invested, which in this case range between AU$10 and AU$47. In addition, the shape of the C+NVC curve is similar to the baseline results, but also somewhat closer to the U shape of the theoretical model with a slightly steeper increase between 15% and 20% (Figure 6.9).

[20] For a detailed explanation of the rationale for an exponential increase in prescribed-burning costs, see Chapter 5, Section 5.3.1.

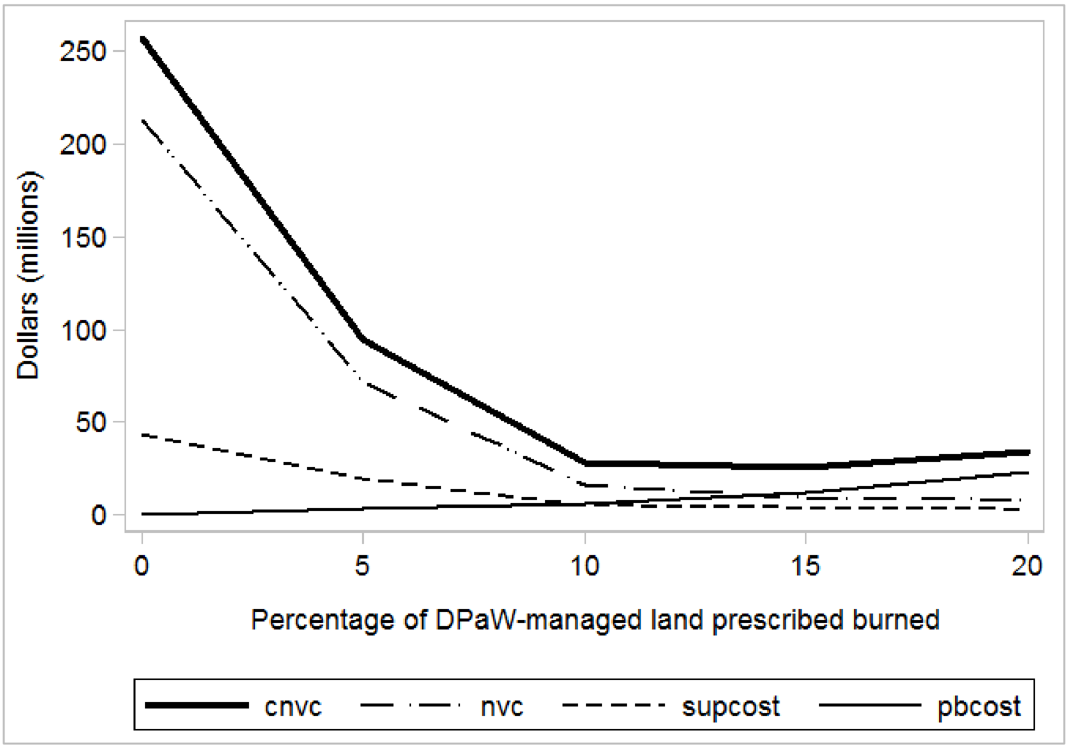

Figure 6.9. Cost plus net value change for different prescribed-burning rates with exponential increase in prescribed burning costs (current strategy, long-term analysis)

The long-term application of prescribed burning may generate other costs that would make this exponential increase more pronounced. For instance, prescribed burning may result in additional health costs due to increased exposure to smoke and biodiversity losses due to a more frequent fire interval. However, since valuation data for these costs is not available for the south-west of WA, they have not been included in the exponential function.

The long-term model was run again using the same value per hectare for all land-use categories, in order to test how similar or how different the results of this approach are in comparison to the baseline results presented in Section 6.3. Figure 6.10 shows the results of the C+NVC using common values for all hectares. Compared to Figure 6.3 in Section 6.3, Figure 6.10 below shows that when a single value for all hectares is used the shape of the C+NVC curve is very similar, decreasing rapidly between 0% and 10% and then flat between 10% and 20%. The model using common values for all hectares suggests that it is beneficial to increase the rate of application of prescribed burning to even higher levels (to 20% of DPaW-managed land instead of 15%). Furthermore, with the value selected for all hectares (AU$8,600 per hectare), it overestimates the benefits of prescribed burning in the long term. The average damages resulting from 0% prescribed burning are AU$44 million higher than if the model is run with different values per hectare for different land-use categories. This is a valuable insight; it shows the importance of differentiating between the values of the assets protected. If this difference in values for different land-use categories is not taken into account, prescribed burning may be thought to yield more benefits in the long term than it actually does (or less, depending on the value selected for all hectares).

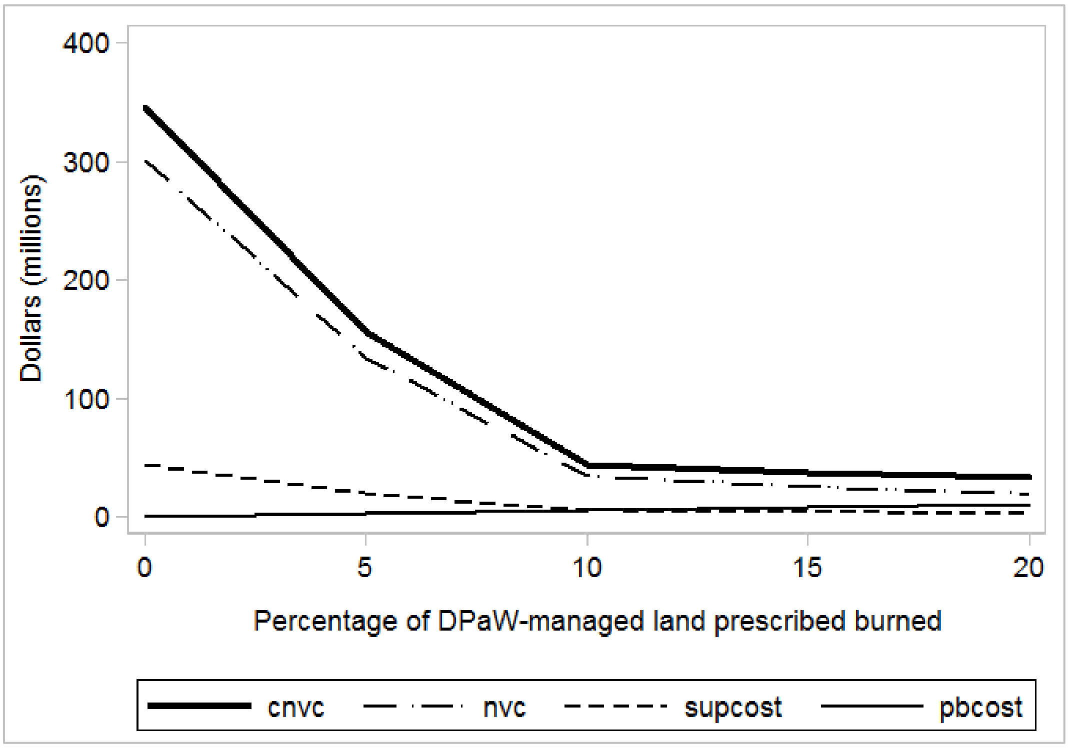

Figure 6.10. Cost plus net value change for different prescribed-burning rates using a single value per hectare for all hectares (current strategy, long-term analysis)

6.5. Comparison of the short-term and the long-term analyses

A comparison between the short-term and the long-term analyses may be useful at this point in order to appraise the full consequences of not taking into account the long-term effects of prescribed burning in the South West forest region. From the results of the long-term analysis, it is clear that the short-term analysis significantly underestimates the benefits of the treatment. Only when the long-term dynamics of fuel accumulation are incorporated in the model can the true costs of not doing any prescribed burning for a long period of time be revealed. In the short term, the 0% prescribed-burning rate (or any other rate) includes the benefits of the application of the treatment in previous years, thus the results are similar to evaluating a 7% prescribed-burning rate in the long term. This explains why the results of a short-term analysis display a flat C+NVC curve: all points in the curve are close to the result that would be obtained for a long-term analysis of the rate applied in previous years. This can be clearly observed in Figure 6.11. The C+NVC curves for the long-term analysis and for the short-term analysis meet approximately at the average level of prescribed burning that has been applied during the last 10 years.

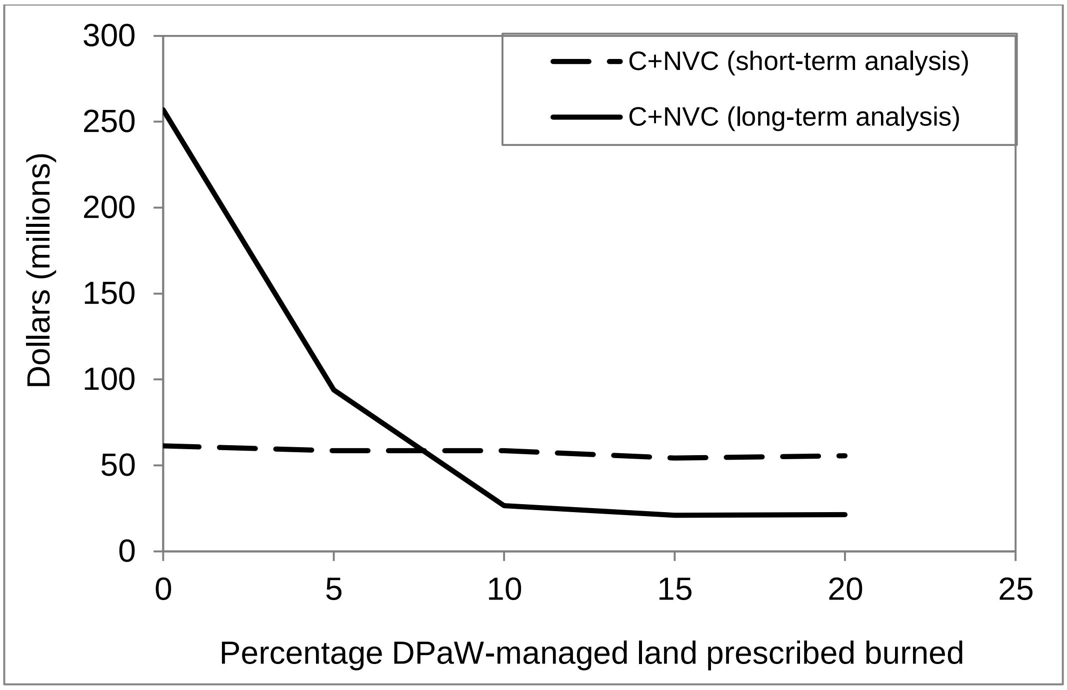

Figure 6.11. Cost plus net value change for different prescribed-burning rates applying the current strategy (short-term analysis vs. long-term analysis)

The full long-term benefits of prescribed burning in terms of average damages avoided and reduced suppression expenditure are considerably higher than those obtained with the short-term analysis. Figure 6.12 illustrates the benefits generated for different levels of investment in prescribed burning in the long term and in the short term. In the long term, each dollar per year invested in prescribed burning generates between AU$23 and AU$63 benefits every year compared to AU$1.5 to AU$2 in the short term. Because the results of the short-term analysis are significantly influenced by what has happened before in the landscape studied, it cannot be used to determine the strategic optimal rate of prescribed burning for the region. The short-term analysis can only be used as a source of information in preparation for the upcoming fire season, but not as a strategic decision making tool. Any decisions regarding the area to be treated or the funds required in any given year must be part of a long-term analysis. This is a very important message for an activity that is highly politicised and influenced by recent wildfire events and community pressures in Australia (Burrows and McCaw 2013). Taking a short-term perspective on this issue will limit the information available for decision making because it does not provide a complete picture of the costs and benefits of the treatment. Each dollar invested in prescribed burning in any one year produces benefits for several years after, which can only be captured in a long-term analysis.

