WILDFIRES, FIRE MANAGEMENT, AND OPTIMAL LAND CARBON POLICY

A dissertation submitted by

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Abstract

In this dissertation I consider the social costs of carbon emissions from wildland fire, and current and future climate-conscious wildfire management in Alaska. Chapter 1 estimates how effectively current Alaskan fire management is able to reduce burned area by exploiting pre-defined spatial variation in the standard response to ignitions as an instrument for cost. This analysis finds that a 1% increase in management expenditures decrease burned area by 0.21% on average, and that the average cost per ton of CO_2 emissions avoided is $12.63. Not only is fire management effective at reducing burned area, it has potential as a relatively cost-effective climate mitigation tool. Chapter 2 presents a projection of fire and management regimes in Alaska through 2100, and models the optimal level of suppression effort over time by balancing the cost of fire management with the social costs imposed by carbon emissions. Burned area in Alaska is expected to double by the end of the century due to the increasing influence of climate change, but this research demonstrates that a significant increase in suppression effort can constrain burned area to near-historical levels. Further, the results highlight the large social costs imposed by carbon emissions during fire. When these costs are included, a 5-10x increase in management spending by 2100 will minimize combined costs by reducing burned area and carbon emissions. Finally, Chapter 3 evaluates the social cost imposed by wildfire carbon emissions by incorporating the emissions during and after combustion for a small fire into an Integrated Assessment Model. The results suggest the fire emissions impose an annual social cost of $39 billion across the United States. I also explore different modelling and discounting approaches.

Acknowledgements

None of this work would have been possible without the support of my committee. Dr. Jeffrey Zabel, Dr. Gilbert Metcalf, and Dr. Shinsuke Tanaka all provided both invaluable economic insights and emotional support and encouragement throughout the research process. I feel particularly lucky to have worked with economists who supported me in my adventures slightly outside the lines of traditional economics research.

I owe a special thank you to my fourth committee member, Dr. Brendan Rogers. He probably did not expect that the Fletcher intern he brought on in 2018 would become such a long term project. I started working with him and the rest of the Boreal team, particularly Dr. Carly Phillips, Dr. Peter Frumhoff, and Stefano Potter, with essentially no knowledge about the science of wildfires. Without their patience and support, who knows what this dissertation would have been about. I am incredibly grateful for the introduction to this fascinating and important topic.

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I am grateful to all of the friends and loved ones who have supported me throughout this journey. In particular, my mother, Dawn Duddleson, was an excellent personal research librarian. My father, Jim Elder, was always willing to talk through math and modelling problems.

Both are probably excited to never have to copy-edit another manuscript. Elaina and Maureen Thomas have always supported and believed in me (even when I was a very grumpy roommate – sorry Elaina). Finally, my partner Jackson Wallner never failed to lift my spirits with love, encouragement, and baked goods.

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Executive Summary

Wildland fire across the United States imposes significant social costs, both direct and indirect. Fires often receive significant attention for their impacts on human health and infrastructure. However, active fires are also a large source of atmospheric carbon dioxide, which exacerbates climate change and has long term impacts on human wellbeing as a result. Furthermore, as climate change leads to intensifying fire regimes, this positive fire-climate feedback will only strengthen. Each chapter of this dissertation explores a different aspect of wildfire and fire management policy in the context of a changing climate.

Chapter 1 examines current fire management in Alaskan and estimates its effectiveness at reducing burned area. I use an Instrumental Variables approach to obtain a causal estimate of the impact of a marginal change in fire management spending, by using Fire Management Zones, pre-assigned designations of the standard response to an ignition which vary geographically, as an instrument for cost. This avoids the problem of reverse causality, in which larger fires are more costly to manage than smaller fires, which obscures management effectiveness. I find that a 1% increase in management expenditures decreases burned area by 0.21% on average, with high significance. Further, the estimated management cost to avoid an additional ton of CO_2 emissions is $12.63, which is a much lower cost per ton than other emissions reduction technology like direct air capture.

In Alaska, burned area is expected to double by the end of the century due to the increasing influence of climate change. Chapter 2 presents a model for projecting fire and management regimes in Alaska through 2100, which incorporates responsive management effort, the growing influence of climate change on fire regimes, and the dampening influence of recent

combustion on flammability. The primary specification models the optimal level of suppression effort over time by minimizing the combined cost of fire management and the social costs imposed by carbon emissions. In this cost-minimizing specification, fire management expenditures increase 5-10 fold by the end of the century in order to constrain burned area to near-historical levels. Alternate specifications explore optimal management under other potential climate outcomes, as well as burned area and cost outcomes under different management goals. The results also highlight the large social costs imposed by fire carbon emissions, which are an order of magnitude larger than management costs by the end of the century if management expenditures remain at their current levels.

Finally, in Chapter 3 I estimate the social cost imposed by theoretical wildfire carbon emissions. I modify an existing Integrated Assessment Model, which is typically used to calculate the Social Cost of Carbon, by simulating the emissions from a small, 1000 hectare fire in central Oregon. I incorporate both the large amount of carbon emitted during combustion, and the positive and then negative emissions which occur due to decomposition and then forest regrowth over the subsequent 300 years, and compare total economic welfare with and without this 300 year emissions curve. I find that the small fire imposes $27 million in social damages, and the social cost of the emissions from all burned forest across the United States is approximately $39 billion annually. I also explore alternative modelling and discounting approaches, and highlight some potential pitfalls of these common methods.

Overall, this research highlights and quantifies the costs imposed by wildfire carbon emissions through the channel of climate change. While these costs are large, I also find that fire management is an effective tool for reducing burned area, and that its increased application is a cost-effective way to decrease burned area and carbon emissions, particularly in Alaska.

Chapter 1: The Effectiveness of Fire Management for Reducing Burned Area in Alaskan Boreal Forests

Molly Elder[1]

Abstract: Alaskan boreal forests store large amounts of carbon, but because of climate-driven, intensifying wildfire regimes, these forests are expected to become a net source of carbon into the atmosphere in the near future. The primary means to address wildfire in the state is post-ignition fire suppression. This paper uses the pre-assigned and spatially variable standard responses to detected ignitions as an instrument for management costs in order to produce a causal estimate of the effectiveness of fire management for reducing burned area. We find that management spending is effective, and a 1% increase in spending leads on average to a 0.21% reduction in burned area. Further, we use this coefficient estimate to calculate that the cost per ton of CO_2 emissions avoided is $12.63, much lower than the marginal damages imposed by additional emissions. Fire management is an effective and efficient tool for reducing burned area and the resulting carbon emissions.

[1] This chapter contributes to the paper Phillips, C. A., B. M. Rogers, M. Elder, S. Cooperdock, M. Moubarak, J. T. Randerson and P. C. Frumhoff (2022). "Escalating carbon emissions from North American boreal forest wildfires and the climate mitigation potential of fire management " Science Advances **8**. The extended text presented here is included with permission from AAAS.

1.1 Introduction

Boreal forests are a conifer-dominated forest biome which comprise a band of land below the globe's northern pole, and account for about 10% of all land area globally. Despite covering only about 10% of Earth's land area, these forests store between 30% and 40% of Earth's terrestrial carbon, primarily in deep, carbon-rich organic soil layers (Carlson et al. 2009). However, forest fires burn millions of acres of boreal forest each year, releasing large amounts of carbon into the atmosphere. In Alaska alone, 1.2 million acres burn during an average fire season (Division of Forestry 2015).

Historically, boreal forests were net carbon sinks, meaning they fixed more carbon in vegetation and soil each year than was released by decomposition and fires. However, as climate change brings warmer, drier weather, the length and intensity of the average fire season has been increasing in boreal regions around the globe, including Alaska (Kasischke et al. 2010, Turetsky et al. 2011), Canada (Gillett et al. 2004), and Siberia (Ponomarev et al. 2016). The average number of acres burned each year has been increasing, and as fires burn hotter and more intensely and penetrate deeper into the organic soil, more carbon is being released (Turetsky et al. 2011). Because of these trends, scientists predict that boreal forests will soon become a net source of carbon into the atmosphere, rather than a net sink (Walker et al. 2019). The impact of climate change on fire regimes, and the climate implications of these changing regimes, make boreal fires an important area of study for mitigating the effects of climate change.

In this paper, I investigate how effective fire management is at reducing burned area in Alaskan boreal fires. In Alaska, the primary fire management tool is suppression once a fire has started, so I focus on these efforts[2] rather than on defensive measures that can be taken before

[2] I will not address the effectiveness of fuel management, the process of limiting burnable material to reduce the risk that a fire occurs or spreads to a specific area. While fuel management has a very important role in populous and

ignitions occur. Further, while there are large swaths of boreal forest in Alaska, Canada, Russia, and Northern Europe, this paper focuses on Alaskan forests alone. Siberian data is very scarce, and while Canada has fairly plentiful data, policy variation across provinces makes a consistent analysis of management practices challenging. In contrast, Alaska has a unified management strategy for the whole state based on well-defined management zones, and management is handled by only three main agencies and is clearly defined. The Bureau of Land Management is responsible for most federal land, as well as native corporation land; the Alaska Department of Natural Resources' Division of Forestry is responsible for state, private, and municipal land; and the US Forest Service is responsible for remaining federal land, primarily national parks and forests (Division of Forestry 2000). These factors make analysis much more straightforward in Alaska than in Canada.

In an average year, CO_2 emissions from Alaskan fires far exceed both emissions from human activity (called anthropogenic emissions) in Alaska and fire emissions from other states, including California, which has one of the highest levels of annual fire emissions in the lower 48 (Wiedinmyer and Neff 2007). Because of Alaska's limited population, Alaskan fires generally do not put as many human lives at risk as the high-profile fires on the West Coast, but as a portion of the nation's emissions they are much more significant: on average, Alaskan fires emit 80 gigatons (Gt) of CO_2 annually (with very large annual variation), equivalent to about 1% of total average US anthropogenic greenhouse gas (GHG) emissions[3] (Wiedinmyer and Neff 2007, Janssens-Maenhout et al. 2017). Moreover, Alaska is significantly overrepresented as a share of national emissions from wildfires: despite making up only 18% of US land area, Alaskan fires

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accessible areas in the wildland urban interface (WUI) like the outskirts of Fairbanks or Anchorage, it is not feasible to extend this strategy to the entire state.

[3]The US emits approximately 6500 Gt of CO_2 equivalent per year from anthropogenic sources.

contributed half of US fire emissions from 2001-2020 (MTRI 2023). In 2004, Alaskan fires emitted an estimated 250 Gt CO_2, the equivalent of nearly 4% of US anthropogenic emissions (Veraverbeke et al. 2017). Because of their significant climate implications, Alaskan fires need to be an area of focus for American fire and climate policies, and focusing this paper on Alaska rather than on a wider range of boreal areas will make the results directly relevant to policy considerations in this country.

In addition to a desire to inform policy, this paper focuses on Alaska because of the state's unique approach to fire management. All of the land in Alaska is sorted into one of four Fire Management Zones (FMZs), based on the potential of a fire to damage values at risk. These zones are also referred to as Fire Management Options or Fire Protection Options in official documents. Typically, causal identification of the effectiveness of fire management is difficult, because larger fires are more expensive to manage, introducing a reverse causality problem. However, because the standard management response to a detected ignition differs by FMZ, but FMZ designations are relatively inflexible over time, FMZ designations can be used as the basis of an Instrumental Variables identification strategy. Alaska is a promising laboratory to study the effectiveness of management expenditure (and effort) on fire size.

The rest of the paper is organized as follows: Section 1.2 reviews the relevant economic and ecological literature, Section 1.3 describes the data approach, Section 1.4 presents the model and identification strategy, and Sections 1.5 and 1.6 present and discuss the results.

1.2 Literature review

Damages associated with wildfires have been widely studied in the economic literature (Melvin et al. 2017, Thomas et al. 2017), including the negative health effects of smoke inhalation (Jones et al. 2016, Kochi et al. 2016), the effect of wildfires on real estate dynamics (McCoy and Walsh 2018, Mueller et al. 2018), and moral hazard problems involving wildfire risk and building decisions (Baylis and Boomhower 2019). However, there is only limited research into the costs associated with fighting fires or the relationship between management expenditures and fire size, and even less research into the causal impact of management spending. Focusing on the Wildland Urban Interface (WUI), Baylis and Boomhower (2019) find that there is a positive relationship between the proximity of the point of ignition to homes and the resulting management expenditures in the Western continental US, but they do not comment explicitly on the effectiveness of those expenditures in protecting the endangered homes. There is evidence of a tight correlation between burned area and management expenditures (Calkin et al. 2005), but this is indicative of the huge expense of fighting large fires, and does not directly speak to the causal relationship this paper hopes to elucidate.