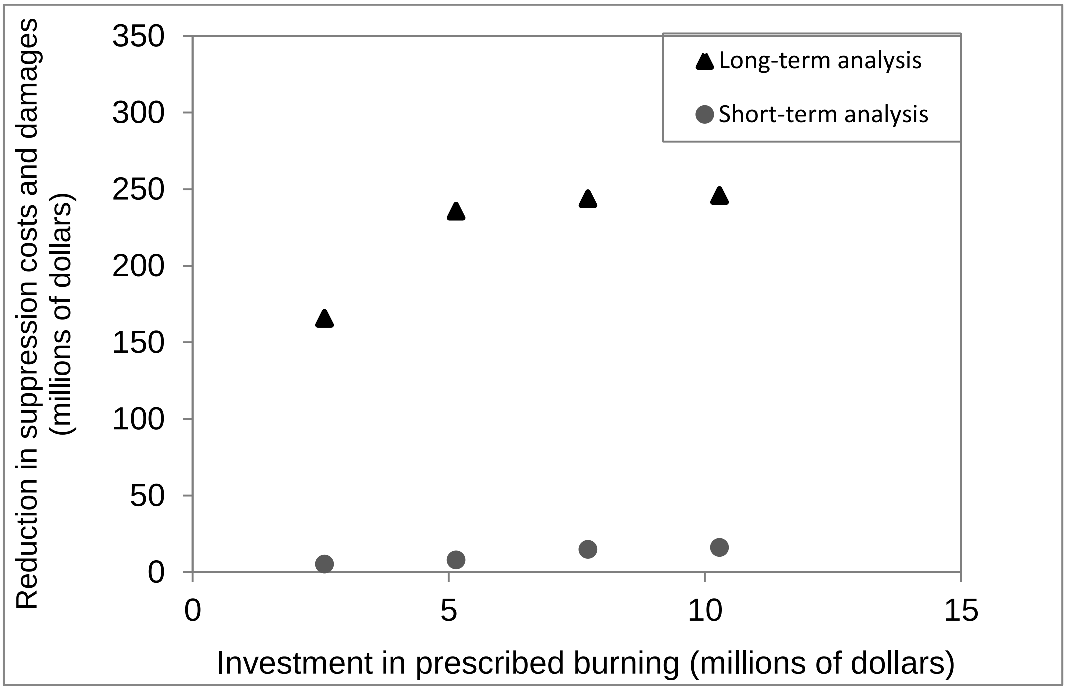

Figure 6.12. Reduction in suppression costs and damages compared to no-prevention (0% prescribed burning) for different levels of investment in prescribed burning (short-term analysis vs. long-term analysis)

6.6. Discussion

The optimal rate for the long-term model in this study points to a relatively high level of prescribed burning (15% of DPaW-managed land). However, this is not an uncommon result in the economics of fuel treatments. Other studies that have investigated the long-term costs and benefits of fuel reduction treatments, including prescribed burning, have also suggested the extension of these treatments to relatively high levels (Snider *et al*. 2006; Mercer *et al*. 2007; Butry *et al*. 2010). For instance, Mercer *et al*. (2007) suggested an increase in prescribed burning from 4.8% to 13% per year for the forested lands in the Volusia County (Florida, US) in order to efficiently minimise the economic impact of wildfires in the County. In another study, Snider *et al*. (2006) recommended the implementation of fuel reductions at a rate of 15% per year for 1/3 of the ponderosa pine and dry mixed-conifer forests of New Mexico and Arizona in the US, based on the amount of savings in suppression costs that can be achieved from fuel reduction treatments. Other authors have suggested a more moderate increase in fuel reduction treatments to 4% per decade in the boreal forests of north-eastern China and recommended a combination of mechanical fuel treatments and prescribed burning (e.g. Liu *et* *al*. 2010). However, Liu *et al*. (2010) analysed the cost-effectiveness of different combinations of treatments in reducing area burned by wildfires instead of reducing suppression costs and/or damages. Furthermore, since previous studies have used a common value for all hectares in their models and it has been shown in this Chapter that this may lead to overestimation of the benefits of prescribed burning (or underestimation depending on the value per hectare used), it may be possible that prescribed burning results in less benefits in other landscapes studied in the literature (or more benefits if the value per hectare used is relatively low).

When the long-term effects of prescribed burning are analysed, the number of prescribed burning hectares needed to reduce wildfire area by one hectare (i.e. treatment leverage) is similar to that obtained from historical records in other forested areas in the south-west of WA and in eastern Australia. For the average prescribed-burning rate that is currently applied in the South West forest region (i.e. between 6% and 7% of DPaW-managed land per year) the estimated treatment leverage is between 4:1 and 5:1 (i.e., four to five hectares of prescribed burning to avoid one hectare of wildfire). This result is close to what Boer *et al*. (2009) estimated for the Warren region (4:1), an area also located in the south-west of Western Australia to the east of the South West forest region, which has been treated at a rate of 5 to 10% of DPaW-managed land per year. Another similar result is found in Price and Bradstock (2011), who estimated a leverage of 3:1 for the forested parts of the coast and mountains around Sydney, which had been burned by hazard reduction burns at a rate of 2.75% per year on average. This shows a high level of consistency between the different studies on the capacity of prescribed burning to reduce wildfire area, but what is more interesting from an economic point of view is the capacity of prescribed burning to reduce wildfire damages and suppression costs. In this regard, there is less consistency in the literature.

In terms of the benefits gained per dollar invested in prescribed-burning programs, the existing economic analyses on fire management programs rarely report on these values. Furthermore, the values reported are calculated differently and cannot be easily compared with the results from this study. For instance, Benetton *et al*. (1998) estimated a benefit:cost ratio of 52:1 for the Fire Management Program (FMP) of the Victorian Department of Natural Resources and Environment. However, these benefits were calculated from the Department's investment in several fire management activities, not solely for prescribed burning. The authors included the benefits of investing in prescribed burning, suppression and education activities in their calculations. Nonetheless, their results are comparable to the values reported in this Chapter, which indicated that long-term investments in prescribed burning may generate between AU$23 and AU$63 benefits per dollar invested every year. In another study, Butry (2009) estimated a benefit: cost ratio of US$1.53:1 for a prescribed-burning rate of slightly less than 2.3% in the St. Johns River Water Management District (in the northeast of Florida, US), but the benefits correspond to damages avoided only and do not include reductions in suppression costs. If only the reduction in wildfire damages was considered for this study, for every dollar invested in prescribed burning, a damage of AU$55 would be avoided on average per year if a rate of 5% is applied in the long term. The results of Butry (2009) are substantially lower compared to the results of this study. This difference may be explained by the divergences in land-use mixes between the two areas and by the fact that Butry (2009) used a common value for all hectares in the landscape.

In any case, the benefits of prescribed burning stem primarily from the reduction in damages, rather than the reduction in suppression costs. The literature examining fuel treatment impacts on fire suppression cost savings has generally found that savings in suppression costs are unlikely to offset fuel treatment costs (Thompson and Anderson 2015). In this Chapter, we have shown that it may or may not be the case, depending on the level of prescribed-burning currently applied. If a 0% or a 5% prescribed-burning rate is applied, increasing the level of prescribed burning generates suppression costs savings that are higher than the increase in prescribed-burning costs. However, if higher levels of prescribed burning are applied (i.e. 10% or more), savings in suppression costs are not offset by the increase in prescribed-burning costs.

The results presented in this Chapter indicate that there are substantial benefits to be gained from increasing the application of prescribed burning in the South West forest region. They are consistent with the general recommendations of the existing fire economics literature. Indeed, there is a general agreement (implicit or explicit) in the fire economics literature that suggests that a larger proportion of management funds should be used for fuel treatments rather than only focusing on suppression and continuing the increase in suppression expenditures (Snider *et al*. 2006; Butry *et al*. 2010; Stockmann *et al*. 2010a; Bar Massada *et al*. 2011). However, the specific type of fuel reduction treatment that is most appropriate for a region and the optimal rate of application will vary from region to region and cannot be generalised to other areas. The specific attributes of a landscape, such as the land-use mix, topography, vegetation type and fire history, may influence fire behaviour (Fernandes and Bothelo 2003) and consequently affect the results. Thus any quantitative economic analysis of fuel reduction treatments is only valid for the area studied. The results for the South West forest region cannot be generalised to other areas in the south-west of WA or elsewhere, because the topography, land-use mix, vegetation type and fire history are different.

This study and other economic analyses of fuel reduction treatments have made two points clear: (1) if no treatment is applied, damages and suppression expenditure would be considerably higher (Snider *et al*. 2006; Mercer *et al*. 2007; this Chapter); and (2) the full benefits of prescribed burning and other fuel reduction treatments can only be realised when the treatment is applied for long periods of time (Mercer *et al*. 2007; Butry *et al*. 2010; this Chapter). These two points have major implications for the strategic use of fuel reduction treatments: (1) in order to protect valuable human and environmental assets from wildfires, a minimum level of prescribed burning has to be maintained in the long term. The most appropriate level will depend on the decisions made in accordance with the numerous trade-offs inherent in fire management issues. And (2) because fuel reduction treatments are a long-term solution, they offer little flexibility from year to year. For this reason, some authors have suggested the use of prescribed burning as a strategic long-term management option and use other fire management activities, such as wildfire prevention education, as more effective short-term option (e.g. Butry *et al*. 2010) because education campaigns can reduce the potential number of accidental ignitions in the short term.

6.6.1. Implications for wildfire management in the long term in the South West forest region

This study emphasises the importance of keeping a minimum level of prescribed burning per year in the South West forest region. The long-term analysis shows that not doing any prescribed burning for several years can be very costly for the region, with large increases in damages and in suppression expenditures. If no prescribed burning was applied in the South West forest region, average annual suppression expenditure would be around 4 times higher than the current level and average damages would be around 5 times higher. With the current prescribed-burning program that DPaW have in place, applying the treatment to an average of 6% to 7% of their managed land in the region, they are already generating substantial benefits for the community living in the area and for the government. On average, the current prescribed-burning program generates AU$31 million savings per year in suppression expenditures and AU$169 savings in damages compared to a no-prescribed-burning scenario.

Since the full benefits of prescribed burning can only be realised when it is applied in the long term, the frequent and often substantial changes in the prescribed-burning rate applied in the South West forest region may result in some of the long-term benefits of the treatment being lost. Decisions over the prescribed-burning rate applied in the region have been influenced in the past by catastrophic wildfire events, which often result in an increase in the area treated, or by escaped prescribed burns, which lead to a reduction in the area treated, particularly for the treatments located close to towns (DEC 2012). This reduces the capacity of previous investments in prescribed burning to generate the full benefits of past treatments; thus some the investments done in the past may have lost their value.

The results in this Chapter indicate that there are additional potential benefits to be gained from increasing the amount of area that is prescribed burned per year, but this will come at an additional cost in terms of additional resources and other probable costs that are not accounted for in this analysis, such as increased health costs from smoke exposure and biodiversity losses. In addition, communities living in fire-prone areas would need to increase their acceptance of prescribed burning, recognise that it is not a risk-free activity and that there is a chance of escaped treatments, but also acknowledge that prescribed burning does not eliminate the risk entirely so there will always be a residual wildfire risk (Gill and Stephens 2009; Bradstock *et al.* 2012b; Calkin *et al.* 2014; this Chapter).

The optimal prescribed-burning rate (15%) suggested by the long-term model is higher than current practice. Although this high level of prescribed burning was applied in the 1960s and 1970s in the region, in today's context it may be not be possible to achieve it. Climate change, past fuel accumulation and fire exclusion policies, and the expansion of the WUI, have all contributed to create a landscape in which the application of high rates of prescribed burning is increasingly difficult (Chapter 3).

6.6.2. Implications for the design of economic analyses of fire management programs

The results in this Chapter demonstrate that to evaluate the full benefits of prescribed burning, the long-term dynamics of fuel reductions and fuel accumulation processes have to be incorporated in the model. This study has developed a way to include these dynamics in the C+NVC model, which is a model that does not inherently account for the long-term effects of fire management activities.

The results here show that the C+NVC model run for a single year is unlikely to provide reliable information on optimal rates of prescribed burning and optimal funding, because the long-term benefits of the treatment are large. In other words, short-term economic evaluations of changes to prescribed-burning programs may be inadequate because the results will be influenced by the history of prescribed burning in the landscape and cannot fully reflect the impacts of those changes. A short-term analysis may indicate that if no prescribed burning is applied in a given year it does not have negative consequences because the result is relatively similar to the application of other rates of prescribed burning. Therefore, strategic decisions regarding the optimal area to be treated and the optimal prescribed-burning investment are better evaluated using models that represent the long-term benefits of prescribed burning.

Another important point highlighted by this study, other fire economics studies and fire behaviour studies in the literature is that wildfire risk can never be entirely eliminated; there will always be some residual risk (Gill 2005; Mercer *et al*. 2007; Butry 2009; Butry *et al*. 2010; Bradstock *et al*. 2012). Prescribed burning and other fuel reduction treatments partially mitigate wildfire risk in forested landscapes but cannot stop wildfires from occurring. Even at very high levels of prescribed burning (i.e. 20% of DPaW-managed land in the South West forest region), the average level of damages that can be expected in any one year after the treatment has been applied for long periods of time can reach AU$8 million. In order to reduce wildfire risk to lower levels, the rate of prescribed burning applied in the long term would have to be higher than the rates tested in this study (i.e. >20% DPaW-managed land). As Bradstock *et al*. (2012) pointed out, “this prospect raises questions for communities about the level of risk that may be acceptable and their willingness to pay to achieve it” (p. 73). The level of risk that the communities in the south-west of WA are willing to accept and their willingness to pay for it are two important pieces of information that fire managers and decision makers also need to make strategic decisions about prescribed burning, but to this date they remain largely unknown for the south-west of WA and other regions in Australia.

The following Chapter provides some information relevant to this issue by looking at the distribution of the annual cost and benefits, rather than just the means as it was reported in Chapters 5 and 6.

Chapter 7.