Fire size is an important metric for this research because it is closely tied to the total amount of carbon released in a fire, so limiting the acres burned can be a straightforward way to limit fire emissions. Ecologists have long understood the importance of both weather and vegetation type in predicting fire size (Calkin et al. 2005, Coffield et al. 2019), the measure of fire severity that this paper focuses on. There are a number of different indices related to drought and other fire-facilitating weather which have been developed over the years (Canada 2019), detailed in Table S1.1. The closest research to this paper, by Coffield et al., attempts to use vegetation cover and VPD (Vapor Pressure Deficit) to predict whether a fire will become "large," and focuses on boreal Alaskan forests over a similar time period to the observations used in this paper. Among other results, the authors find that their model does not predict large fires as well in more actively managed zones, suggesting that more intense suppression efforts which occur in these zones are effective in reducing burned area from intervention-free levels. This paper goes further by using management expenditures for each fire, rather than using only average levels of effort represented by different zones. This allows me to explore not only whether management is effective at reducing burned area, but how much it costs to reduce fire size, opening the door for comparisons to other cliamte mitigation strategies and concrete policy applications.

There appears to be a gap in both the ecological and the economic literature on the causal effect and cost-effectiveness of fire management on reducing burned area. Without knowing the cost-effectiveness of management, it is difficult to make budget projections for burned area policy goals, or to assess fire management's role in climate mitigation. As the introduction makes clear, this is an area of interest as climate change intensifies fire regimes and increases burned area and GHG emissions. This paper attempts to fill the existing literature gap.

1.3 Data

The primary dataset for this analysis consists of 849 records from the US Bureau of Land Management (BLM), of individual fires which occurred in interior Alaska between 2007 and 2015 in the months of June, July, and August. Each record includes BLM's management expenditures on the fire as well as the total burned area, the cause of the fire (human or lightning) and the number of burned acres under BLM protection. The records also include the Fire Management Zone (FMZ) in which each ignition occurs. FMZ (Figure 1.1) is used by fire managers to determine the appropriate response to an ignition. This response varies from immediate action to observation only.

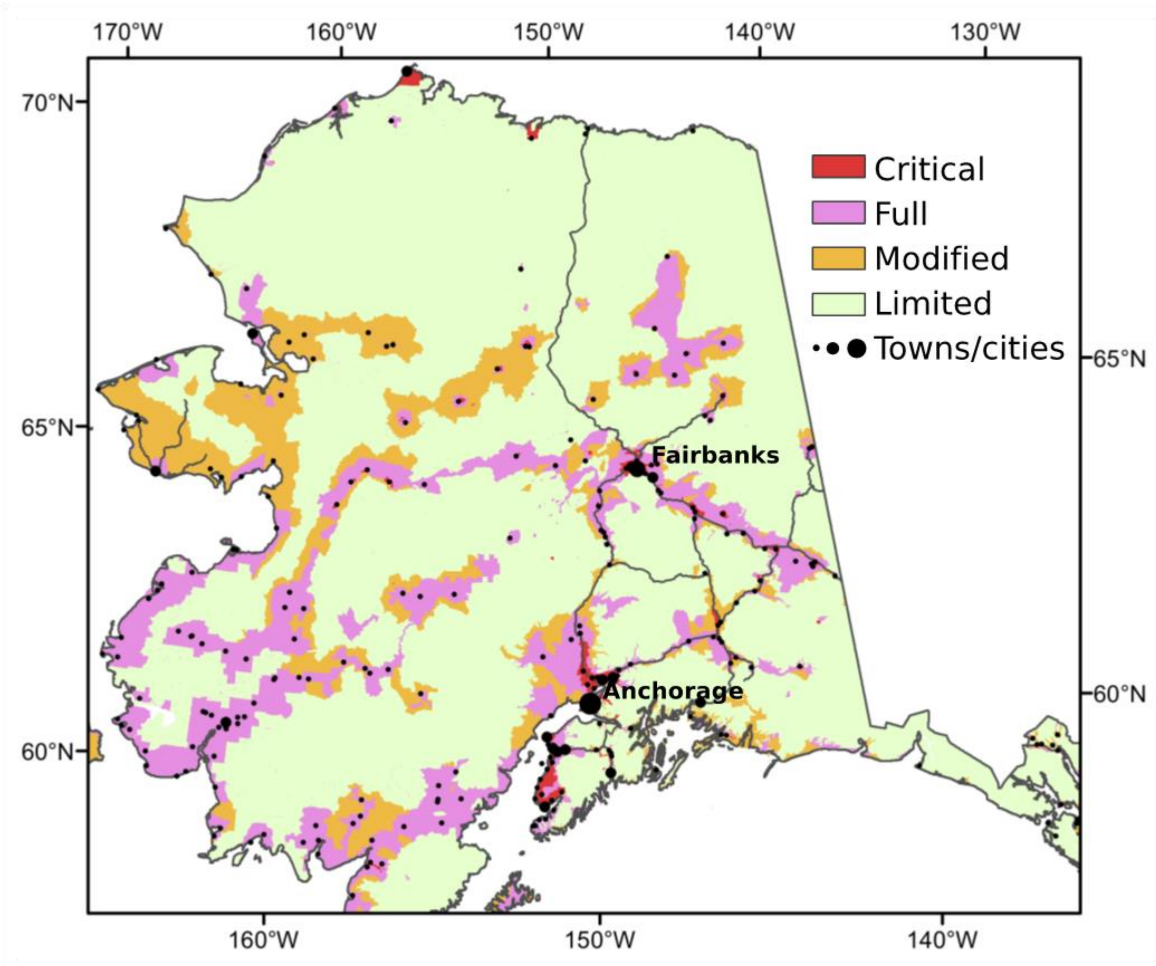

Figure 1.1: Map of Alaskan Fire Management Zones. Red areas are designated Critical; orange designates Full; yellow designates Modified; green designates Limited. Reproduced from Phillips et al. 2022.

The FMZ categories are Critical, Full, Modified, and Limited, and each one has a standard response if a fire is detected. The Critical FMZ designates areas where a fire could threaten human life, inhabited property, or infrastructure, and is generally reserved for larger population centers (red areas in Figure 1.1). The goal of management in areas designated Critical is to limit the fire to "the smallest acreage reasonably possible" (Division of Forestry). Areas designated Full (orange in Figure 1.1) have uninhabited private property like hunting cabins, or are valued for natural resource extraction or cultural significance including Alaska Native lands (Division of

Forestry 2015). The goal of management in Full is also to limit burned acreage as much as possible, as long as resources are not needed for fires in Critical. Modified areas are treated like Full at the beginning of the fire season, and then after the "conversion date," which typically occurs in early July, fires in these areas are typically allowed to burn (yellow in Figure 1.1). In our data set, we designate these areas as Full prior to the conversion date and Modified afterward, to capture this change. Table S1.2 lists conversion dates by year for the sample period[4]. Finally, areas in Limited protection are generally quite remote and far from defined values at risk (green in Figure 1.1). The standard response to fires in this zone is monitoring, unless they start to threaten structures or other points that might require action, allowing fires to perform their "ecological function" (Alaska Wildland Fire Coordinating Group 2019).

The available cost data include only costs incurred by the Bureau of Land Management (BLM) of managing each fire, not the total cost of managing each fire. In Alaska, individual fires often burn over land that is owned by multiple federal and state agencies, native corporations, and private owners. The three agencies responsible for fire management in Alaska are the Bureau of Land Management (responsible for most federal land, and native corporation land) and the State of Alaska Department of Natural Resources' Division of Forestry (responsible for state, private, and municipal land) and the US Forest Service (Division of Forestry 2000, Slaughter 2019). To estimate the total expenditures on each fire, I used the fact that, in the Critical, Full, and Modified zones, the percentage of the total cost borne by BLM is approximately equal to the percentage of burned land under BLM protection (Slaughter 2019). In other words, if 50% of the acres burned by Fire 1 are on federal land, BLM will ultimately settle with the State of Alaska so that it pays 50% of the costs. If BLM took the lead on management while the fire was active, it

[4] All conversion dates occur within a fifteen day period. The earliest date is July 6, and the latest is July 21. In more than half the years sampled, the conversion date is July 10.

might have paid all of the cost initially, but at the end of the season it will be reimbursed by the state for 50% of costs.

I estimate the total costs of each fire by comparing the total acres burned to the number of burned acres under BLM jurisdiction or protection, conditional on zone. In the Limited zone, all fire costs are borne by BLM (Slaughter 2019). In the other three zones, there are 216 out of 338 fires for which BLM reported positive costs, but did not report having responsibility for any acres. Nonetheless, these fires are relatively small – all are under 100 acres, and 75% are under 6 acres. In these cases, I assume that these fires were attacked quickly by BLM, which took responsibility for all of the costs. Because BLM was not going to have to settle with other agencies, the BLM-jurisdiction acres were the same as total acres and were not worth reporting separately.

For 88 fires, the acres attributed to BLM exceed the total reported acres burned. However, only in 25 cases do BLM acres exceed total acres by more than 1%. I exclude these 25 fires from analyses using estimated total cost, because there is no way to confirm which value was correct. For the other 63 fires, I assume that the small discrepancies (<1%) are due to rounding differences (i.e., 9.9 acres burned total, 10 BLM acres reported) and that BLM was responsible for all the burned area. This brings the total number of observations to 832. There are 46 fires for which BLM had positive costs and shared responsibility, and for these I estimate the total management costs using the fraction of burned area for which BLM was responsible.

In addition to suppression efforts, weather conditions and vegetation play a large role in the ultimate size a fire will achieve (Coffield et al. 2019). For each observation, mean and maximum measures of fire-relevant weather indices over the duration of the fire from the Global Fire Weather Database (Field et al. 2015) were gathered, as well as elevation at the point of ignition and the share of different types of vegetation coverage across the burned area from the USDA's LANDFIRE (Ottmar et al. 2007, LANDFIRE 2008). These disparate datasets were merged by C Phillips (Phillips et al. 2022). Table S3.3 reports summary statistics for a selection of these variables.

1.4 Model

There is a large body of literature on the effect of environmental drivers on patterns of fire occurrence, dynamics, and burned area (Veraverbeke et al. 2017, Coffield et al. 2019). However, once these factors are controlled for, there is still large variation in fire size (Coffield et al. 2019). I hypothesize, following Coffield and others, that a driving factor behind this remaining variation is the level of management effort or expenditure allocated to a given fire. Unfortunately, performing a simple OLS regression of burned area on expenditure does not capture this relationship successfully, because there is reverse causality between cost and burned area. Other factors held constant, it will take more effort (and hence require higher expenditures) to control a large fire than a small one, so the size of a fire could drive cost. This effect is shown in Column 1 of Table 1.1 (in Section 1.5), where a positive coefficient on cost suggests that increasing expenditure will increase fire size.

To address this endogeneity issue, I use an instrumental variables (IV) approach. The goal is to find an instrument that is correlated with cost but will not be driven by burned area. I propose that the Fire Management Zone (FMZ) at the point of ignition is a good candidate. To justify the use of FMZ, I must argue convincingly that FMZ is correlated with cost (the "relevance restriction"), and that it affects burned area only through the channel of cost (the "exclusion restriction").

It is relatively easy to satisfy the relevance restriction, that FMZ is a determinant of cost. The initial response to a fire is largely determined by the FMZ in which that fire starts. Fires in Critical and Full are attacked aggressively, which means they have relatively high costs, and fires in Limited are only monitored by fly-overs or through satellite images, which sometimes means that these fires have no cost attributed to them at all. FMZs are assigned only based on the value at risk from fire in a particular area and can give an indication of human presence, which drives management choices. While zone boundaries are reviewed annually to respond to changes in land use or values at risk that may occur, these changes are made during the winter, not during fire season (Alaska Wildland Fire Coordinating Group 2019) and should be primarily a response to shifting concentrations of human population and activity. Figure 1.2 shows the annual percent change in the number of pixels in each management zone (Figure 1.2a) and the amount of land covered by each zone every year (Figure 1.2b). Full, Modified, and Limited zones exhibit only small changes in area throughout the sample period, and other than one year with a 20% decline, Critical is also fairly stable.