RISK AND SKEWNESS IN A LONG-TERM ECONOMIC ANALYSIS OF PRESCRIBED BURNING USING THE C+NVC MODEL

7.1. Introduction

In the literature examining the long-term economic effects of prescribed burning, most studies use a measure of central tendency (usually the mean) to present the results (e.g. Mercer *et al*. 2007; Butry *et al*. 2010). Given the stochastic nature of the simulation models used, there also exist other moments of the distribution of results, but these are not analysed or reported, despite potentially providing valuable information to fire managers. The distribution of the results for a prescribed-burning rate that is applied in the long term may have implications for management and for decision making about the protection of valuable assets. The same way that examining the effects of prescribed burning on wildfire area alone is only part of a bigger picture, looking at the mean of the economic results without looking at their distribution is only part of a comprehensive analysis. The variance of the results and the skewness of the distributions can also put a new light on what different levels of investment in prescribed-burning programs can do to protect people, human assets and the environment from wildfire threat. Moreover, the mean is only relevant if fire managers are risk neutral, but if they are risk averse, then it is important to look at the distributions. Information about the distribution of results may be used to make decisions in accordance with the maximum level of risk that society is willing to bear, which is not possible if only the mean of the results is reported.

As the mean of the C+NVC is reduced with additional area prescribed burned, we may expect the variance of the distribution of the results to decrease as well. Since this has never been explored before, it needs to be confirmed by simulation modelling in order to determine whether and by how much prescribed burning can reduce the variance of the distribution. In addition, analysing the distributions would provide information about the number of years between fire seasons of a particular severity that can be expected in the South West forest region, given the amount of prescribed burning that is applied in the long term.

Note that the results presented here are derived from the same set of simulations that generated the results in Chapter 6. The figures shown in the following sections represent the distributions of the observations from the simulated fires for each prescribed-burning rate (i.e. 220 fire seasons for each rate of prescribed burning).

This Chapter is organised as follows. Section 7.2 presents the results of the distributions for the C+NVC for different prescribed-burning rates. Section 7.3 has a closer look at the bulk of the distribution (i.e. excluding the tails) in order to examine them in more detail. Section 7.4 analyses the distribution of damages, suppression cost and area burned by wildfires. Finally, Section 7.5 discusses these results.

7.2. The distribution of the C+NVC results

Simulation modelling confirms that when the area prescribed burned is increased, the mean and the variance of the distribution of the C+NVC results are both reduced (see Figure 7.1 and Figure 7.2). The box indicates the 25th to 75th percentiles while the lines indicate the full range of results. The circle indicates the mean. In Figure 7.1 it is clearly visible that the variance of the C+NVC is much greater for prescribed-burning rates between 0% and 10% than it is for 15% and 20%. The box and the whiskers for 0% prescribed-burning rate have much wider ranges than for any other rate. The box for the C+NVC for a 0% prescribed-burning rate ranges from AU$6.7 million to AU$165 million per year, while the whiskers extend down to AU$1.7 million and up to AU$1,680 million. It is also apparent that the distribution of results for all prescribed-burning rates exhibit positive skew, i.e. a long right-handed tail with the mass of the distribution concentrated on the left (see Figure 7.2 and the measure of skewness shown in Table 7.1, which is >0 for all prescribed-burning rates) and that all distributions are heavy tailed (i.e. kurtosis >3).

For low levels of prescribed burning, the mean is noticeably separated from the median because of the high value of some of the observations. In comparison, the box for 5% extends from AU$10 million to AU$72 million. The results for 10%, 15% and 20% prescribed-burning rates are much more concentrated around a smaller range of values, with all of the boxes falling within a range of AU$10 to AU$23 million. The full shape of the distributions can be better observed in Figure 7.2, which shows the distribution of the C+NVC results with a width for the histogram bars of AU$5 million.

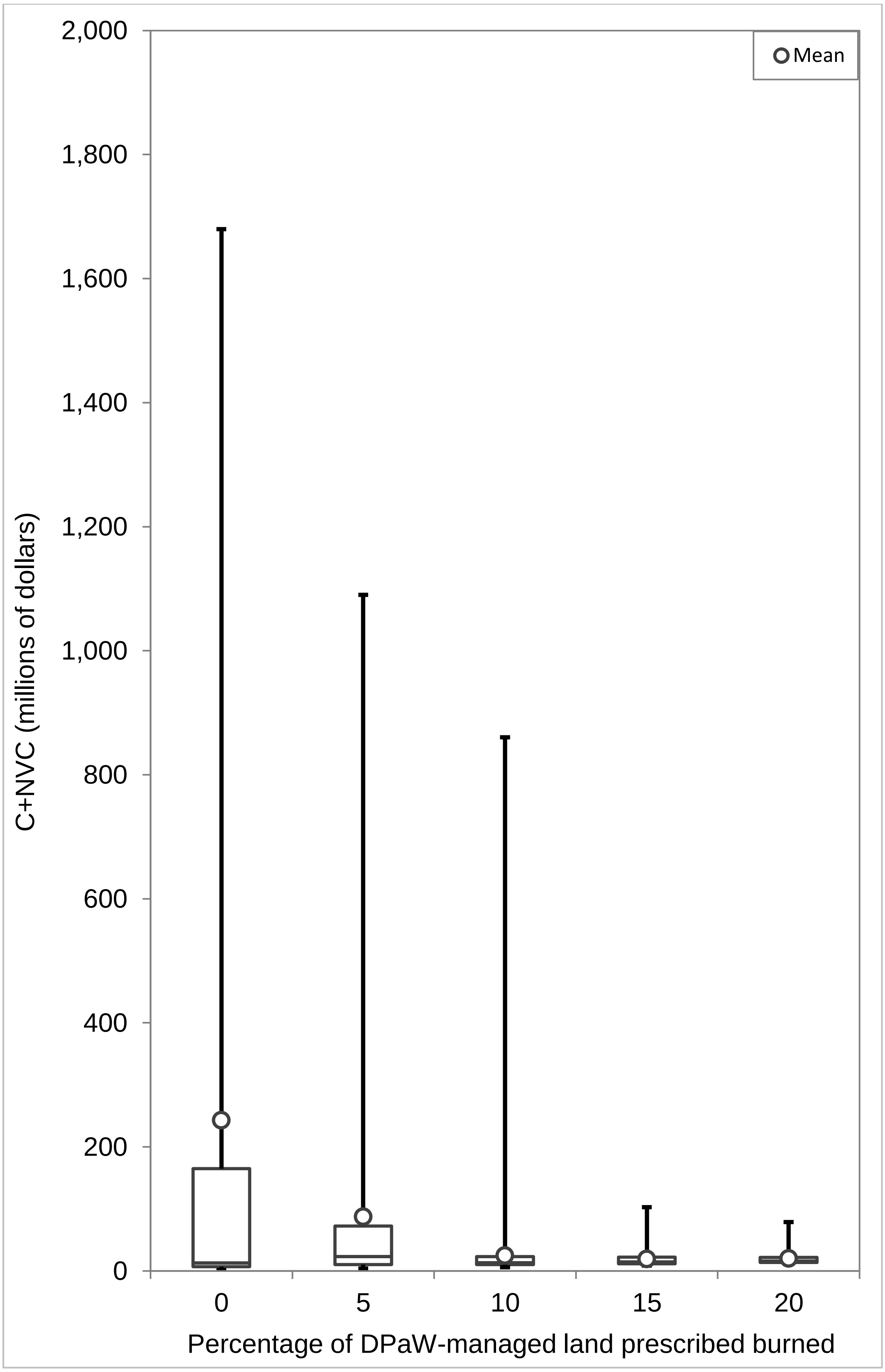

Figure 7.1. Range of the C+NVC results for different prescribed-burning rates.

Each box extends from the 25th to 75th percentiles and the whiskers extent to the smallest and the largest values of the C+NVC results for each prescribed-burning rate. The line inside each box represents the median and the white circle with a grey outline indicates the mean.

The 10% prescribed-burning rate shows an interesting result. Although the box in Figure 7.1 is relatively small and approximately the same size as for 15% and 20% prescribed-burning rates, the top whisker is considerably longer for 10%. The kurtosis is much higher than for other prescribed-burning rates (see Table 7.1 below), which suggests that more of the variance is explained by a low number of extreme deviations–instead of a high number of small deviations. In the results from the simulations, around 5% of the observations for the 10% prescribed-burning rate fall between AU$72 and AU$860 million. This suggests that approximately every twenty years, a fire season that may cost in total between AU$72 and AU$860 million in management costs and damages can be expected to occur in the South West forest region, even if a 10% prescribed-burning rate is applied in the long- term. Thus, despite substantially reducing the average sum of management costs and damages compared to a 0% or to a 5% rate, the application of a 10% prescribed-burning rate still has a small risk of very bad outcomes – much more so than the 15% and 20% prescribed-burning rates, even though the average C+NVC is relatively close for 10%, 15% and 20% rates.

The C+NVC results for 15% and 20% prescribed-burning rates have distributions with a considerably smaller variance. However, note that even at these high levels of prescribed burning, there is still a chance of having a relatively costly fire season with substantial management costs and damages–about 25% of the C+NVC results fall between AU$23 and AU$100 million (Figure 7.1 and Figure 7.2). Even so, as pointed out in Chapter 6, at these levels of prescribed burning, the majority of the costs are management costs.

Table 7.1. Mean, standard deviation, skewness and kurtosis for different prescribed-burning rates applying the current strategy in the long term[21]

Rate	C+NVC mean (AU$ millions)	Std. Dev. (AU$ millions)	Skewness	Kurtosis
0%	256.93	444.06	1.72	4.54
5%	93.96	182.80	3.67	17.45
10%	26.53	61.20	12.03	164.56
15%	20.86	15.92	3.04	14.30
20%	21.37	13.45	2.88	11.85

[21] Standard deviation, skewness and kurtosis were calculated using STATA version 13 (StataCorp 2013). The measure of skewness for a symmetric distribution, such as the normal distribution, is zero. The kurtosis for a normal distribution is three.

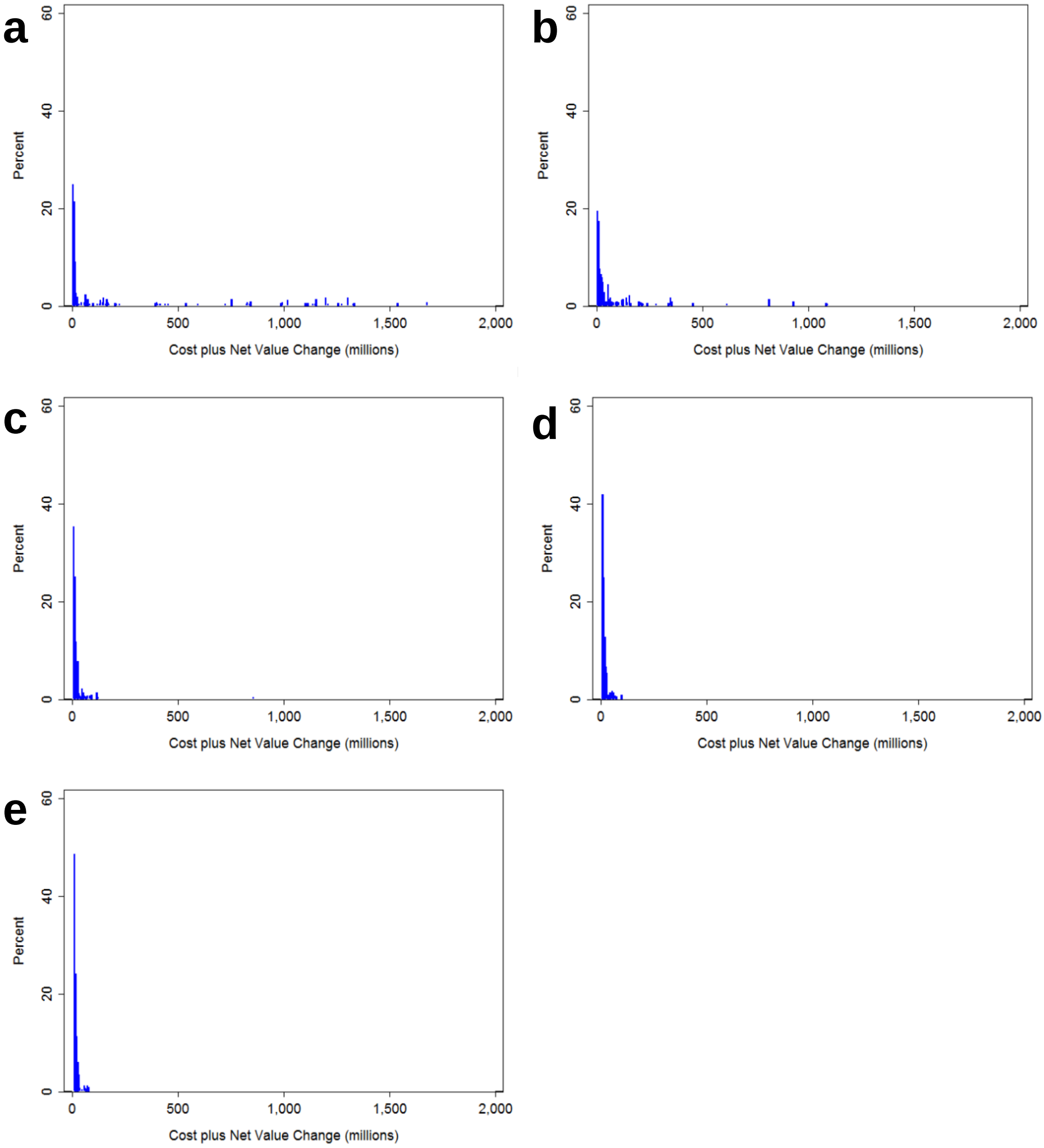

Figure 7.2. Distribution of the C+NVC results for each prescribed-burning rate

a. 0%, b. 5%, c. 10%, d. 15%, and e. 20%

The difference in the skewness of the distributions for the different prescribed-burning rates can be clearly observed with a cumulative graph of all the distributions, which is presented in Figure 7.3. The skewness of the distributions is quickly reduced by additional area of prescribed burning; for 15% and 20% prescribed-burning rates, the tail is very short.