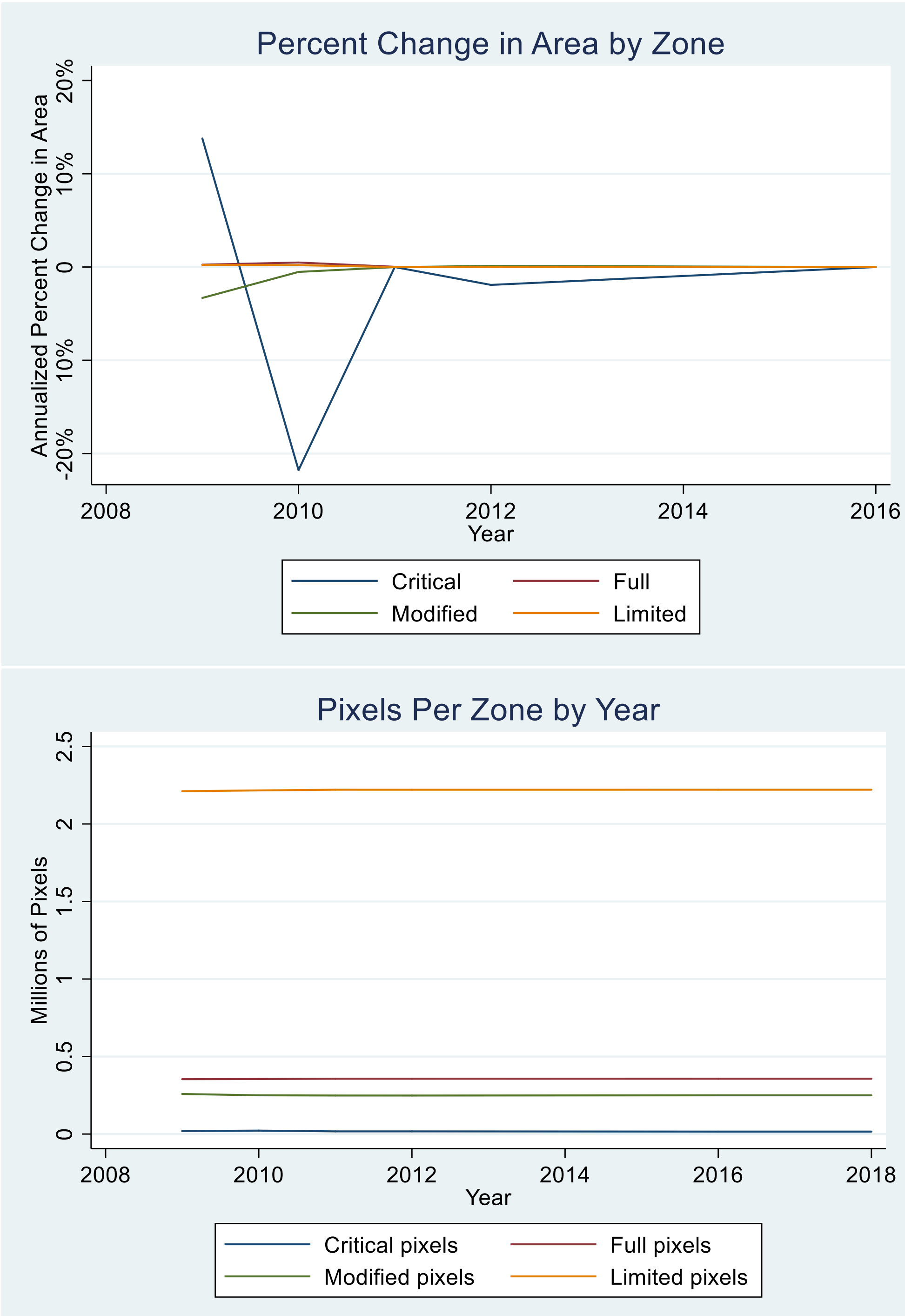

Figure 1.2: Annualized percent change in number of pixels in each zone (a) and pixels by zone by year (b). Each pixel is a 500m x 500m square

Any area where human life or inhabited structures could be threatened is in Critical, areas with uninhabited structures or culturally or economically valuable sites are in Full, and Modified and Limited areas have minimal human presence. Figure 1.3 shows the average BLM cost and cost per acre for fires in each zone – average BLM costs are highest in the Critical zone and decrease by zone until reaching the lowest costs in Limited. Costs per acre are much higher in Critical and Full than in Modified or Limited. Per acre costs are a bit higher for Full than for Critical, which I interpret as a reflection of a combination of the difficulty of fighting fires once they have grown in size, and the cost of fighting fires further from population centers. Table 1.2 (also in Section 1.5) gives first stage results, which show that FMZ is a significant predictor of cost.

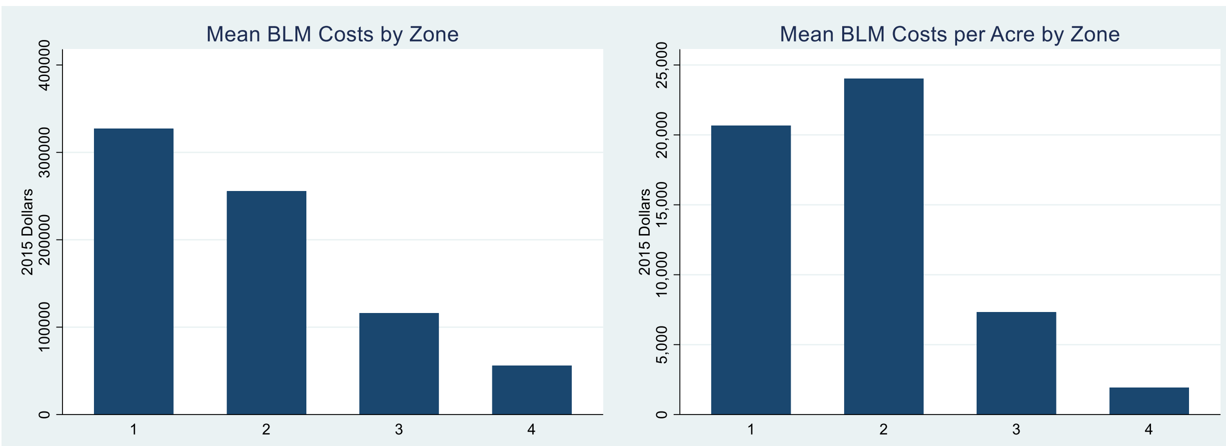

Figure 1.3: Average BLM costs and per-acre costs by zone, from 1 (Critical) to 4 (Limited)

For FMZ to be an acceptable instrument, it must not only be correlated with cost (the relevance restriction), but also affect burned area only through the channel of cost (the exclusion restriction). To be comfortable using FMZ as an instrument, the reader must be convinced that the only way fire size is affected by FMZ (approximately, the proximity of a fire to areas of interest to humans) is that proximity increases the interest humans have in putting out the fire and thus the resources poured into managing that fire. More simply, FMZ must only affect burned area through the channel of management expenditure. I address several concerns about the exclusion restriction in this case.

One possible way for FMZ to affect fire size directly is that the sites where human settlements are built could be geographically dissimilar to unpopulated areas in ways that affect fire size. To address this, I control for latitude and longitude at the point of ignition, as well as types of vegetation coverage across the burned area. Table S1.4 presents a balance table of mean weather and vegetation variables by zone, as well as burned acres. The means do not appear to be statistically different by zone, except in the case of acres burned, the variable I expect to be influenced by zone.

A potentially more concerning channel through which FMZ might affect fire size directly is that fires near population centers (Critical or Full options) might differ in some way from fires far from population centers. One way this could be true is that fires started by human activity (arson, bonfires, malfunctioning equipment, etc.), which are more likely to occur in high-population areas, might start under different weather and vegetation conditions, or be managed differently in the early stages than fires caused by lightning. For example, fires started by human accident might be reported very quickly compared to lightning-caused fires. I address this concern by controlling for whether a fire was caused by humans (70 observations) or lightning (786 observations). Only 1 fire in the sample has an unknown cause.

Another way fires could differ across FMZs in a way that affects fire size is through discovery time; in other words, fires that start far from populated areas might burn for more time before they are discovered. This might be of greater concern if the dataset included fires from before 2000, but since the earliest observations are from 2007, the use of satellite technology should ensure that almost all fires are discovered within a short period after ignition. Modern remote fire detection relies on hot spot data from two satellite sensors, MODIS (the Moderate Resolution Imaging Spectrometer) and VIIRS (the Visible Infrared Imaging Radiometer Suite) (Alaska Wildland Fire Coordinating Group 2019). MODIS includes two satellites, Aqua and Terra, and has provided scans approximately eight times daily since 2002, while VIIRS has provided twice-daily scans since its launch in 2011 (NASA 2019, NOAA 2019). Lightning, the primary ignition source of remote fires, is detected using the Alaska Lightning Detection Network, which has been in operation since the early 1980s and updates in nearly real time (Fronterhouse 2012, Alaska Wildland Fire Coordinating Group 2019). Because the earliest fires in the dataset occurred in 2007, well after all of these instruments came online and were adopted for operational purposes, we can assume that even fires in very remote areas are discovered quite quickly, so any significant difference in response time can plausibly be attributed to intentional policy difference. Furthermore, since the standard response to remote fires is to monitor them without significant intervention, a slightly longer time prior to discovery is unlikely to have a large impact on the ultimate management response.

Having justified the need for an IV approach and the appropriateness of using FMZ as an instrument, the resulting IV model is given by Equation 1.1 (the First Stage) and Equation 1.2 (the Second Stage).

$$ln(cost)_i = a_0 + a_1 FMZ_i + \alpha_2 X_i + \varepsilon_i \tag{1.1}$$

$$ln(firesize_i) = \beta_0 + \beta_1\, ln(\widehat{cost_i}) + \beta_2 X_i + u_i \tag{1.2}$$

where X_i is a vector of control variables including the weather, elevation, vegetation type, year and month fixed effects, and cause for fire i, and u_i and ε_i are error terms. The reduced form of the IV model is given by Equation 1.3.

$$ln(firesize_i) = \beta_0 + \beta_1 FMZ_i + \beta_3 X_i + u_i \quad (1.3)$$

To select the best controls to include in X_i from the long list of weather and vegetation candidates, I implemented a LASSO approach. The LASSO works by minimizing the sum of squared residuals plus a penalty term based on the sum of all estimated coefficients. This method incentivizes LASSO to set some coefficients to zero. We can thus use the LASSO for model selection by observing to which control variables the LASSO assigns coefficients of zero, and then running the desired 2SLS IV regressions excluding those variables. However, there may be controls that should not be excluded, whether or not they are individually significant. For this paper, these necessary controls include year, month, and ecoregion dummies, and latitude and longitude. To run the LASSO with these controls already included, I regress the outcome variable on the desired included controls only, and then use the residuals as the new outcome variable for the LASSO. I also use a double debiased approach as recommended in Belloni et al.(Belloni et al. 2014): run the LASSO on the outcome variable and candidate control variables, excluding the treatment, and separately run the LASSO on the treatment and the candidate controls. Controls with non-zero coefficients in either result are included in post-LASSO regressions. Running both specifications helps avoid introducing omitted variables bias.

Finally, the results from Equation 1.2 can be used to estimate the cost of avoiding CO_2 emissions, which would have occurred without additional suppression effort. Using β_1 from

Equation 1.2, the acre-weighted average cost to avoid an additional ton of CO_2 emissions (the Cost per Ton, or CpT) is given by Equation 1.4.

$$CpT = \frac{A_i}{\Sigma A_i} \sum_{i=1}^{n} \left(\frac{C_i}{A_i}\right) * \left(\frac{1}{\beta * e_i}\right) \quad (1.4)$$

A_i is acres burned in fire i, C_i is the management cost of fire i, β is the effect on average fire size of increasing management expenditures by 1% (the coefficient of interest from the preferred regression specification, β_1), and e_i is the CO_2 emissions per acre burned. The emission factor e_i is a simple average of the emissions-per-acre in Alaskan fires in the ABoVe database over the period 1983-2016 (Walker et al. 2020), valued at 3.325 kgC/m^2, or 54.39 tons CO_2/acre.

1.5 Results

The results from the preferred IV model, reported in Column 2 of Table 1.1, indicate that all other variables constant, a 1% increase in fire management expenditures will result in an average decrease in burned area of 0.2063%. This result gives strong support to the argument that fire management is effective at reducing fire size and hence overall burned area, and that increasing management funding could help control the predicted increases in fire regimes and GHG emissions. Column 1 of Table 1.1 gives the OLS results, which are positive and demonstrate the reverse causality problem which necessitates the use of an IV specification. Importantly, this model is log-linear, so if the true relationship between management expenditures and burned acres is not isoelastic, the results might over- or under-predict the true effect of large changes in the management regime.

Table 1.1: Main Results

	OLS	IV
ln(total cost)	0.1587	-0.2063
	(0.0311)**	(0.0906)*
R^2	0.65	0.58
N	832	832

* *p*<0.05; ** *p*<0.01

Controls include mean and maximum values of FFMC, ISI, BUI, FWI, DSR, temperature, relative humidity, snow depth, and windspeed, mean DMC, maximum DC, elevation, white and black spruce coverage, deciduous coverage, grassland coverage, "burnable" coverage, latitude and longitude, and month and year fixed effects.

Table 1.2 gives the first stage results for the preferred regression specification using total costs and BLM costs. The F-stat for the preferred specification is 33.41, well above the common weak instrument cut-off of 10 (Staiger and Stock 1994). This suggests that FMZ is a high-quality instrument for fire cost.

Table 1.2: First Stage Results

	ln(total cost)	ln(BLM cost)
modified	2.3137	2.0842
	(0.3413)**	(0.3269)**
full	2.8448	2.6132
	(0.3201)**	(0.3119)**
critical	1.9782	1.8770
	(0.7519)**	(0.7541)*
R^2	0.62	0.61
F-stat	33.41	29.46
N	832	832

* *p*<0.05; ** *p*<0.01

Controls include mean and maximum values of FFMC, ISI, BUI, FWI, DSR, temperature, relative humidity, snow depth, and windspeed, mean DMC, maximum DC, elevation, white and black spruce coverage, deciduous coverage, grassland coverage, "burnable" coverage, latitude and longitude, and month and year fixed effects.

In addition to the primary specification, I run several robustness checks, the results of which are shown in Table 1.3. All of the alternative specifications yield results that are statistically indistinguishable from the main result, and all point estimates are within 0.05 of the main result. Column 1 displays the results of the preferred specification, run using pure BLM costs rather than the estimated total costs used in Table 1.1. Column 2 excludes all human-caused fires, which are likely to be highly correlated with FMZ and also may receive different responses or have other unobserved differences. Since 191 fires report zero suppression costs, I test two alternative data transformations. In the main specification, I add 1 to all costs before taking logs. This is a common approach and does not change the coefficient interpretations. In Column 3 of Table 1.3, I use the same specification, but add 0.01 before taking logs. Finally, in Column 4, I use an Inverse Hyperbolic Sine transformation on cost and acres instead of a log transformation.

Table 1.3: Alternate Specifications

	IV	Lightning Only	log(Cost + 0.01)	IHS Cost
ln(BLM cost)	-0.2268			
	(0.0997)*			
ln(total cost)		-0.2046	-0.1799	
		(0.0934)*	(0.0786)*	
IHS total cost				-0.2017
				(0.0885)*
R^2	0.58	0.54	0.58	0.58
N	832	762	832	832

* p<0.05; ** p<0.01

Controls include mean and maximum values of FFMC, ISI, BUI, FWI, DSR, temperature, relative humidity, snow depth, and windspeed, mean DMC, maximum DC, elevation, white and black spruce coverage, deciduous coverage, grassland coverage, "burnable" coverage, latitude and longitude, and month and year fixed effects.

Using the results from the preferred model specification, I find that the cost to avoid one ton of CO_2 emissions is $12.63 (in 2015 USD). The range using one standard error of the preferred coefficient is ($8.87, $22.52), and the 95% confidence interval is ($6.79, $90.76). As Figure 1.4 shows, the range is asymmetric about the mean and the upper bound is quite high, because the original regression coefficient is in the denominator of the formula and is quite small at the upper bound.

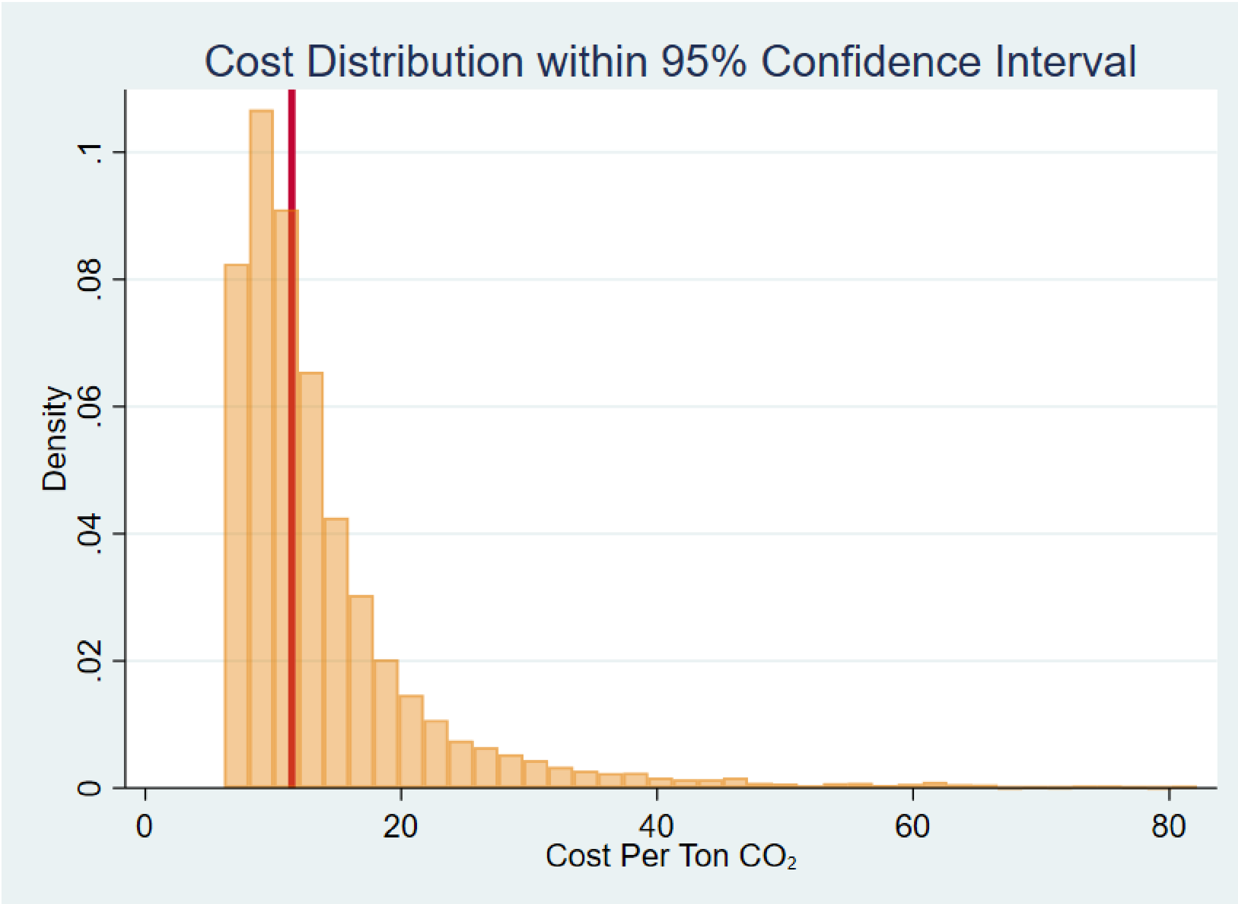

Figure 1.4: Simulated probability distribution of the cost per ton of avoided CO_2 emissions, with the central estimate of $12.63 highlighted by the vertical bar. I simulated a normal distribution of management effectiveness using the regression coefficient and its standard error, then used Equation 4 to generate a distribution of the cost per ton. Values are displayed within the 95% confidence interval.

1.6 Discussion and Conclusion

A severe fire year in Alaska can result in over 100 Gt CO_2 emitted, and these numbers are expected to worsen as climate change intensifies fire regimes, so it will be increasingly important for policymakers to understand the impact of uncontrolled fires and the potential for fire management to reduce these emissions (Wiedinmyer and Neff 2007, Kasischke et al. 2010). The results of this research suggest that wildfire management in boreal forests is an effective and low cost method for limiting burned area and emissions. In fact, the results in this paper likely underestimate fire management's climate mitigation potential, because current policy has not made the protection of high-carbon areas a management goal.

At a cost of only \$12.63 per ton of CO_2 emissions avoided, the results of this paper suggest that fire management compares favorably to other climate mitigation options. Mitigation approaches that rely on ecosystem dynamics, like soil sequestration using biochar, reforestation, and coastal wetland restoration are all estimated to cost more than \$100/ton CO_2eq (Griscom et al. 2017). Some methods of emission avoidance are more comparable, like subsidizing existing nuclear plants (\$18 – \$97 per ton CO_2 emissions avoided (Roth and Jaramillo 2017)) while others like Carbon Capture and Storage can cost several hundred dollars per ton depending on the method and type of power plant (Budinis et al. 2018). Furthermore, \$12.63 falls below even the baseline social cost of carbon estimated by William Nordhaus's DICE model for 2020, \$37.30 (Nordhaus 2017). Other estimates of the Social Cost of Carbon are considerably higher. For example, the US Interagency Working Group gives a central estimate of \$51 in 2020 (Interagency Working Group 2021), and the new GIVE model has a central estimate of \$185 (Rennert et al. 2022). Fire management for climate mitigation is not only cost-competitive with alternative approaches, its costs (\$12.63/tCO_2) are significantly exceeded by its climate benefits alone (\$37.30/tCO_2 at minimum).

This paper did not quantitatively consider any benefits of fire management other than reducing CO_2 emissions, but this should by no means be interpreted to mean that this is the only benefit of reducing fire size. The negative impact of fires on health is well documented (Jones et al. 2016, Kochi et al. 2016, Thomas et al. 2017). There are also a number of greenhouse gasses generated during combustion other than CO_2 which can increase warming effects, including methane (CH_4), as well as black carbon particles which can increase surface albedo and melting, and additional emissions which take place for around five years after the initial fire pulse (Randerson et al. 2006). Other factors like indirect aerosol impact on cloud occurrence and lifetime land surface albedo, and forest regrowth can decrease net warming effects, although many of the impacts are local-regional as opposed to global, or short-lived, and do not counteract the overall impact of CO_2 and CH_4 on long term global climate (Oris et al. 2014). It is also important to remember that fires impose a number of other damages to human health and economies, which are included in this analysis.

Conversely, this paper should not be interpreted as advocating for the suppression of all fires. Fires play a very important ecological role, including reducing fuel buildup, habitat, controlling insects and pathogens, releasing nutrients, and restarting the clock on ecological succession (Cudmore 2008). Rather, this research can help provide the foundation of a response to the trend of intensifying fire regimes due to climate change. Understanding the effectiveness of fire management is an important step toward understanding the role that fire management can play in maintaining boreal ecosystems and their carbon output at historical levels. This research finds that wildfire suppression in Alaska is capable of effectively reducing the size of actively managed wildfires, and that such suppression efforts are a relatively low-cost way of avoiding carbon emissions. Combined, these two results suggest that Alaskan wildfire management has significant potential as climate mitigation policy.