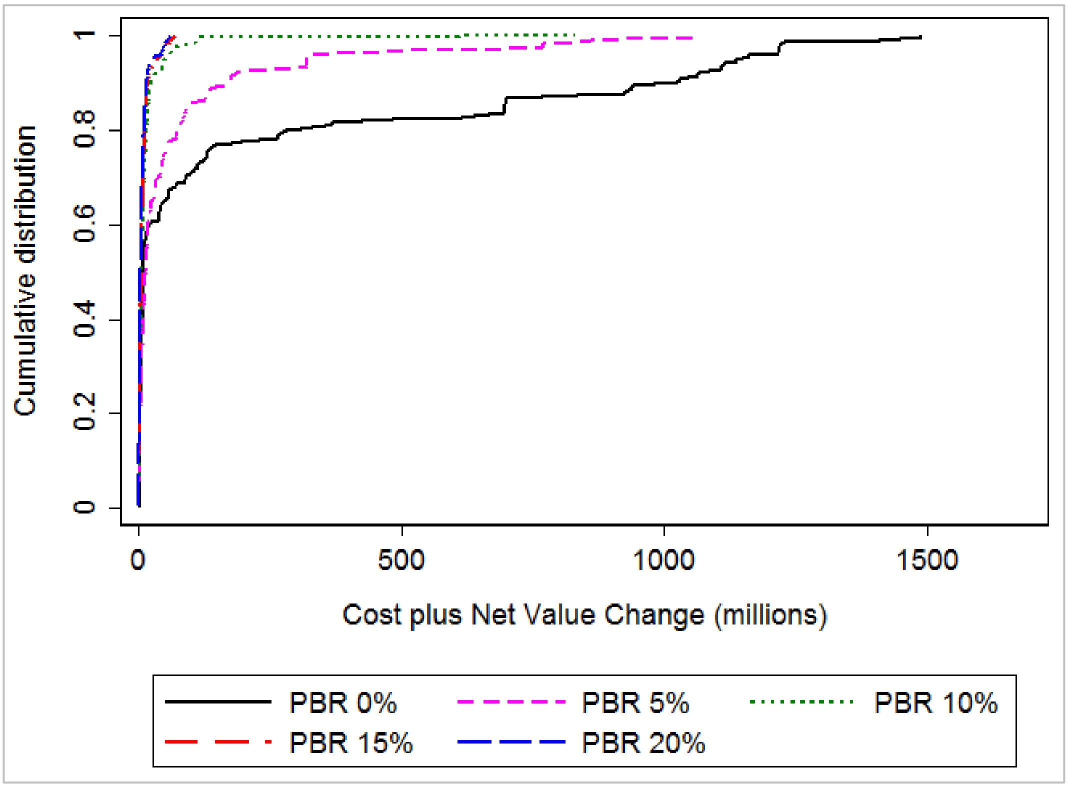

Figure 7.3. Cumulative distribution for different prescribed-burning rates (PBRs)

7.3. A more detailed examination of the distributions

In Figure 7.2 and Figure 7.3, the common x axis has a wide range and it is difficult to see the change in the bulk of the distributions. Thus, it would be useful at this point to explore the distribution of the results between AU$0 and AU$300 million. Within this range it is possible to better appreciate the distance between the mean and median. In Figure 7.4 it is possible to observe how the mass of the distributions is increasingly concentrated below AU$100 million when additional area is prescribed burned. When a prescribed-burning rate of 10% is applied, about 98% of the observations are below AU$100 million, but only higher rates (15% or more) move the entirety of the distribution to be below or equal to AU$100 million. In contrast, at a 0% prescribed-burning rate, only 68% of the observations are below AU$100 million, while at a 5% rate, about 79% are below AU$100 million. This suggests that with no prescribed burning, the sum of management costs and damages for a fire season could exceed AU$100 million about every 3 years.

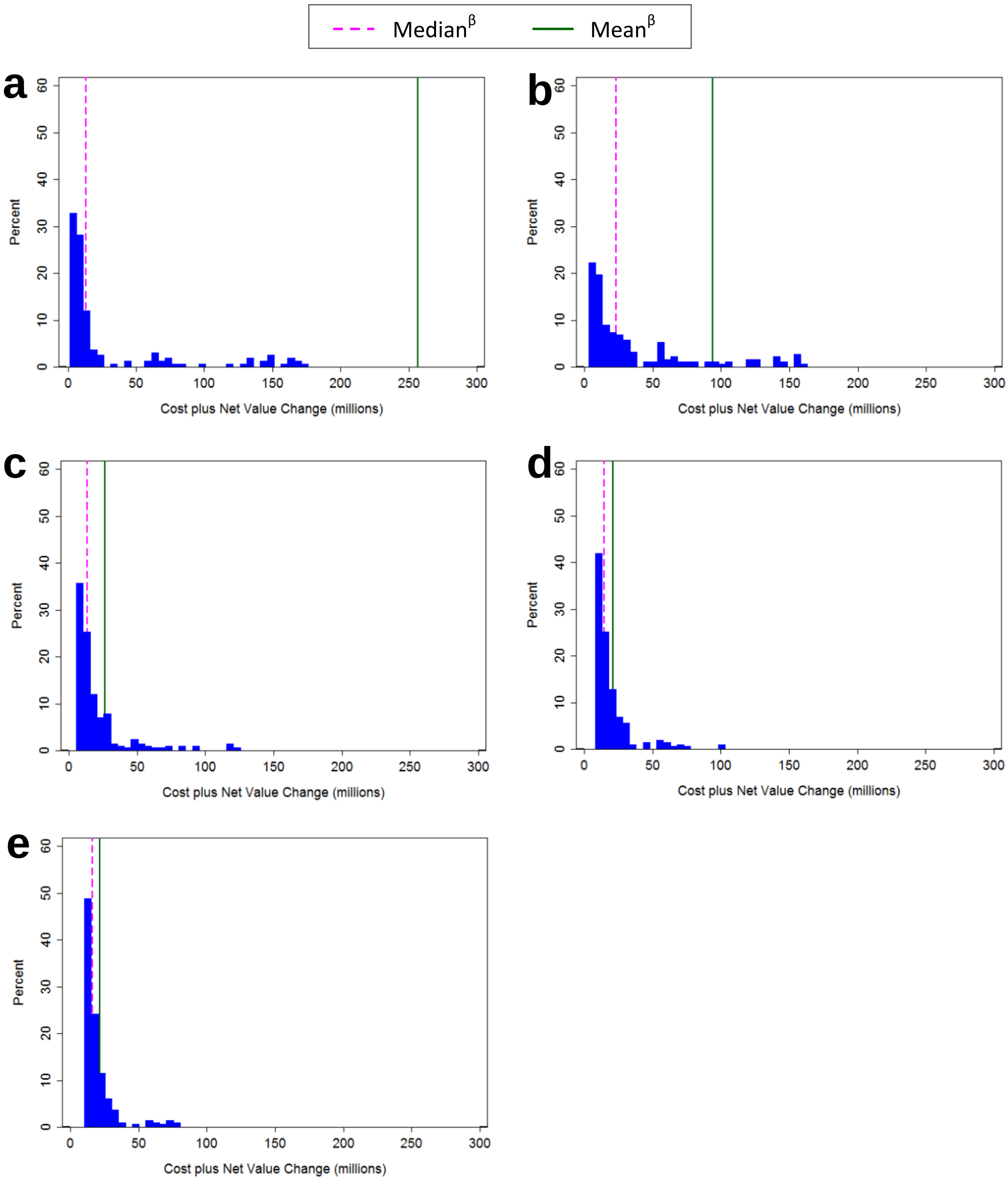

Figure 7.4. Distribution of the C+NVC results for each prescribed-burning rate (image truncated at AU$300 million)

a. 0%, b. 5%, c. 10%, d. 15%, and e. 20%

$^{\beta}$ Mean and Median are estimated from the full distribution

7.4. The distribution of damages, suppression costs and area burned by wildfires

The C+NVC includes prescribed-burning costs, suppression costs and damages, so the proportion of each variable in the distribution is masked. Thus it is useful to see the distributions of damages and suppression costs separately to see what difference can be made with different levels of investments in prescribed burning on each of these variables. Also, the distribution of area burned by wildfires can give an idea of the scale of events that can be expected for a fire season in the South West forest region depending on the prescribed-burning rate applied in the long term. All these distributions are explored in the next Section.

Another relevant consideration is the probability of a catastrophic fire season. The Australian National Emergency Risk Assessment Guidelines (Commonwealth of Australia 2015) defines five fire consequence levels according to their impact in the region: catastrophic, major, moderate, minor and insignificant. These consequence levels are defined on the basis of economic losses. They use a logarithmic scale for consequence levels, to account for the fact that the consequences of emergency events may include several orders of magnitude. The consequence levels and their corresponding criteria are shown in Table 7.2. The Gross Regional Product (GRP) of the greater South West region was $17 billion for the financial year 2013-2014.[22] We used this basis to calculate the thresholds that indicate when a consequence level is reached, shown in the last column in Table 7.2. Thus for instance, for the South West forest region, a fire event or a fire season would be considered catastrophic if it generates more than AU$680 million losses in economic activity and/or asset value (i.e. more than 4% of GRP in economic losses).

[22] The South West region of WA (as defined by the South West Development Commission, n.d.), which extends from north of Harvey to Walpole, comprises 12 local government areas or Shires. The local government areas included in the South West region do not exactly coincide with the local government areas included in the case-study area (the South West forest region), but the sizes of both regions are comparable and thus the thresholds for the consequence levels are likely to be similar. Economic data for the case-study area is not readily available because its boundaries do not coincide with the boundaries of local government areas. Most of the towns and cities of the South West region, as well as a significant part of the agricultural, viticultural, mining and timber-producing areas of the region are located in the case study area. Thus, it was considered appropriate to use the economic data of the South West region as a proxy for the case study area (the South West forest region).

Table 7.2. Consequence levels, criteria and threshold for the south-west of WA

Level	Criteria: Loss in economic activity and/or asset value (percentage of GRP)[a]	Criteria: Lower threshold for the amount of losses (AU$)
Catastrophic	>4%	>680,000,000
Major	>0.4%	>68,000,000
Moderate	>0.04%	>6,800,000
Minor	>0.004%	>680,000
Insignificant	≤0.004%	≤680,000

[a] Source: Commonwealth of Australia (2015)

Although prescribed burning does not eliminate the risk of large wildfires occurring, it does reduce the average area burned in wildfires and the intensity of those fires. As a result, investments in prescribed burning may significantly reduce the likelihood of a catastrophic fire season happening in the South West forest region. This can be clearly seen in Figure 7.5, which shows the distribution of damages for different prescribed-burning rates. Two vertical lines indicate the points where the consequences of a fire season start to be considered major or catastrophic in the South West forest region. When no prescribed burning is applied, a considerable number of observations are in the range that makes them considered major or catastrophic consequences. When a rate of 10% prescribed burning or less is applied, there is still a chance that a catastrophic fire season occurs, but this chance is rapidly reduced as prescribed burning is increased. When a rate of 20% is applied, all of the observations are within the range of insignificant to moderate consequences and there are no major or catastrophic fire seasons.