Chapter 2: The Costs and Benefits of Fire Management for Carbon Mitigation in Alaska Through 2100[5]

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Abstract

Climate change is intensifying fire regimes across boreal regions, and thus both burned area and carbon emissions from combustion are expected to increase significantly over the next several decades. Fire management through initial suppression of fires is effective at reducing burned area, but limited work has addressed the role that fire management can play in reducing wildfire carbon emissions and their impacts on climate change. In this work, we draw on historical data covering fire and fire management in Alaska to project burned area and management outcomes to 2100. We allow management to both respond to and impact variations in annual burned area and carbon emissions, while keeping decadal-average burned area at or above historical levels. The total cost of a fire is calculated as the combination of management expenditures and the social cost of carbon emissions during combustion, using the Social Cost of Carbon framework. Incorporating the tradeoff between management expenditures and burned area, we project that by 2100, increasing management effort by 5-10 times relative to current expenditures would minimize combined management and emissions costs. This is driven by the finding that the social costs of carbon emissions greatly exceed management costs unless burned area is

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constrained to near the average historical level. Our analysis does not include the many health, economic, and non-CO_2 climate impacts from fires, so we likely underestimate the benefits of increased fire suppression and thus the optimal management level. As fire regimes continue to intensify, our work suggests increased management expenditures will be necessary to counteract increasing carbon combustion and lower overall climate impact.

2.1 Introduction

As climate change alters temperature and weather patterns globally, the frequency of warm and dry conditions is increasing in the boreal region, intensifying boreal fire regimes in Alaska (Kasischke et al. 2010, Turetsky et al. 2011), Canada (Gillett et al. 2004, Hanes et al. 2019), and Siberia (Ponomarev et al. 2016, Shvetsov et al. 2021, Talucci et al. 2022). Burned area in Alaska is projected to at least double by the end of the century, barring dramatic changes in emissions trajectories (Bachelet et al. 2005, Balshi et al. 2009, Euskirchen et al. 2009, Rupp et al. 2016, Pastick et al. 2017, Genet et al. 2018, Schultz 2019), suggesting Alaska's boreal forests may soon become a net source of carbon to the atmosphere (Genet et al. 2013), if they are not already (Zhu and McGuire 2016, Virkkala et al. 2021). These trends pose a serious problem as nations seek to limit global average temperature increase to 1.5 ºC above preindustrial levels (Secretariat 2015). If unmitigated, wildfire carbon emissions through 2050 from Alaskan and Canadian wildfires alone have the potential to reduce the world's remaining 1.5 °C global carbon budget by roughly 4% (Phillips et al. 2022). Active fire management is a potentially important tool to mitigate the effects of climate change on boreal fire regimes, but the costs, benefits, and trade-offs of fire management have yet to be quantified.

While fire management is effective at containing fires and reducing burned area, it can be resource intensive. The vast aerial coverage and remoteness of most boreal forests in North America render fuels treatments impractical for managing burned area (Amiro et al. 2001), but post-ignition management is effective at reducing burned area (Phillips et al. 2022), particularly when fires are attacked early (Cumming 2005, Arienti et al. 2006, Podur and Wotton 2010). Fighting wildland fires requires significant resources, including aircraft, ground vehicles, firefighting crews, and teams handling management and operations (AICC 2020). The cost of managing an individual fire in Alaska can extend into the hundreds of thousands or even millions of US dollars (AICC 2021).

While fire management can be costly, the emissions from combustion of vegetation and soil organic matter also generate large economic damages by contributing to climate change. The Social Cost of Carbon (SCC) is typically used to quantify such damages (Nordhaus 1993, Stern et al. 2006, IWG 2010, Arrow et al. 2013, National Academies of Sciences 2017). To calculate the SCC, a small pulse of carbon or CO_2 is "emitted" in an Integrated Assessment Model (IAM), leading to a change in global average temperatures and subsequent economic damages. The discounted sum of these damages represents the social cost of the emissions pulse. SCC is commonly used in benefit-cost analyses for policy to value the cost imposed by emissions (IWG 2010, Greenstone et al. 2020, IWG 2021, Metcalf and Stock 2017, Nordhaus 2017, Anthoff and Emmerling 2019, Tol 2019, Russell et al. 2022). In the Alaskan context, applying the SCC to wildfire emissions allows us to compare the operational costs of fire management to the climate costs of intensifying fire regimes.

To date, the role of boreal fire management as a potentially cost-effective tool to limit future carbon emissions has barely been explored. A limited number of papers attempt to project management costs based on burned area projections, but do not incorporate any effect of increased expenditures on burned area (Melvin et al. 2017, Schultz 2019). As a result, we lack an understanding of 1) how changes in fire management may alter climate-driven increases in burned area, and 2) the economic costs and benefits of adjusting management regimes in response to climate-driven pressures.

Here, we apply the SCC to wildfire emissions in Alaska to compare the operational costs of fire management to the economic costs of carbon emissions from intensifying fire regimes. Carbon emissions and sequestration are already recognized as a potentially important component of wildfire and land management policy (Mills et al. 2015, Haight et al. 2020, Sánchez et al. 2021), although wildfire emissions exacerbated by climate change are not included in SCC estimates. We present a modeling framework that combines projections of burned area from the fire science literature with fire management that is both responsive to changing fire regimes and capable of reducing burned area. We then use this framework to project burned area in Alaska through the end of the century under various management scenarios. The model integrates the direct financial cost of management efforts with the social cost of carbon emissions during combustion to project optimal fire management levels through 2100. This provides an economic framework to assess how changes in wildfire management can be used as a climate mitigation tool.

2.2 Methods

To estimate the management costs and carbon benefits of fire management in Alaska, we first developed a stochastic model, based on models presented in Rogers et al. (2013) and Veraverbeke et al. (2017), that simulates historical burned area patterns in Alaska based on ignition and fire size probabilities (2.1 Parameter Selection Model). We then integrated estimates of the effectiveness of fire suppression on limiting fire size based on agency cost data from Alaska (Phillips et al., 2022), and used this model to simulate the impacts of climate change and altered management regimes on burned area, management costs, and social costs from fire carbon emissions (2.2 Projection Model). The overall study design is illustrated in Figure 2.1.

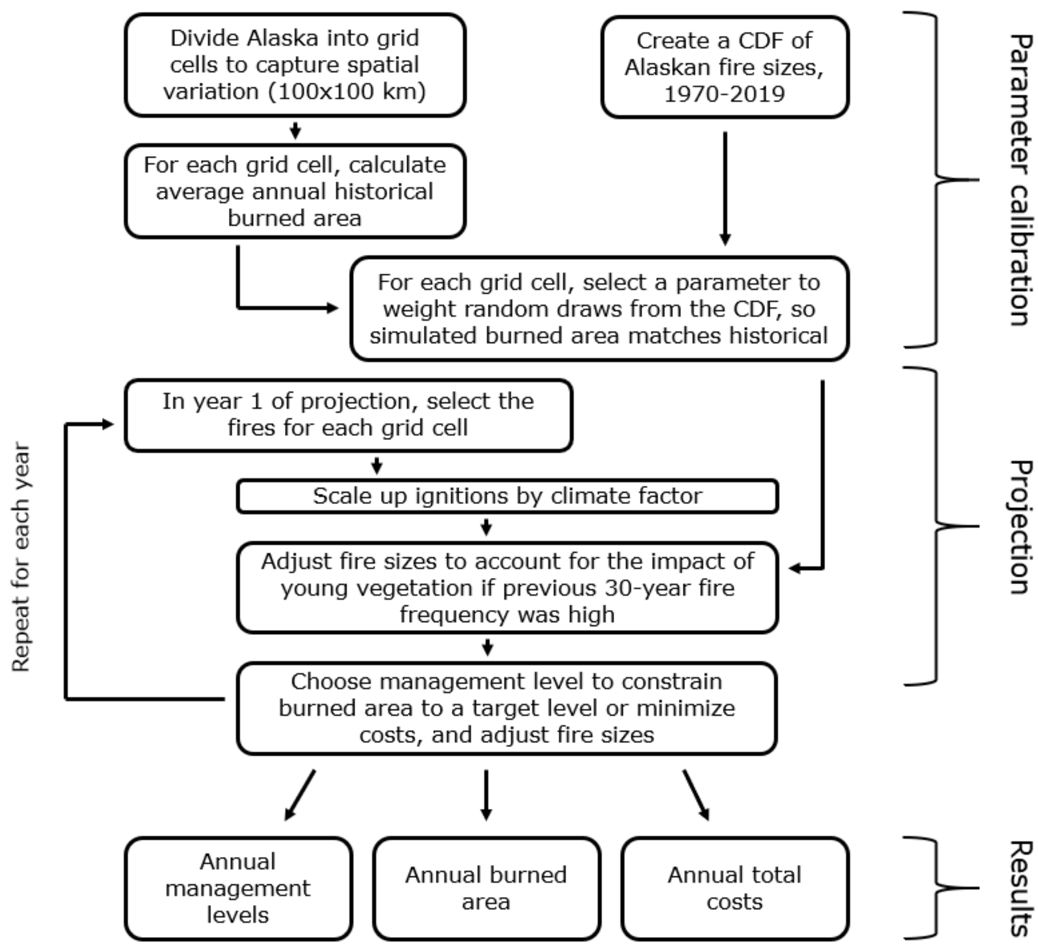

Figure 2.1: Summary of methods. CDF stands for Cumulative Density Function.

2.2.1 Parameter Selection Model

We first divided Alaska into 191 grid cells (100 x 100 km) and assigned fires from the historical record into cells based on their ignition location (Fig. 2.2). The historical record includes fires from 1970-2019 published by the Alaska Interagency Coordination Center (AWFCG 2020). Each fire is represented by a point at the ignition site, and the observation includes the size of the fire. We excluded fires from before 1970 because the existing data include only an estimated 43% of true burned area (Kasischke et al. 2002). Grid cells with no burnable area (containing only categories of “barren” or water) were excluded from further analysis using the Landfire land cover mask (LANDFIRE 2008). We then generated a cumulative distribution function (CDF; Fig. 2.3) of the sizes of all fires across the state, which included over 25,000 observations. 90% of fires were under 200 ha, but there were a small number of very large fires (>200,000 ha), including 29 fires of over 1 million ha.

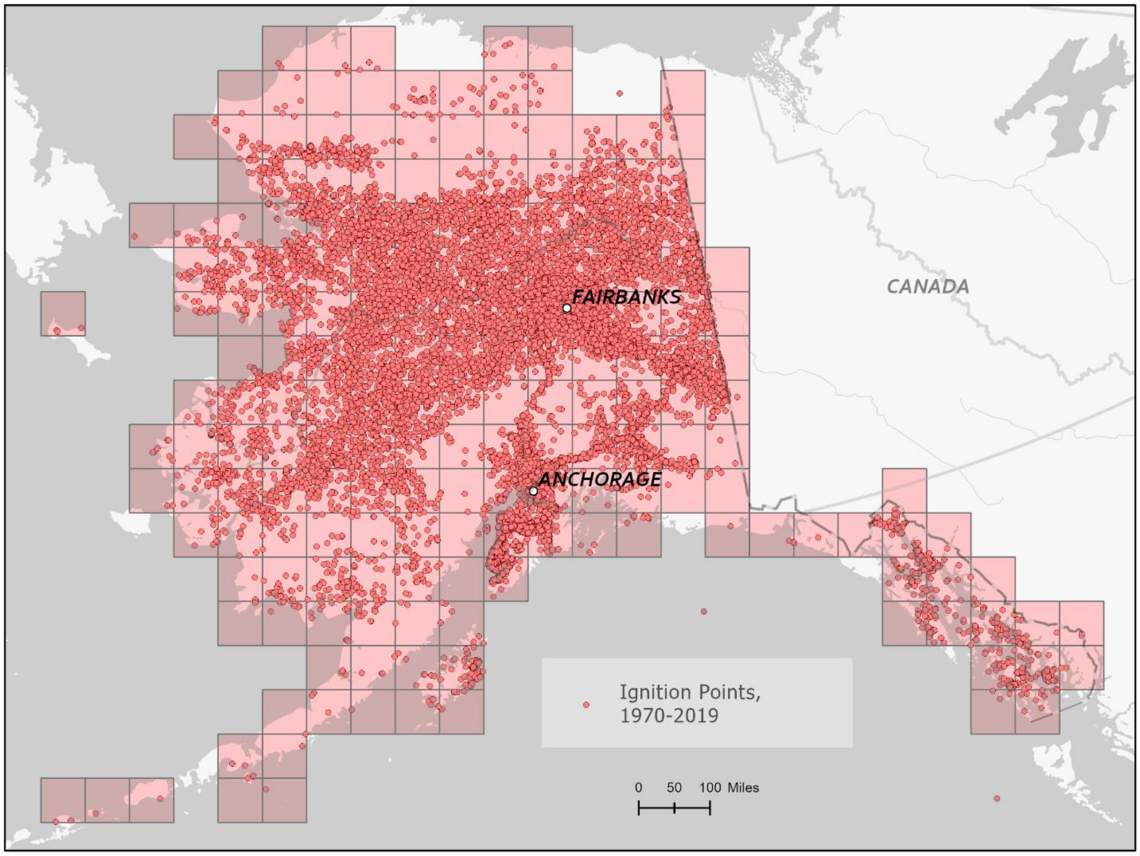

Figure 2.2: Ignition points and analysis grid.

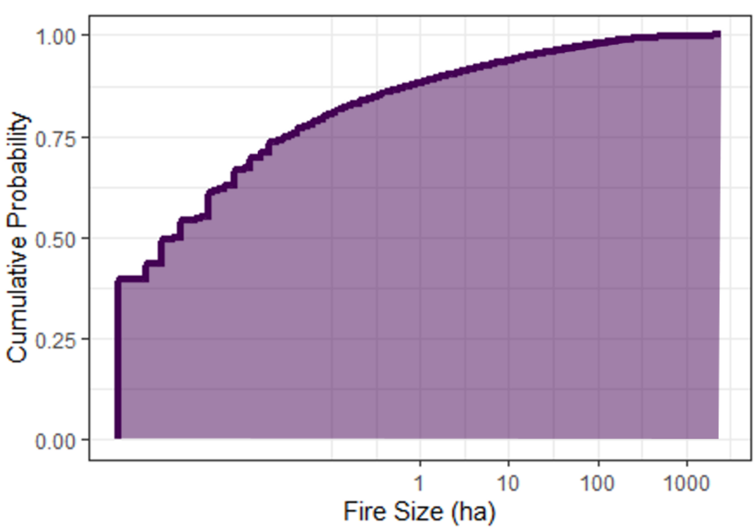

Figure 2.3: Cumulative Density Function of all fire sizes 1970-2019. Note the log scale for fire size.

Our parameter selection model operates by randomly choosing the number of annual ignition events and matching fire sizes for each grid cell in each simulated year for 10,000 years, yielding a simulated annual burned area. Ignitions are drawn randomly with replacement from a grid cell's record of annual ignitions. For each ignition we generate a random number between 0 and 1, and then match it to the nearest fire size in the CDF for the entire state of Alaska, yielding a simulated size. The goal of the parameter selection runs is to derive a scaling factor specific to each grid cell such that the resulting long-term annual burned area closely resembles that of the historical record.

To derive this scaling factor, we compared the average annual burned area to the observed historical average for each grid cell after each simulation. For most grid cells, the average fire size is not equal to the state's average, so the initial simulation did not match the observed historical average. We then assigned an exponent to modify the random number that selects fire sizes from the CDF. A large exponent parameter decreases the random number toward 0, whereas an exponent between 0 and 1 increases the random number toward 1. We iteratively increased or decreased this parameter until the simulated annual average was within 5% of the observed historical average burned area for each grid cell. Exponents ranged from 0.025 to 215, with a median of 2.8375. This approach allowed us to match the historical record without being confined to the available historical variation in fire sizes within a particular grid cell.

2.2.2 Projection Model

After parameters were optimized for each grid cell, we used the model to project fires across Alaska from 2020 to 2100 by increasing burned area due to climate change and incorporating different fire management approaches, as discussed below. Although our parameter selection approach resulted in long-term burned area patterns that matched the historical mean, there are also temporal trends in the historical record (e.g., Veraverbeke et al., 2017). To address this in our projection model, we first divided the data into two periods: 1970-1989 ("pre-climate regime"), and 1990-2019 ("climate change-influenced") (following Kasischke et al. 2010), and estimated the temporal trends in the historical data using a simple linear regression of time on fire size (Table S2.2 in Appendix 2.2). In the first period, there was no significant temporal trend in either ignitions or fire size. In the second period, average fire size increased by an additional 3.37% for each year after 1990, significant at the 0.001 level, with no significant trend in ignitions. We therefore multiplied all the fire sizes generated for the projection model by 1.0337*(2019-1996), a factor of 1.76, beginning in 1996, the ignition-weighted mean year. While this scaling factor results in a slight upward trend in historical burned area, our pre-management burned area at the beginning of the projection period remains consistent with the historical trend. We also adjusted for the dampening effect of recent fires within a particular grid cell on fire size, following the results in Rogers et al. (2013); details are provided in Appendix 2.1.

To account for the influence of climate change on Alaska's burned area during the 21st century, we conducted a literature survey, with methods detailed in Appendix 2.1. We ultimately included five peer reviewed papers (Bachelet et al. 2005, Balshi et al. 2009, Euskirchen et al. 2009, Pastick et al. 2017, Genet et al. 2018) and two government reports (Rupp et al. 2016, Schultz 2019), containing a total of 23 projection scenarios using three fire models (Table S3 in Appendix 2.2). While the magnitude of the increase in burned area varied, all the papers estimated that climate change will increase burned area in Alaska.

Giving each paper equal weight, the mean percent increase in burned area from 2020 to 2100 was 113% (Table S2.4 in Appendix 2.2). Assuming a linear trend as in Phillips et al. (2022), we increased the number of ignitions by adding 1.4% of the historical ignition frequency each year, which then matched the total increase in burned area from the literature by 2100. This approach recognizes the role of increased lightning in changing fire regimes, since lightning strikes are the primary source of wildland ignitions and are projected to more than double by the end of the century (Veraverbeke et al. 2017, Bieniek et al. 2020, Chen et al. 2021). After drawing the number of ignitions from a grid cell's historical record, we scaled it by a factor of (1 + 0.014*(year – 2019)), rounded to the nearest whole number, to derive the climate-adjusted number of ignitions.

Once ignitions and fire sizes were calculated for all grid cells for a given year, we chose the amount of fire management (suppression) to implement across the state. On average, a 1% increase in Alaskan management expenditure from current levels corresponds to a 0.2063% decrease in average fire size (from Phillips et al. 2022, summarized in Appendix 2.1). All Alaskan land is divided into one of four Fire Management Zones, or FMZs (Fig. 2.4a), each of which has its own standard response to fire. In land designated Critical, fires threaten human life, inhabited property, or infrastructure, and thus receive highest priority for management resources (AWFCG 2019). The management goal is complete suppression. In the Full zone, fires threaten uninhabited structures or areas that are used for resource extraction or valued for their cultural significance (DOF 2015). Fires in areas designated Full receive full suppression effort if the resources are not needed for fires in Critical zones. Limited areas are remote, and at most include points (like seasonal hunting cabins) that require protection. The standard response is to monitor fires and perform point protection. Finally, fires in Modified areas are treated like fires in Full areas until early to mid-July, after which they receive the same minimal protection as other Limited areas, since late-season fires are less likely to spread enough to threaten priority sites (AWFCG 2019). The more actively managed FMZs receive both more dollars per designated acre and more dollars per acre burned compared to the less actively managed FMZs.

Since more actively managed zones receive more dollars per acre, grid cells with a large proportion of highly managed acres should receive a greater share of any increases in statewide management spending. We therefore calculated weights to apply to each FMZ using two data sources: the management costs from Phillips et al. (2022), which are extrapolated from spending reported by the Bureau of Land Management, and the costs from Melvin et al. (2017), which include only spending by federal agencies. We calculated a spending ratio for each FMZ defined by the fraction of dollars spent per FMZ to the fraction of total land in that FMZ. We then applied these spending ratios to each grid cell in proportion to its relative coverage by each FMZ. The results from the two datasets are broadly similar (Table 2.1). We used the results from Melvin et al. because the year range is larger and ends closer to the present, and we assumed 2020 FMZ boundaries for future projections (BLM Alaska 2021). Based on these FMZ weights and the proportion of land in each FMZ designation by grid cell, we estimated the expected percent increase in management spending in each grid cell from a 1% increase in total spending.

FMZ	Costs from Melvin et al. 2017 Area (Square Km)	Response Costs, Millions 2015$	Dollars per Square Km	Fraction of dollars spent	Fraction of area	Weight
Critical	116,158	29	250.36	0.09	0.01	6.53
Full	1,109,243	157	141.88	0.48	0.13	3.70
Modified	800,358	39	48.32	0.12	0.09	1.26
Limited	6,505,341	102	15.68	0.31	0.76	0.41
	Total reported federal response costs, 2002-2013					

	Costs from Phillips et al. 2022 Area (Square Km)	Response Costs, Millions 2015$	Dollars per Square Km	Fraction of dollars spent	Fraction of area	Weight
Critical	45,837	9	200.84	0.06	0.01	8.23
Full	798,667	89	111.44	0.57	0.12	4.57
Modified	567,135	29	50.84	0.18	0.09	2.08
Limited	4,986,704	29	5.84	0.19	0.78	0.24
	Estimated total response costs, 2007-2015					

Table 2.1: Fire Management Spending by FMZ

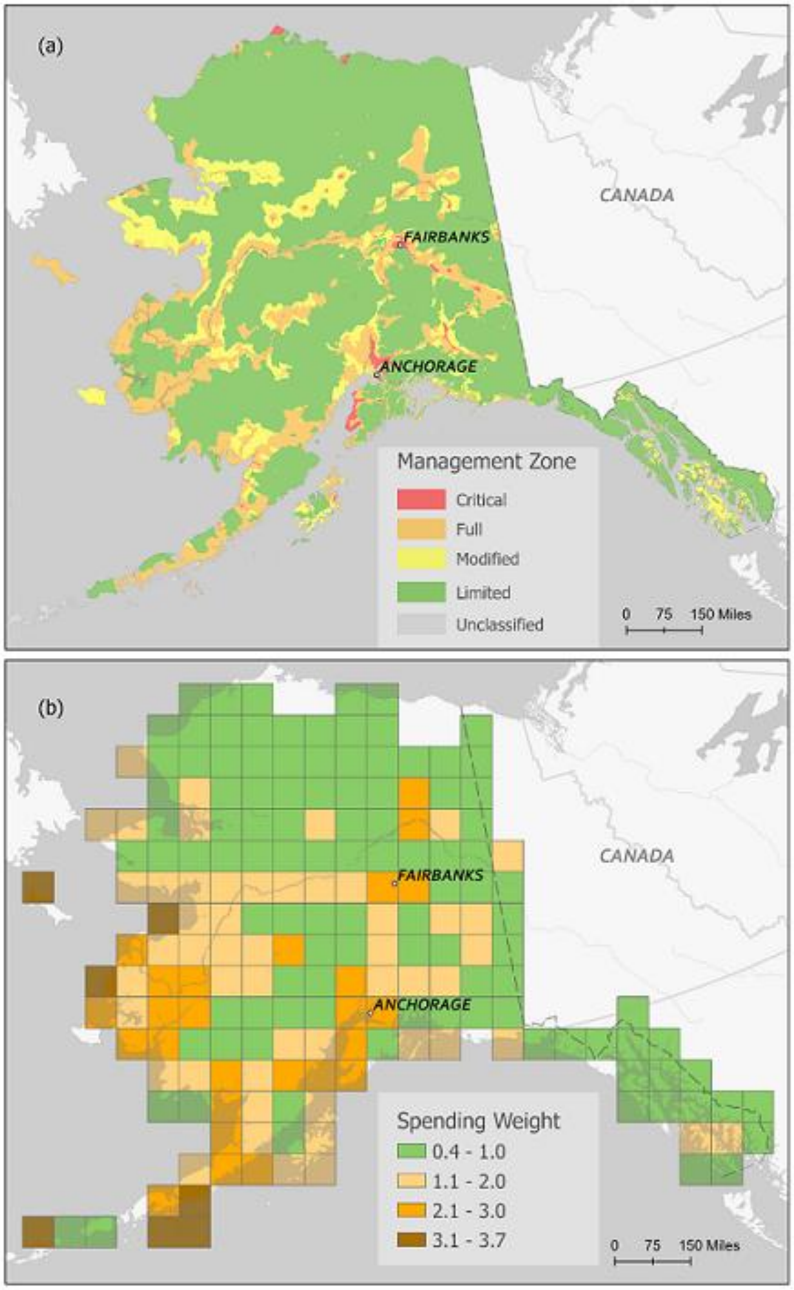

Figure 2.4: Map of Fire Management Zone (FMZ) designations for year 2020 (a) and spending weight per grid cell based on FMZ (b).