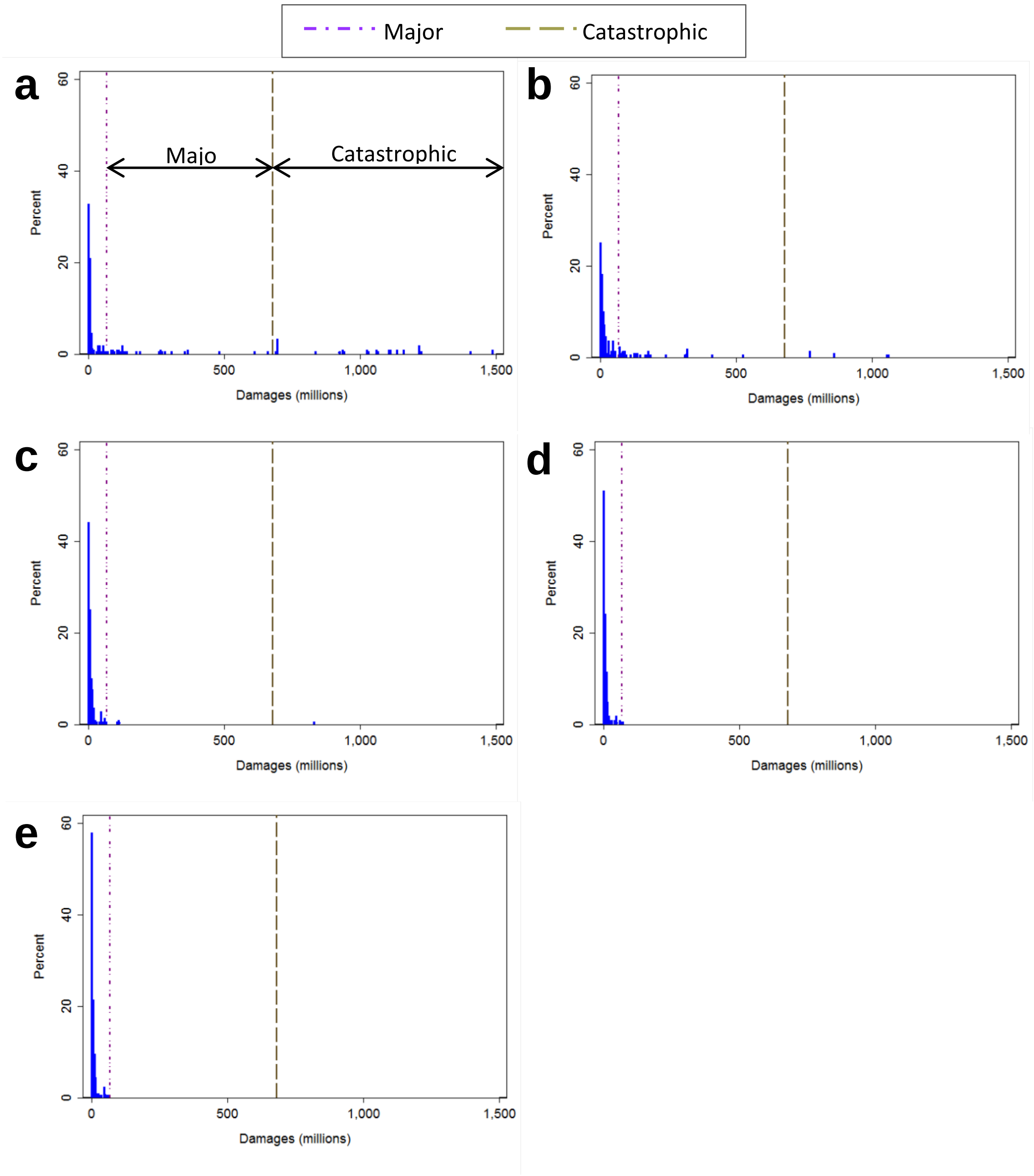

Figure 7.5. Distribution of damages for each prescribed-burning rate indicating thresholds for major and catastrophic fire seasons

a. 0%, b. 5%, c. 10%, d. 15%, and e. 20%

Table 7.3. Proportion and recurrence of fire seasons of a given consequence level for different prescribed-burning rates (PBRs)

Level	**Proportion (% out of 220 observations)**					**Recurrence of fire seasons of a given level (years)**				
PBRs →	**0**	**5**	**10**	**15**	**20**	**0**	**5**	**10**	**15**	**20**
Catastrophic	17%	3%	0.5%	0%	0%	6	33	200	0	0
Major	15%	19%	2%	0.5%	0%	7	5	50	200	0
Moderate	32%	47%	44%	39%	34%	3	2	2	3	3
Minor	36%	30%	52%	57%	62%	3	3	2	2	2
Insignificant	0%	0.5%	2%	3%	4%	0	200	50	33	25

At a 0% prescribed-burning rate, the simulations did not generate fire seasons of category insignificant and more than 2/3 of the fire seasons generated minor or moderate consequences (about 68%, see Table 7.3). But there is a relatively high chance (about 30%) that major and catastrophic fire seasons would occur. 30% corresponds to a high recurrence of these consequence levels; major fire seasons could be expected to occur every 7 years and catastrophic fire seasons every 6 years. However, as the rate of prescribed burning increases, the probability of catastrophic and major fire seasons is rapidly reduced. At a 5% prescribed-burning rate, a catastrophic fire season could be expected to occur approximately every 33 years, an increase of 27 years in the interval (compared to 0% prescribed-burning rate), and major fire seasons about every 5 years. At this rate, the region may occasionally experience fire seasons of insignificant consequence level, but only about every 200 years. At a 10% prescribed-burning rate, catastrophic and major fire seasons would be more sporadic, occurring about every 200 years and every 50 years respectively, while insignificant fire seasons could be expected to occur more frequently. At higher rates of prescribed burning (15% or 20%) no catastrophic fire seasons were recorded by the simulations and the vast majority of fire seasons (more than 95%) generated consequences that would be considered minor or moderate.

These results show that additional investments in prescribed burning may decrease the recurrence of catastrophic fire seasons in the South West forest region. To increase the interval of years between catastrophic fire seasons by one year, it is necessary to invest between AU$15,000 and AU$95,000 dollars in prescribed burning, or less if more than 10% of DPaW-managed land is prescribed burned per year. Note that the interval between catastrophic fire seasons increases exponentially as more prescribed burning is applied. Since prescribed-burning investments increase at an almost constant rate with additional area prescribed burned with the current spatial strategy (Chapters 5 and 6), less and less money is needed to increase the interval between catastrophic fire seasons as more prescribed burning is applied. Put another way, every AU$100,000 invested in prescribed burning in the region may reduce the chances of a catastrophic fire season occurring by 0.2% to 0.5%.

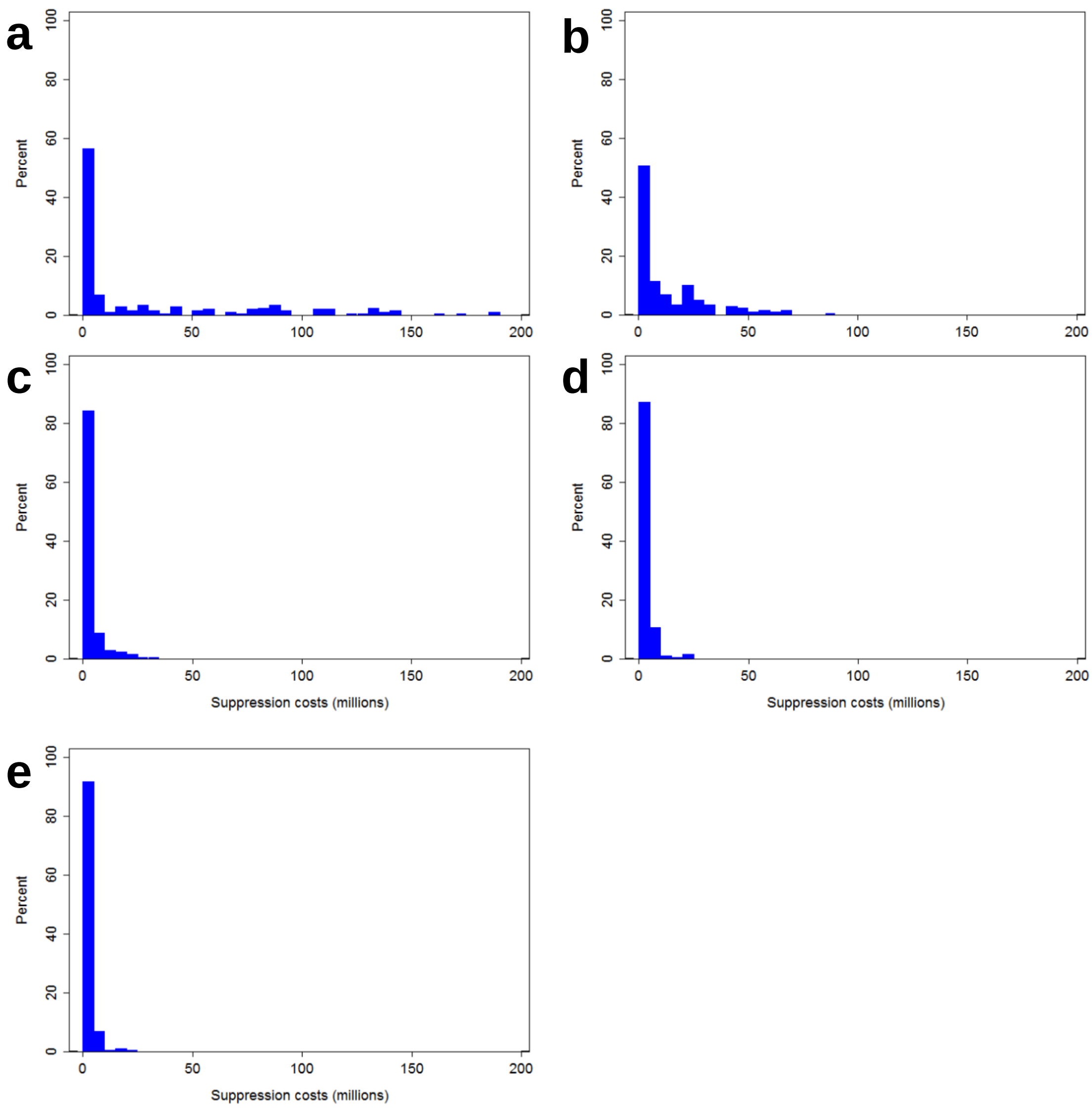

Figure 7.6. Distribution of suppression costs for each prescribed-burning rate

a. 0%, b. 5%, c. 10%, d. 15%, and e. 20%

The positive skew of suppression costs is also reduced with additional area prescribed burned (see Figure 7.6). When 5% or more of DPaW-managed land is prescribed burned, the estimated suppression costs of a fire season do not exceed AU$100 million for any of the simulated fire seasons. At a prescribed-burning rate of 10% or more, they are lower than AU$50 million. This shows that prescribed burning in the South West forest region generates substantial savings in suppression expenditures and reduces the probability of fire seasons that require an extensive use of suppression resources.

Both the average area burned by wildfires and the skewness of the distribution are significantly reduced with additional area prescribed burned (see Figure 7.7). The simulations show that if no prescribed burning is applied in the long term, the wildfires of an entire fire season may burn up to 240,000 hectares in the South West forest region, which corresponds to burning about 14% of the whole region in the space of one year. Although the probability of a fire season burning more than 200,000 hectares is lower than 2% with no prescribed burning, this probability is reduced to zero when 5% of DPAW-managed land is prescribed burned. At a 5% prescribed-burning rate, however, there is a 10% chance that a fire season may burn more than 50,000 hectares in the region (about 3% of the region). When at rate of 10% or more is applied, the area burned by wildfires in any one fire season is less than 45,000 hectares. However, even if a rate of 20% prescribed burning is applied, a fire season in the region could still be expected to burn between 15,000 and 25,000 hectares about 2% of the time, which corresponds to burning between 1% and 1.5% of the South West forest region every 50 years.