The main specification of our projection model (Equation 2.1) is the selection of state-wide management effort each year to minimize combined management and the Social Cost of Carbon, while constraining the ten-year moving average of burned area at or above historical levels, as defined in Equation 2.2, to avoid the ecological consequences of reduced or eliminated fire regimes.

(2.1) $$\text{Obj} = min\left\{\sum_{grid\ cell=i} argmin\begin{pmatrix}\frac{Spending\ weight_i}{\sum Spending\ weight_i} * C * M\ , \\ \frac{Spending\ weight_i}{\sum Spending\ weight_i} * C * S\end{pmatrix} + \left(BA_{year} * CO_2 * SCC_{year}\right)\right\}$$

(2.2) $$\left(\sum_{year=Y-9}^{Y} BA_{year}\right) / \ 10 \geq BA_h$$

Spending weight$_i$ is change in grid cell spending relative to state spending; *C* is the baseline cost of management; *M* is the level of statewide management, with 1 representing no change from baseline; *S* is the grid cell management saturation point; BA_{year} is the statewide burned area in the year under consideration; CO_2 is an estimate of carbon emissions per ha burned; SCC_{year} is the SCC in the year under consideration; and BA_h is the average historical level of burned area. The total costs from a fire season are the sum of management costs across all grid cells, which vary based on FMZ distribution and are defined according to the grid cell spending weight described above and summarized in Fig. 2.4b, and the societal harm caused by carbon emissions during combustion, quantified using the SCC. The baseline cost of management is the average annual combined federal and state management cost from 2009-2015, or $99,386,015 in 2015 dollars (Jandt, personal communication). The social cost is positive when annual burned area exceeds the historical average, and negative when annual burned area is less than the historical average, implying a social benefit from fewer than average emissions. In historical fire regimes, this approach yields costs averaging close to zero over time, but under intensifying fire regimes this will result in positive and growing social costs in the absence of increased management. For the amount of CO_2 emitted per square kilometer burned, we use the average from Alaskan sites included in the ABoVE combustion database (Walker et al. 2020). The average carbon emissions from Alaskan fires in the database, which covers the period 1983-2016, are 3.325 kgC/m^2 or

102.4 metric tons CO_2/ha (CO_2), after accounting for the fraction of carbon that does not convert to CO_2 (Akagi et al. 2011, Phillips et al. 2022). We used SCC values from the DICE (Dynamic Integrated Climate-Economy) model (Nordhaus 2020) run under a Business As Usual (BAU) climate and policy pathway with a 3.9 ºC temperature increase relative to preindustrial levels, to model costs through 2100. Further explanation of our SCC choice can be found in Appendix 2.1. This temperature increase is within the likely range for the most recent IPCC estimates for "high" and "very high" emissions scenarios, and is between the best-estimate values for the two scenarios (IPCC 2021).

Management in each grid cell is bound above by a saturation level, which is based on the management intensity from that grid cell's mix of FMZs. At the saturation level of management, burned area is reduced to near-zero (5% of pre-intervention level) in the grid cell, and further state-wide management increases cannot decrease burned area or increase cost in that grid cell.

In the primary specification of our model, management was chosen to minimize total annual costs. The BAU specification kept management at its current level, while including the impact of climate through the simulation period. In addition, we ran several alternate specifications to test the robustness of our modeling framework (Table 2.2). We included a scenario that halved management effectiveness (Halved Effectiveness), since management is less effective when resources are stretched thin such as during large fire years and when fire weather is more intense. Another scenario halved the SCC applied to fire emissions (Halved SCC) to adjust for the fact that post-fire forest regrowth makes combustion emissions less persistent and thus may decrease the cumulative economic damages relative to GHG emissions from fossil fuel combustion. We also included scenarios where management was chosen to approximate a prespecified burned area level, either historical levels (approximately 0.2 Mha per year, 1970-89, Historical BA Target) or recent levels (approximately 0.51 Mha per year, 2010-2019, Modern BA Target) using the 10-year average burned area. Finally, we modeled cost-minimizing management under a range of climate scenarios: minimum, median, maximum, and 25th and 75th percentile burned area projections from the literature survey rather than the mean.

Category	**Specification Name**	**Description**
Primary Specification	Cost Minimizing Mgmt	Optimize management annually to minimize total costs, while keeping 10-year average burned area at or above historical levels. Full SCC, full management effectiveness, mean climate impact on fire regimes
BAU	BAU Mgmt.	Maintain management at current levels, with mean climate impacts
Burned Area Target	Modern BA Target	Choose management annually so that 10-year burned area average matches the average from 2010-2019, 0.51 Mha
	Historical BA Target	Choose management annually so that 10-year burned area average matches the average from 1970-1989, 0.2 Mha
Cost Sensitivity	Halved SCC	As in Primary, optimize management annually to minimize costs, with halved SCC value. Accounts for the fact that post-fire forest regrowth makes combustion emissions less persistent and thus may decrease the cumulative economic damages
	Halved Effectiveness	As in Primary, optimize management annually to minimize costs, with halved management effectiveness. Accounts for the fact that management may be less effective when resources are stretched thin, and when fire weather is more intense, in large fire years
Climate Sensitivity	Min Climate	As in Primary, but with minimum climate impacts from the literature survey (9% increase in burned area 2020-2100)
	25 Pct Climate	As in Primary, but with 25th percentile climate impacts from the literature survey (37% increase in burned area 2020-2100)
	Median Climate	As in Primary, but with median climate impacts from the literature survey (67% increase in burned area 2020-2100)
	75 Pct Climate	As in Primary, but with 75th percentile climate impacts from the literature survey (114% increase in burned area 2020-2100). Note this is very similar to mean, 113%
	Max Climate	As in Primary, but with maximum climate impacts from the literature survey (530% increase in burned area 2020-2100)

Table 2.2: Projection Scenarios

2.3 Results

In our primary, cost minimizing specification under mean climate influence (a 3.3 ºC increase from preindustrial temperatures, leading to a 113% increase in burned area 2020-2100; Fig. 2.5), a major increase in management spending constrained burned area to near-historical levels throughout the projection period. Combined social and management costs rose throughout the 21st century, driven primarily by the social costs of emissions (Fig. 2.6), despite burned area remaining near historical levels. Rising social costs were driven by both a slight increase in burned area over the period, and SCC growth from $35/ton in 2020 to $225/ton by 2100, a standard feature of SCC estimates driven by both increasing marginal damages and closer future damages. In contrast, under BAU management, burned area doubled from its current level to over 1 Mha per year. Total costs, driven almost exclusively by the social cost of emissions, exceeded $20B annually by the end of the period, nearly 10 times the costs associated with an optimal management regime.

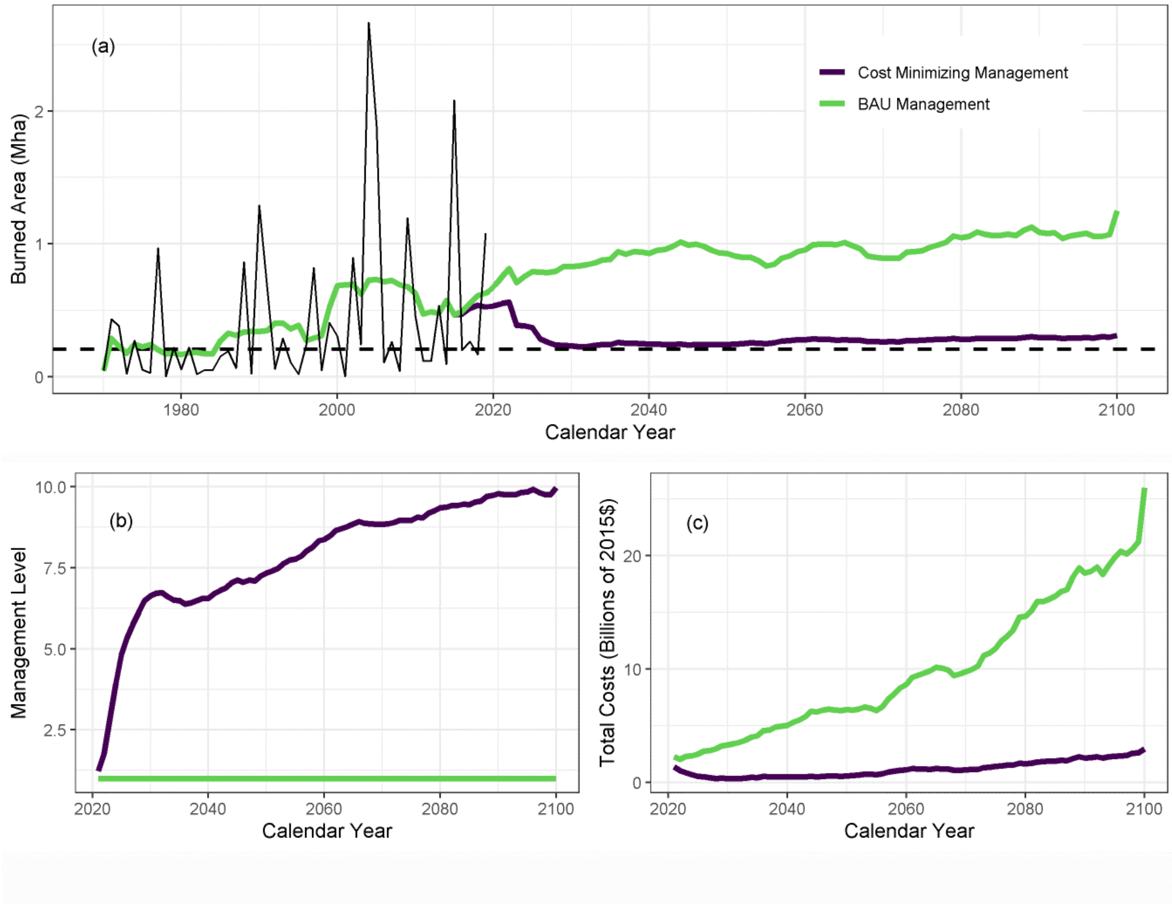

Figure 2.5: Management level, burned area, and cost outcomes for primary cost-minimizing specification and BAU specification. The historical data, with moving average, show that annual burned area in Alaska has already increased relative to the historical baseline (dashed line) (a, left). Projected burned area continues to increase under BAU management (green line) but remains near historical levels under optimal management (a, right). Optimal management levels increase over the projection period (b). Projected total costs are shown in billions of 2015 dollars (c). All values display the 11-year moving average, centered on the year displayed.

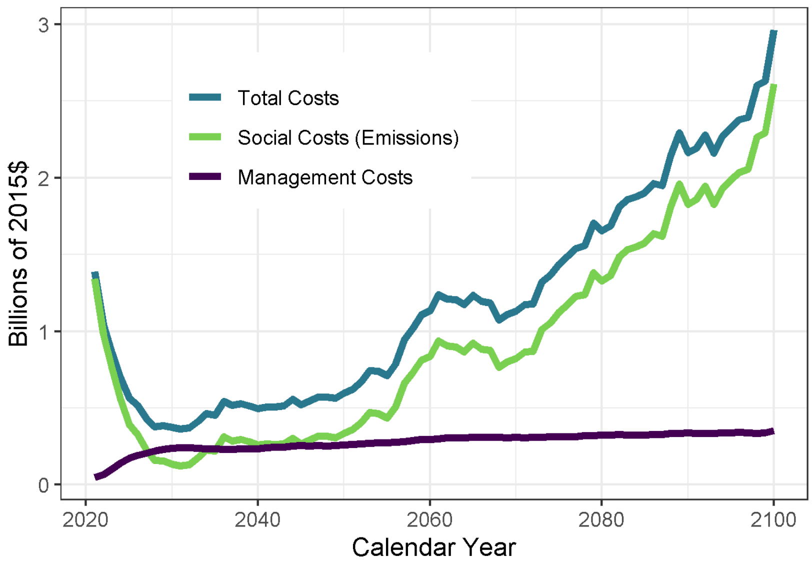

Figure 2.6: Total costs broken into social and management costs for primary cost-minimizing specification. All values display the 11-year moving average, centered on the year displayed, with shorter moving averages at the beginning and end of the time series.