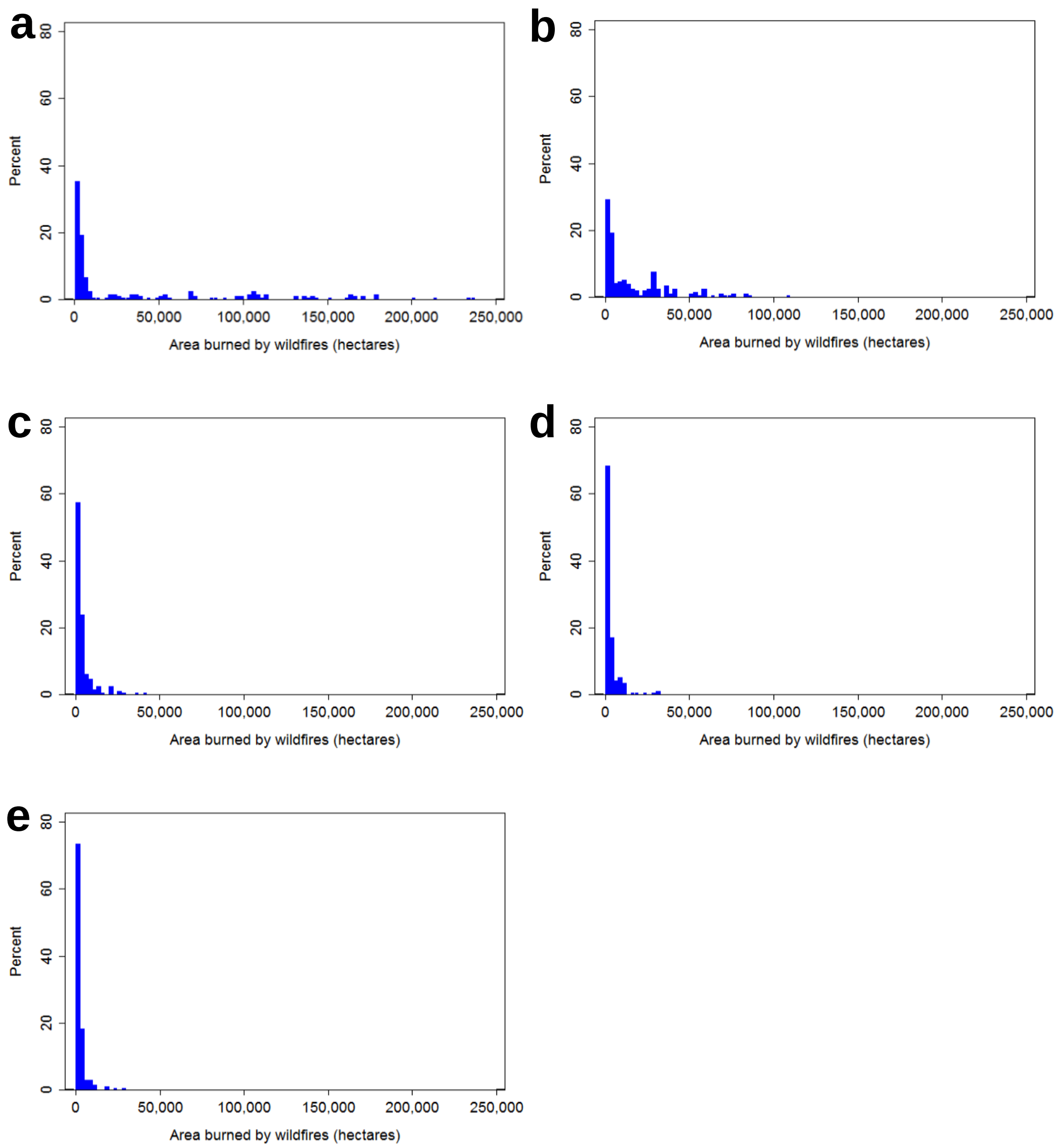

Figure 7.7. Distribution of area burned by wildfires for each prescribed-burning rate

a. 0%, b. 5%, c. 10%, d. 15%, and e. 20%

7.5. Discussion

The long-term simulation model shows that prescribed burning decreases the probability of large areas being burned by fire seasons in the South West forest region. The history of wildfires in the region has also shown this. When an average prescribed-burning rate of 5% has been applied in the region for a decade, the area burned by wildfires in a fire season for that decade has ranged between 2,000 and 81,000 hectares. When an average rate of 10% was applied for a decade, a fire season in that decade burned between 1,700 and 54,000 hectares. When an average rate >10% was applied for a decade, fire seasons in that decade burned between 1,100 and 31,000 hectares. The only exception occurred in the 1960s, when a high rate of prescribed burning was applied and a large area was burned in a single fire season: 1960/61. This experience prompted a significant increase in prescribed-burning in the region after the wildfires of that fire season burned more than 71,000 hectares.

The model also shows that investments in prescribed burning decrease the recurrence times for catastrophic and major fire seasons in the region. It is clear from these results that, at very low levels of prescribed burning, the variance of damages is significantly greater, the distributions are positively skewed with a long right-handed tail, and the chances of getting exceedingly costly fire seasons are greatly increased. In contrast, at high levels of prescribed burning, catastrophic and major fire seasons are unlikely to occur. Although the risk is not eliminated, prescribed burning has a considerable influence on the probabilities of major and catastrophic fire. This is a valuable insight and an important piece of information for fire managers. Knowing the extent to which additional investments in prescribed burning can reduce the chances of catastrophic fire seasons occurring can assist greatly when making investment decisions about the application of the treatment. It can also help inform the communities living in fire-prone areas about the expected recurrence of different levels of damages in the region.

In this case study, up to about 15% prescribed burning, there are high marginal benefits from additional prescribed burning. Beyond that point, additional prescribed burning has a much smaller benefit, although there may always be a residual risk. Therefore, even if the optimal prescribed-burning rate indicated by the simulation model is applied, relatively costly fire seasons may still occur. Thus, another important aspect of protecting the communities living in fire-prone areas in the south-west of WA is for these communities to learn to respond appropriately to the risk of wildfires that comes with living in a fire-prone region (Gill and Stephens 2009; Bradstock *et al.* 2012b).

Based on this analysis, at the current level of prescribed burning in the South West forest region (between 6% and 7% of their managed land on average per year), a fire season causing more than AU$680 million damages to the region can be expected to occur approximately every 60 to 100 years.[23] Fire seasons causing major damages (i.e. more than AU$68 million in damages) can be expected to occur about every 15 to 25 years. Although the level of

[23] These are the estimated values for 6% and 7% if a linear curve is traced between 5% and 10% prescribed-burning rates.

prescribed burning applied in the wider south-west region of Western Australia has varied greatly in the last 80 years, at least two catastrophic wildfires have occurred in the last 80 years (Dwellingup 1961 and Waroona-Yarloop 2016), and around 6 major wildfires, which corresponds to approximately one catastrophic fire every 80 years and one major fire every 13 years. Several large wildfires have caused significant damages in the region in the years:

- In 1961, 160 houses were destroyed, the town of Dwellingup was destroyed;
- In 1978, 2 lives lost and 6 buildings destroyed, narrow escape for 4 major towns;
- In 1997, 2 lives lost, 21 injuries and 17 houses destroyed;
- In 2003, 2 lives lost, more than 2 million hectares of forest destroyed;
- In 2007, 16 houses destroyed in Dwellingup, 3 lives lost and a major highway closed for 2 weeks generating significant losses to some industries;
- In 2011, 34 houses destroyed;
- In 2015, 4 lives lost and a major highway closed for several days generating losses to the dairy industry; and
- In 2016, 2 lives were lost, 181 properties were destroyed, the town of Yarloop was destroyed.

The application of very low levels of prescribed burning in the South West forest region would give rise to distributions with very long tails. Taking this into account has important implications for wildfire research and policy. For instance, perhaps more emphasis should be placed on researching the economics of extreme tails rather than on researching central tendencies. Unfortunately to date, most research on the economics of wildfire management reports only the measures of central tendency and does not focus on investigating tail events. As my research has shown, tail events are an important issue that need careful analysis and may require the use of different economic tools than those habitually used for other types of problems that do not exhibit such long tails (Nordhaus 2011; Weitzman 2011).

Chapter 8.

CONCLUSION

8.1. Introduction

In this thesis I have investigated the economic impacts of different investments in prescribed burning in the short term and in the long term in the South West forest region in Western Australia (WA). This Chapter summarises the main conclusions and implications of this study, and highlights some aspects of the research that could be addressed in further work.

The aim was to answer five key research questions for the use of prescribed burning in the South West forest region and bring to light the implications for prescribed-burning investment decisions in the region:

1. What prescribed-burning regime minimises the sum of fire management costs and damages in the South West forest region of WA?
2. Is the answer clear-cut or is there a wide range of near-optimal options?
3. What is the difference in the results between an economic analysis of prescribed burning in the short term and in the long term, and how could these results impact decision making?
4. What difference does it make to take into account the differences in asset values of different types of assets in the landscape, and the difference in prescribed-burning costs depending on size and location?
5. Given the inherently stochastic nature of the problem, what can an analysis of the distribution of results tell us in addition to the analysis of their means?

To answer these questions, a spatially-explicit fire simulation model was developed that accounts for the differences in value between different types of assets and the changes in prescribed-burning costs with treatment size and location, two aspects that had not been previously incorporated in economic evaluations of fire management options. The approach developed in this thesis integrated an economic model with a wildfire simulator and applied it to the South West forest region of Western Australia (WA) to evaluate five different prescribed-burning rates (i.e. 0, 5, 10, 15 and 20% of DPaW-managed land prescribed burned per year). The model was used to perform a short-term analysis (one year) and a long-term analysis (at “equilibrium” loads) in order to compare the results of different timeframes and elucidate the implications for short-term and long-term decision making in the case study area. Another novelty in this study was provided by the analysis of the distribution of the results in addition to the analysis of central tendencies, which provided valuable information on the relationship between prescribed-burning investments and the risk of major and catastrophic fire seasons in the region.

8.2. Summary of the results

In Chapter 2 I reviewed the literature on the economics of fire management since the early works of Headley (1916), Lovejoy (1916) and Sparhawk (1925). This Chapter revealed that despite the abundance of theoretical studies on the subject, empirical studies remain scarce and today there is a pressing need for more empirical economic analyses that account for the long-term effects of fuel management strategies. Further, economic models need to provide fire managers with tools that allow them to make well-informed decisions that effectively balance short-term and long-term fire risks. For this purpose it is necessary to elucidate the trade-offs between different fire-management activities. However, it was found in Chapter 2 that integrated economic assessments of wildfire management that compare the costs and benefits of different management options are scarce, and additional research is required in order to quantify the trade-offs between investments in fuel management, suppression costs and damages. In addition, the few integrated economic analysis available generally use the same value for all hectares in the landscape to estimate the damages and do not differentiate between the different types of assets in the landscape and their unique values. Finally, although prescribed-burning costs per unit area have been shown to change with the size and location of the treatments (i.e. lower costs per hectare for larger treatments and higher costs per hectare for treatments located close to the wildland-urban interface), these changes in costs have not been incorporated in previous analysis. This study sought to fill some of these gaps in research. For this purpose, the cost plus net value change (C+NVC) model was modified to incorporate the heterogeneity of assets in the landscape and their values, as well as changes in prescribed-burning costs per hectare with treatment size and location (Chapter 4).

A review of the history of fire and fire management in the south-west of Western Australia (Chapter 3) showed that fire is a natural and integral part of the south-west Australian ecosystems and cannot be avoided. With humans in the landscape, there is always a source of ignition, so the region will continue to experience fires every year during the hottest and driest months of the year. Therefore, it is unreasonable to consider a fire exclusion policy for the region and a new fire regime needs to be implemented in order to meet a range of objectives and challenges.

From the application of the short-term and the long-term economic models developed in this study to prescribed burning in the South West forest region, the following conclusions can be drawn:

- For a short-term analysis (one year) of prescribed burning in the South West forest region there is not a significant difference in the economic results when the level of prescribed burning is varied over a wide range of values (Chapter 5). In other words, the sum of management costs and damages (C+NVC) from prescribed burning 0% of DPaW-managed land appeared little different from the sum of management costs and damages from prescribed burning 5, 10, 15 and 20% of DPaW-managed land;
- Accounting for the differences in value between different types of assets in the landscape is important. It shows that what is usually burned by wildfires in the landscape is not what generates the greatest damages (Chapter 5 and 6);
- Including the changes in prescribed-burning costs per hectare with size and location puts a new light on the evaluation of the spatial distribution of the treatments. In the short term, the additional costs of applying more treatments close to the wildland-urban interface (WUI) are not compensated by the benefits gained (Chapter 5);
- The optimal rate of prescribed burning in both the short-term and the long-term models points to relatively high levels of prescribed burning (Chapters 5 and 6);
- Only a long-term analysis of prescribed burning in the region shows the real benefits of the application of the treatment. The long-term analysis shows that without prescribed burning in the South West forest region, average annual wildfire damages and suppression expenditures would be significantly increased (Chapter 6); and
- Examining the distributions of the results and, in particular, observing the effects of prescribed burning on high-impact, low-probability events provides a great deal of additional information about the benefits of the treatment. The analysis of the distributions shows that the application of prescribed burning considerably reduces the chances of catastrophic and major fire seasons occurring in the region (Chapter 7).

Although some studies have investigated the optimal spatial distribution of prescribed-burning treatments to minimise area burned by wildfires, the economic impacts of changing the location of the treatments had not been explored before. Chapter 5 evaluated two different strategies for the spatial distribution of the treatments in the short term accounting for the differences in asset values and the changes in prescribed-burning costs per hectare with the size and the location of the treatments. The analysis showed that although damages decrease more rapidly with additional prescribed burning when a larger proportion of the treatments are located close to the (WUI), the increase in prescribed-burning costs offsets the increase in benefits. Therefore, the strategy representing DPaW's current spatial distribution of the treatments provides higher value for money in the short term and would most likely be preferred, regardless of the prescribed-burning rate applied. However, this is yet to be tested in a long-term analysis.

Chapter 6 demonstrated that there are significant differences in the results between the short-term and long-term analyses of prescribed burning in the South West forest region. Firstly, the short-term analysis indicated a much lower benefit per dollar invested in the treatment. Secondly, as noted earlier, the short-term results indicate that there is not much difference in the C+NVC results between the different prescribed-burning rates and the model does not show a clear-cut answer. In contrast, the long-term analysis (which evaluates the C+NVC at an equilibrium state) showed a large difference between low and high prescribed-burning rates. Thirdly, the results of the long-term model indicated that the benefits of a prescribed-burning regime are only realised when the regime is applied for a long period of time, and it also demonstrated that without prescribed burning the South West forest region would experience a substantial increase in average annual wildfire damages and suppression costs. Finally, the proportions of management costs and wildfire damages for different prescribed-burning rates changed with the spatial distribution of the treatments and the timeframe used: in the short term, the proportion of damages was higher than the proportion of management costs for all prescribed-burning rates with DPaW's current spatial distribution of the treatments; but if more treatments were applied in the WUI in the short term or if the current spatial distribution was applied for long periods of time, wildfire damages could be substituted by management costs with high levels of prescribed burning.

An important novelty in this study was the development of a model that was able to determine where the highest benefits were generated in the landscape. Chapter 5 and Chapter 6 showed that although the areas most frequently burned in the case-study area were conservation areas and State forests (usually forested areas), the largest proportion of damages in dollars stemmed from the destruction of assets in the WUI. Thus, the highest benefits arose from the protection of assets in the WUI. However, this study also showed that even when the treatments were located in relatively remote areas, far away from high-value human assets, they still provided a certain level of protection to these assets because they reduced the chances of wildfires becoming large enough to reach the WUI.

The benefits of the application of prescribed burning in the long term were further emphasised in Chapter 7, which showed that the application of prescribed burning reduced the annual probability of catastrophic and major fire seasons happening in the South West forest region. By analysing the distribution of the results it was possible to investigate how additional investments in prescribed burning reduced the variance of the distributions and consequently reduced the recurrence of the most costly fire seasons in the region. This is an important conclusion and a major contribution from this study, particularly because the distribution of the C+NVC results and its implications have not been reported in previous economic studies of prescribed burning. The analysis in Chapter 7 highlighted the fact that although wildfire risk can never be entirely eliminated, when high levels of prescribed burning were applied in the long term, the majority of fire seasons generated insignificant to moderate consequences.

8.3. Policy implications

The results from this study had a number of policy implications that can help fire managers in their complex decision making with regards to investments in prescribed burning in the South West forest region.

The first policy implication is that strategic decision making must adopt a long-term perspective. This implication stemmed from the comparison of the short-term and the long-term economic analysis of prescribed burning investments in the case study area. Because the results of the short-term analysis were significantly influenced by what has happened before in the landscape, it could not provide a full picture of the consequences of not doing any prescribed burning. The long-term analysis clearly showed that the long-term benefits of the treatment are large and, if no treatment is applied, wildfire damages and suppression costs would be considerably higher. In contrast, the short-term analysis did not show much difference in the C+NVC results between different prescribed-burning rates. Thus, the results of short-term evaluations of prescribed burning investments in the South West forest region are unlikely to provide reliable information on optimal rates of prescribed burning and optimal funding. Therefore, decisions about the strategic optimal rate of prescribed burning for the region or the funds required in any given year for the application of the treatment must be derived from a long-term analysis. The wildfire management problem should not be approached from an annual budgeting perspective or follow the cycle of the State's general elections and needs to be tackled independently from them.

A second policy implication is that focusing exclusively on reductions in wildfire area can result in sub-optimal decisions about prescribed burning strategies. Several studies in the literature have investigated the optimal location of prescribed burning treatments to maximise the reduction in wildfire area (e.g. Liang *et al*. 2010; Bradstock *et al*. 2012). However, this study has shown that the prescribed-burning rate that achieves the highest reduction in average wildfire area does not necessarily correspond to the rate that minimises the sum of management costs and damages (Chapter 6). The results of the long-term model showed that a 5% prescribed-burning rate achieved the highest reduction in the average wildfire area (compared to a 0% rate), but the rate that minimises the sum of management costs and damages was 15%. Other studies have investigated the most cost effective method of minimising wildfire risk to houses (e.g. Penman *et al.* 2014). But quantifying the reduction in risk without quantifying the potential reduction in damages does not provide a full picture of the economic implications of different prescribed-burning strategies. Therefore, concentrating only on the capacity of prescribed burning to reduce wildfire area or wildfire risk without looking at its capacity to reduce suppression costs and damages can lead to sub-optimal decisions. Investigating reductions in area burned by wildfires and wildfire risk is important, but is only part of a bigger picture.

The third policy implication is that the optimal prescribed-burning rates derived from the model pointed to rates that are considerably higher than current practice in the case study area. The long-term optimal rate, that is, 15% of DPaW-managed land prescribed burned per year, was comparable to the rate that was applied the 1960s and 1970s in the region. However, in today's management context, applying such high rates may not be feasible, given the reduction in the climatic window for the application of the treatment due to climate change, past fuel accumulations and the expansion of the WUI in the region. Importantly, this study provided information for fire managers on what can be expected in the long term from the application of the optimal rate and the application of lower rates of prescribed burning.

The fourth implication is that the current spatial distribution of the treatments generates benefits from the protection of assets located far away from the treatments. This study showed that prescribed burning in the long term provides a certain level of protection to high-value human assets, even if the treatments are located more than 10 km away from those assets. With DPaW's current spatial strategy, more than 80% of the total area prescribed burned is done in treatments located more than 10 km from the WUI. The results from the long-term model showed that the damages (as measured in dollars) were predominantly caused by the destruction of assets in the WUI. Since the application of prescribed burning in the long term significantly reduced the average level of damages, this suggested that the application of prescribed burning with the current spatial strategy in the South West forest region provided a high level of protection to assets in the WUI, even when the majority of the prescribed burning treatments were located in forested areas and only a minority were placed close to high-value human assets. This also highlighted the importance of using spatially explicit models for the economic analysis of prescribed burning that account for the difference in value of the assets protected. Otherwise, it would not have been possible to discern the areas that were most frequently burned by wildfires from the assets that generated the highest damages in dollars.

The fifth implication is that investments in prescribed burning significantly reduce the frequency of catastrophic and major fire seasons in the region. The long-term results showed that there was not much difference in the average C+NVC between prescribed burning 10, 15 or 20% of DPaW-managed land in the South West forest region. However, there was a substantial difference in the probability of catastrophic and major fire seasons that could be expected from the application of each rate in the long term. At a rate of 10% there was still a chance, albeit small, of having a fire season with major or catastrophic consequences for the region. At a rate of 15% the probability of catastrophic fire seasons was reduced to zero and the probability of major fire seasons was considerably reduced. At a rate of 20%, only insignificant to moderate fire seasons were expected to occur. This analysis indicated that there are important trade-offs between the amount of area prescribed burned (and the costs that may result from it) and the expected recurrence of different levels of damages to be considered in decision making.

The information derived from the analysis of the distribution of the results may greatly assist fire managers in making decisions about investments in prescribed burning. The analysis quantified the extent to which additional investments in prescribed burning can reduce the chances of catastrophic fire seasons occurring in the region. Moreover, it showed that with the current levels of prescribed burning applied in the region, although the risk of major and catastrophic fire seasons is significantly reduced compared to 0% prescribed burning, these costly fire seasons may still occur, probably more frequently than people living in the area may think (about every 15 to 25 years for a major fire season, and every 60 to 100 years for a catastrophic fire season). This study also highlights the point that wildfire risk can never be entirely eliminated and even if the optimal prescribed-burning rate is applied, relatively costly fire seasons may still occur.

The sixth and last policy implication is that the total amount of area in the landscape that is subjected to fire of any kind (prescribed or wildfire) is much larger when prescribed burning is applied. The effect of prescribed burning is therefore a change in the fire regime, from a small number of large, intense and infrequent wildfires to a much larger number of smaller, less intense and more frequent fires. It also shifts the spatial distribution of the patches burned. This means that if people living in the South West forest region want to reduce the wildfire threat and the probability of having a catastrophic or a major fire season occurring, they would have to be willing to accept that a larger proportion of the region be subjected to fire. In addition, they would also have to be willing to tolerate more frequent prescribed burns, which may cause considerable discomfort due to increased amounts of smoke, increased potential road disruption, and an increased risk of escaped prescribed burns.

8.4. Limitations

This study had a number of limitations that related to the simulations, the data inputs used and the assumptions about risk. The first limitation is the lack of a proper suppression model that would simulate the deployment of suppression resources to each fire and together with weather conditions determine when the fire would be stopped and the costs of suppression. The model in this study used a suppression rule based on fire intensity derived from historical data, and this rule was used as a proxy for fire suppression. The costs of fire suppression were determined by the area burned by the fires. The problem with this rule, is that in non-average weather conditions, the suppression rule may have stopped the fires earlier (or later) than they would have actually lasted. In addition, for the largest fires, because the number of data points from which the rule was derived was smaller, the stopping rule for these fires may be underestimating (or overestimating) the area that would be burned under the most extreme weather conditions, high fuels and overstrained resources. For suppression models to be integrated in the fire simulations, there is a need for simulators that can be easily adapted and modified for this purpose.

The second limitation is the uncertainty associated with some of the inputs in the model. This type of integrated economic assessment has large data requirements, and all the data needed is not always readily unavailable. In some instances, the data available was aggregated at levels that impeded the estimation of some of the coefficients, e.g. the multiplier for the increase in prescribed-burning costs per hectare as the treatments were applied closer to the WUI could not be straightforwardly estimated from DPaW's data on prescribed-burning costs (which was aggregated per type of activity and per region). In other cases, there was very little information to work with, e.g. the estimated level of damages as a function of the level of intensity. In the published literature and in fire agency reports, the amount of damage caused by the fires and the intensity of the fires was only sporadically reported. A generalisation from the information available for a few fires to all the simulated fires also limits the capacity of the model to accurately represent the potential damages. Thus, improved information on the data inputs would improve the quality of the results, particularly for those inputs that may substantially influence the results of the study, such as prescribed-burning costs per ha, suppression costs for different types of fires, the relationship between fire intensity and damage, and the demand on additional resources as the area prescribed burned increases. In order to make better strategic decisions about investments in fire management programs, the acquisition of better information that can be used for integrated economic analysis should be made a priority in fire agencies.

A third limitation is the use of an aggregated land-use layer to differentiate the value of different types of assets in the landscape. Although this approach improves upon previous fire economic models by differentiating the value of the type of assets at risk instead of using a common value for all hectares, it is limited in that it assumes heterogeneity within a land-use type. However, the value of a land-use type may vary significantly from one place to another. For instance, urban areas at the edge of the WUI that include commercial, industrial or infrastructure assets are likely to have a higher value than urban areas where there are only residential buildings. This variation in value within a land-use type is not accounted for. Every effort has been made to include all possible values associated with a land-use type. However, some values have not been incorporated. For instance, the value of life and potential life losses due to wildfires were not included in any of the land-uses. Although there are few deaths in Western Australia due to wildfires compared to the eastern States, changes to wildfire impacts on human life due to prescribed burning may have a considerable influence on policy decisions.

Another limitation of the model was the lack of a proper multi-year simulator for the long-term analysis. Because the process of updating the area burned by wildfires year by year had to be done manually with the AUSTRALIS wildfire simulator, it was not possible to simulate a large set of equilibrium fuel layers. For the long-term model in this study, only one equilibrium fuel layer was used per prescribed-burning rate, and each equilibrium layer was the result of prescribed burning (or not) in different locations every year and simulating a different set of ignitions every year for 15 years. Ideally, this process would have been repeated a large number of times, to be able to create a larger sample of layers representing the long-term application of a given prescribed-burning rate in order to generate a larger number of observations. In addition, by changing slightly the location of the treatments each year from those that were used for this study, different outcomes could have resulted for the fuels in the landscape, potentially generating somewhat different results. Therefore, although the long-term model captured some of the long-term dynamics of prescribed burning and wildfire occurrence, it may not have captured fully the stochasticity of the wildfire problem. With a proper multi-year dynamic simulator, many more combinations of fuel levels resulting from the long-term application of prescribed burning could have been tested, thereby increasing the confidence in the results. For this, simulators need to be developed that can be easily automated and modified to be used in long-term economic evaluations of prescribed burning.

Lastly, the short-term and long-term models indicated the optimal prescribed-burning rate under the assumption of risk neutrality. This is another limitation of the model, because risk preferences may affect decision making. Fire managers have been known to select sub-optimal strategies that aim to reduce risk more than is optimal for society when making decisions about the deployment of suppression resources. Managers may over-allocate suppression resources even when the potential fire damage is low because they are generally risk averse (Wilson *et al*. 2011; Wibbenmeyer *et al*. 2013). But this has not been explored for decisions about prescribed burning. For a full risk analysis, different levels of risk aversion or risk seeking towards prescribed burning strategies would need to be included and tested in order to evaluate how risk perceptions affect management decisions. However, we do not have information on the risk perception of fire managers in WA.

8.5. Future research

The research in this study could be extended in a number of ways. First, in order to integrate a full suppression model, more research needs to be conducted on the types of resources and the amount of suppression resources needed for different types of fires. Although some research has been conducted on what drives suppression costs and how to forecast them (Gonzalez-Caban 1984; Gebert and Schuster 1999; Gebert *et al*. 2007; Donovan *et al*. 2008; Liang *et al*. 2008; Prestemon *et al*. 2008; Abt *et al*. 2009; Donovan *et al*. 2011, Gude *et al*. 2013), no studies have been able to determine the amount of damages (or area burned by wildfires) that could be avoided with additional investments in suppression. Butry (2009) took a step in this direction when he estimated the reduction in area burned that could be achieved by reducing the time of arrival of suppression resources to the fire, but since the costs of this time reduction could not be estimated, he could not estimate the amount of dollars needed to reduce wildfire area by a certain amount. Beyond a certain level of fire intensity, the potential for fire suppression is reduced to nil (intensity of kW/m > 10,000 and flame height >10m) (Bradstock *et al.* 2010); however, the relationship between suppression investment and damages avoided (or area burned avoided) for lower levels of intensity has not been explored.

With improved knowledge about outcomes of suppression investments and better suppression models, it would be possible to answer other research questions that cannot be answered with the current state of the model used in this study, such as: what would be the outcome for a given level of investment in prescribed burning if the investment in suppression is increased or decreased? Would the results change for all prescribed-burning rates, or only for some, or would it make no difference? In this study, suppression is modelled as an output rather than an input of the model. However, decisions about the use of resources for suppression are usually made independently from decisions about prescribed burning (Donovan and Rideout 2003b). Thus, the model would better reflect reality if suppression was modelled as an input of the model. This would also allow the trade-offs between prescribed burning, suppression and damages to be better clarified. However, for this research to be possible, fire agencies would need to collect more information on the number of resources deployed per fire, the costs of those resources, the durations of the fires, and fire behaviour at different stages of the fire in order to create datasets that can be analysed.

Secondly, this research has shown that only spatially explicit models that account for the differences in asset values and the change in prescribed-burning costs per hectare with size and location are equipped to evaluate the economic consequences of changing the arrangement of prescribed-burning treatments in the landscape. A major contribution of this study is to move in the direction of the development of models that can inform fire managers about where in the landscape and for which management options they can obtain the highest returns on investment. There is a need for more research in this area to provide more support to fire managers in identifying the trade-offs between protecting different types of asset and the trade-offs between investing in different fire-management activities. An interesting research direction would be to extend the model developed for this study to include other management options, such as education, detection, pre-positioning of fire-fighting resources, and/or initial attack. This study has also shown that it is important that this type of analyses be clear about the activities evaluated, how the benefits are calculated, and the timeframe of the approach. Otherwise, they may have little applicability. In addition, to increase the applicability of this type of analyses, the costs of transitioning to the different options evaluated need to be explored, because additional resources may be required over several years until the targeted option is successfully applied and fire managers would also need to evaluate this information in order to make decisions.

Thirdly, there is a need to develop the existing wildfire simulators into support-tools for strategic decision making. Most of the existing wildfire simulators were initially developed to support operational activities (e.g. real-time prediction of potential fire spread to help making decisions about the deployment of resources or set boundaries for emergency warnings) or to be used for training purposes. Examples of wildfire simulators that were initially developed to assist emergency management operations in Australia include AUSTRALIS (Johnson *et al*. 2008), developed at the University of Western Australia; PHOENIX RapidFire (Penman *et al.* 2013; Pugnet *et al.* 2013), developed at the University of Melbourne; and SPARK (Hilton *et al.* 2015; Miller *et al.* 2015) developed by the Commonwealth Scientific & Industrial Research Organisation (CSIRO). Because the simulator used in this study (AUSTRALIS) was originally designed to assist operational activities, the simulator could not process and generate the considerably large amounts of data needed for this economic analysis without being extensively modified. Even after the extensive modifications that were done to AUSTRALIS for this study, not all processes could be automated. For this reason, it was not possible within the scope of this thesis to conduct the long-term analysis of the close-to-towns strategy. However, this will be the subject of future work using this model.

Fourthly, this study has highlighted the importance of differentiating the values at risk and understanding society's willingness to pay for their protection. However, little is understood about the levels of risk that society is willing to accept and the willingness of society to make trade-offs between the values at risk. Fire management policies such as prescribed burning are required to meet multiple objectives of land management, community protection and environmental protection, while working under administrative and budget constraints (Stockmann *et al.* 2010b). Thus an increased understanding of society's trade-offs between the values at risk would provide fire managers with valuable information to make more economically efficient decisions about resource allocation (Venn and Calkin 2011). In 2011, Venn and Calkin called for wildfire economics research to include non-market valuation techniques (e.g. choice modelling) or multi-criteria decision analysis techniques in order to better account for the effects of wildfires on non-market goods and services. This thesis has also highlighted the need for such research to be integrated in wildfire economic analyses. Several studies have investigated people's willingness to pay for fire prevention programs (e.g. Fried *et al*. 1999; Winter and Fried 2001; Kaval *et al*. 2007; Walker *et al*. 2007; Kaval 2009; Mavsar and Ferreras 2009; Gonzalez-Caban *et al*. 2013) or their willingness to protect non-market values (such as habitat for threatened species, e.g. Loomis and Gonzalez-Caban 1994, 1997, 1998, 1999) or their willingness to pay for different landscape quality attributes (e.g. Molina *et al*. 2016), but the results of this research have not been integrated in analyses that evaluate the trade-offs between fuel management, suppression and damages and more information is needed to quantify the trade-offs that society makes between the protection of different types of assets. In addition, the non-market costs of prescribed burning, such as potential negative impacts on health (Kochi *et al.* 2010; Bowman and Johnston 2014) and biodiversity (Parr and Andersen 2006; Bradstock *et al.* 2012b), need to be investigated and included in these analyses.

Finally, the analysis of the distributions of the results emphasised the need for more research on the economics of low-probability, high-impact events rather than a continued focus on analyses based on averages. Models such as the C+NVC have been mainly designed for the analysis of average fire seasons, but in the light of climate change–which may result in a potential increase in the number of extreme weather days–and its implication for wildfires (Cardil *et al.* 2014), it is crucial to move towards the use of models that are more consistent with a risk-based management approach.

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APPENDIX 1

The two Figures below show the observed and simulated frequency distributions of wildfire size for the historical ignitions occurring in the South West forest region between 2003 and 2013.

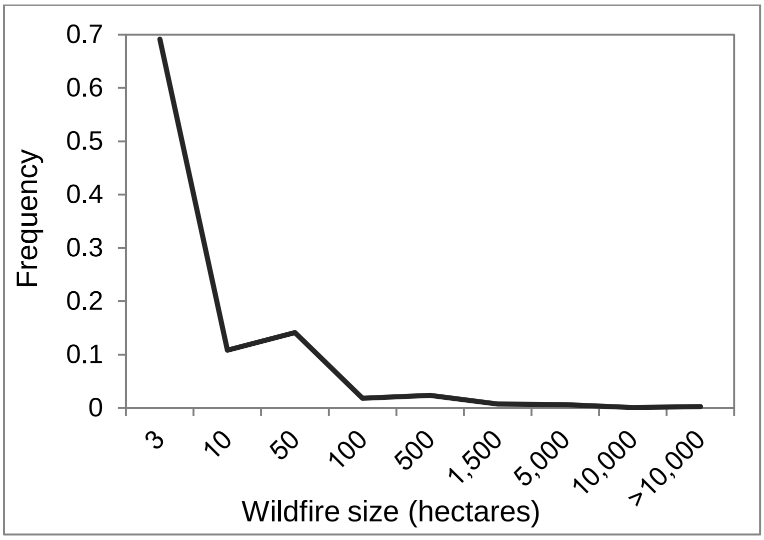

Figure A.1. Observed frequency distribution of wildfire size for the historical ignitions in the South West forest region 2003-2013

Total area burned by all wildfires (n = 1099): 119,639 hectares.

Maximum size of a wildfire: 51,594

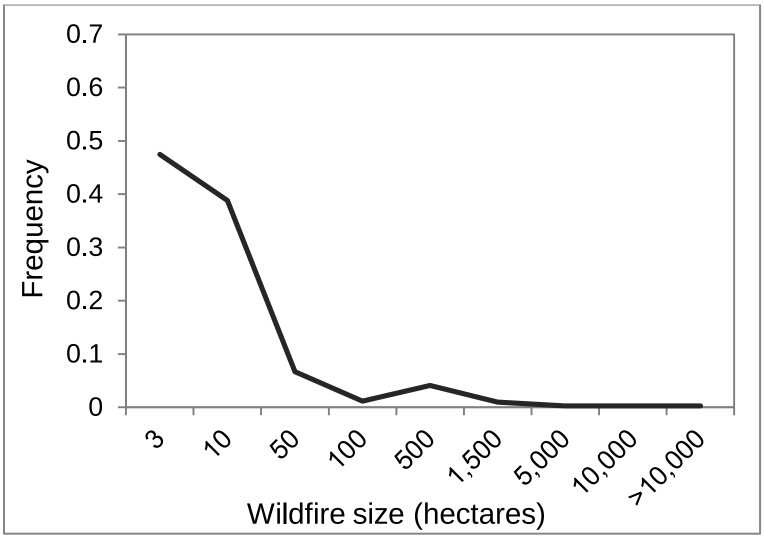

Figure A.2. Simulated frequency distribution of wildfire size for the historical ignitions in the South West forest region 2003-2013

Total area burned by all wildfires (n = 1099): 108,515 hectares

Maximum size of a wildfire: 29,262

APPENDIX 2

To simulate wildfires during 15 years with no prescribed burning, 15 new ignition patterns were created and simulated. These ignition patterns followed the same rules in terms of ignition location (according to the probabilities of ignition of each pixel), ignition time and ignition date. Figure A.3 shows the area burned by wildfire, during the 15 year cycle. Fuel load was updated annually to account for previous burns. Following a period of extremely high levels of area burned, the system stabilises to an equilibrium.

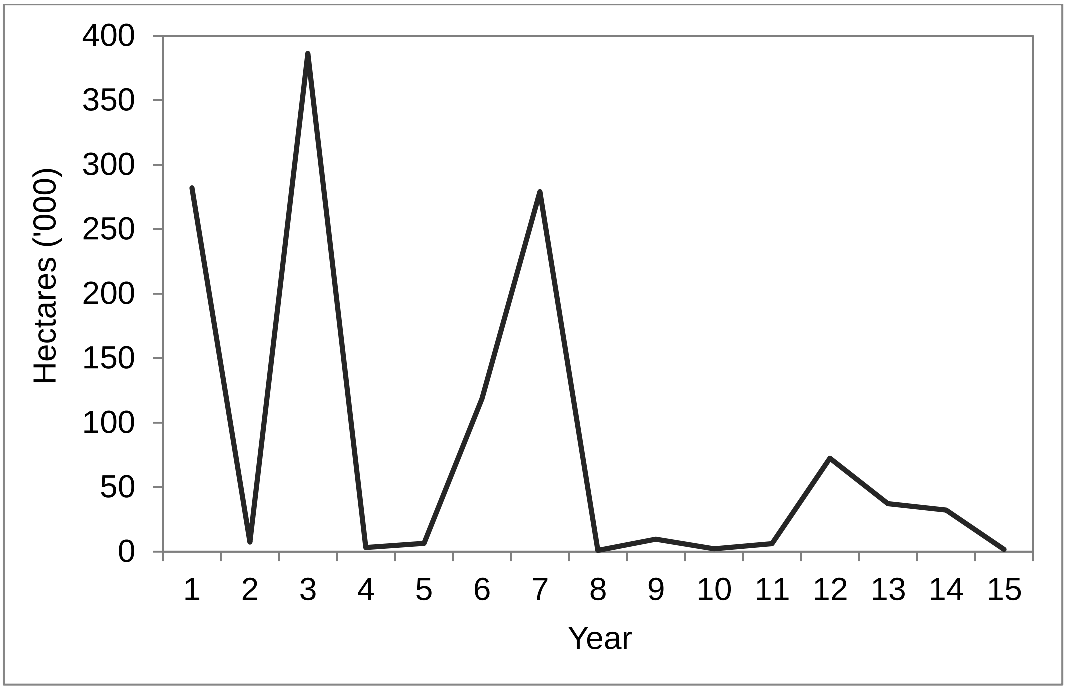

Figure A.3. Evolution of wildfire size through time for a 0% prescribed-burning rate.