When we compared these results to our alternate specifications (Fig. 2.7), we found our conclusions are robust to model formulation and assumptions. Changing the SCC or management effectiveness had a relatively small effect on optimal burned area and costs, although management increased significantly to compensate for lower effectiveness. Most alternate climate scenarios showed similar results, although the model struggled to contain burned area under the maximum climate influence scenario. Non-cost-minimizing scenarios generally resulted in lower management levels and substantially higher burned area and total costs.

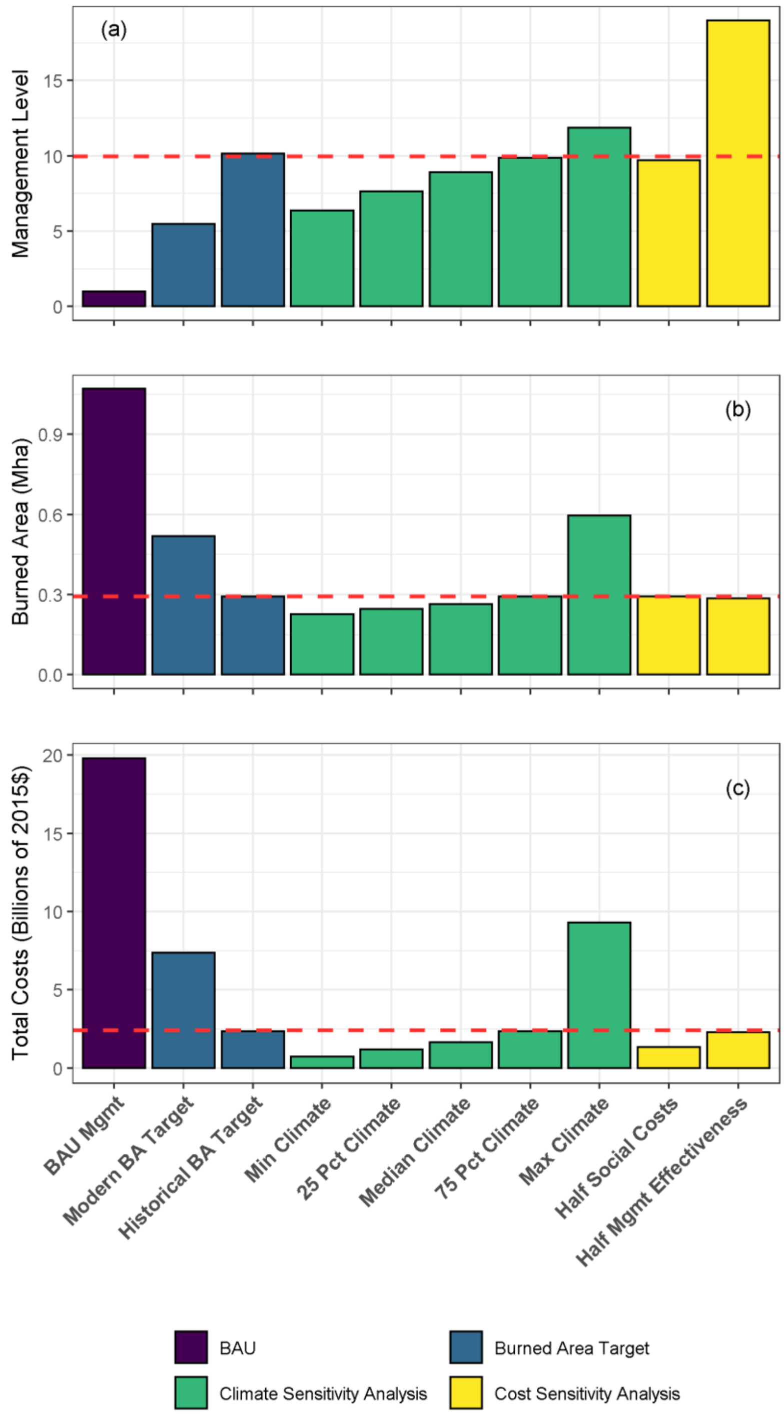

Figure 2.7: Comparison of primary specification results averaged over the final decade of projection (red dashed line) to alternate specifications, including average management levels in the period 2090-2100 across specifications (a), average burned area (Mha) (b), and total costs (billions of 2015 dollars) (c). Management level gives the amount of management expenditure relative to baseline; for example, a management level of 2 means doubled expenditure relative to baseline. Alternate specifications are described in Table 2.2.

2.4 Discussion

As fire regimes intensify, fire management spending, and by implication the resources available for fire management, will need to increase significantly if wildfires and their associated emissions are to be constrained. To limit the climate damages from wildfire carbon emissions, management spending may need to increase roughly tenfold by the end of the century under a high global emissions scenario. Even under the minimum climate impact scenario where burned area increases by only 9% from 2020, optimal management may need to increase more than fivefold to return burned area to historical levels. While this represents a large investment relative to historical spending, particularly under more pessimistic climate-fire scenarios, Alaska receives on average only 4% of U.S. federal resources for wildfire management despite representing a fifth of US land area and over half the nation's carbon emissions from wildfires (Phillips et al. 2022). The cost per ton of CO_2 emissions avoided by additional management is also low relative to other climate mitigation measures (Phillips et al. 2022). As described below, our analysis likely underestimates the harms from fires in several ways. Taken together, these suggest that our estimate of optimal management may in fact be more accurately regarded as a lower bound.

2.4.1 Additional Social Benefits of Increased Fire Management

This analysis does not attempt to include every dimension along which fire and fire management can affect human welfare, and nearly all the effects of increased management and reduced burned area that are excluded from our model are benefits to society. During large fire events in the North American Arctic, air quality can exceed the EPA's highest danger rating due to smoke (Trainor et al. 2009), with exposure extended across Canada, the conterminous US, and even to Europe (Le et al. 2014, Brey et al. 2018), although Indigenous communities in the Arctic are often disproportionately affected. Smoke exposure can lead to elevated rates of asthma, chronic obstructive pulmonary disease, bronchitis, pneumonia, respiratory infection, cardiovascular morbidity, and all-cause mortality (Liu et al. 2015, Reid et al. 2016, Cascio 2018).

In addition to their health impacts, wildfires often involve direct infrastructure losses, as well as impacts on forage and denning habitat for many animal species that affect subsistence resources for Indigenous communities (Nelson et al. 2008, Kuuluvainen and Gauthier 2018). Wildfires can also cause direct deaths and injuries, disrupt transportation networks and supply chains, reduce water quality and erosion control, depress housing markets, and decrease revenue from recreation and tourism (Thomas et al. 2017). Since most of this literature takes a total damages approach rather than estimating damages from marginal increases in burned area, a reasonable approach since many of these damages are highly locationally dependent, we were not able to incorporate these costs into our model framework. However, to provide a sense of scale, total damages from wildfire in the US were recently estimated at $64B - $285B per year. This is roughly 0.3 - 1.5% of US GDP and an order of magnitude more than what is spent on fire prevention and preparedness, fuels management, and active suppression (Thomas et al. 2017).

We also likely underestimate the benefits of additional management due to reductions in other climate impacts from wildfire. We use only estimates of the CO_2 released during combustion, but there are numerous other GHGs released that contribute to climate forcing (Randerson et al. 2006, Ward et al. 2012), including CH_4, N_2O, O_3, and O_3 precursors including NOx, non-methanogenic volatile organic carbons, and CO (Akagi et al. 2011, Ward et al. 2012, Huang et al. 2016, Wiggins et al. 2016). These collectively can more than double the radiative forcing from boreal fires compared to CO_2 alone (Huang et al. 2016). Fires also release organic and black carbon aerosols, which can impact the atmospheric radiative budget and generate large positive surface forcings when deposited on snow and ice (Flanner et al. 2007, Kostrykin et al. 2021). Moreover, by removing insulating organic layers, combustion can initiate permafrost degradation, leading to subsequent GHG emissions (Treharne et al. 2022, Genet et al. 2013, Jafarov et al. 2013, Holloway et al. 2020). Each of these wildfire impacts increases the benefits of management, further suggesting that our cost-minimizing specifications may considerably underestimate the optimal level of management.

Finally, we quantify emissions costs using the SCC, which has faced criticism for underestimating the likelihood of or altogether excluding potential large feedbacks like permafrost thaw (Weitzman 2014, Weitzman 2020) or intensifying fire regimes and their attendant emissions, and relying on outdated estimates of economic damages (Carleton and Greenstone 2021). Hence, the social cost of the combustion emissions we include could itself be an underestimate. Other critiques of SCC, like disagreements over the appropriate discount rate (Dasgupta 2008, Arrow et al. 2013), failure to incorporate intersectoral and interregional feedbacks (National Academies of Sciences 2017), and objections to the basic modelling approach (Pindyck 2017), add uncertainty to our estimation of the social cost of fire emissions but do not clearly bias our estimate up or down. Moreover, while SCC has been applied to forest carbon sinks and fire emissions in previous literature (e.g. Mills et al. 2015, Haight et al. 2020), forest regrowth adds a novel challenge to its application. While we attempt to address this by including a halved SCC scenario, this issue merits further study.

2.4.2 Evolving Fire Regimes and Other Caveats

To make our model tractable, we relied on several simplifying assumptions. We limited the influence of climate to the number of ignition events, and the influence of fire management to post-ignition suppression, although climate is predicted to impact ignitions, fire size, and intensity. We also selected pre-climate fire sizes solely to match historical burned area, although in reality fire size is determined by regional climate, fire weather, vegetation, topography, spatial patterns of land cover, and human influence. Our model also relies heavily on a single estimate of management's effectiveness at reducing fire size, although management's effectiveness is also dependent on many of the factors above. Finally, we optimized annually, rather than across the entire projection period (see Appendix 2.1 for further discussion). We adopted these simplifications because the goal of this work is to model the effects of climate and management on burned area, not to accurately reproduce every facet of the ecological system.

Despite these caveats, our model can replicate both historical burned area trends and projections from the literature while integrating responsive and effective management. Under more optimistic climate trajectories, for example those closely aligned with the 1.5 ºC target established in the Paris Climate Agreement, such a drastic shift in management policy may not be required.

Although our model accounts for lower flammability in younger boreal forests, it may not account for the interacting ways in which shorter-interval fires limit subsequent burning and emissions (Higuera et al. 2009, Parks et al. 2015, Bernier et al. 2016). Because of these effects, a "let it burn" approach may result in higher deciduous cover (Rogers et al. 2013), decreased flammability (Foster et al. 2022), and increased carbon storage (Mack et al. 2021). However, some studies and fire management reports (Walker et al. 2019, Fall Fire Review personal communication 2019, Dieleman et al. 2020) indicate that fuel feedbacks are limited during late season severe fire seasons, and may not be as effective under intensified climate change. Furthermore, any increase in carbon storage from conifer to deciduous forests, even neglecting the associated degradation of permafrost, will take many decades, whereas global climate goals emphasize the importance of lowering emissions over the next few decades (Masson-Delmotte et al. 2018). Hence, a “let it burn” approach is unlikely to result in net benefits by mid- or even late 21st century.

2.4.3 Implications for Management Policy

Our model framework does not address the implementation of increased management spending. It is widely documented that once fires become large, especially when combined with intense fire weather, they are difficult if not impossible to contain and control (Podur and Martell 2007, de Groot et al. 2013). We therefore suggest a focus on expanding initial attack efforts. The majority of fires in boreal North America targeted for suppression are contained after initial attack (80 - 99%) (Cumming 2005, Arienti et al. 2006, Podur and Wotton 2010), and the remaining untargeted or escaped fires are responsible for over 95% of the burned area (Kasischke et al. 2002, Stockset al. 2002). Analysis also shows that most initial attack failures are due to slow initial responses, as opposed to “containment” failures (Arienti et al. 2006). Moreover, the majority of ignitions in Alaska occur in June, whereas the majority of burned area occurs in July (Veraverbeke et al. 2017); implying that most burned area occurs as a result of fires that remain small for several days or even weeks before spreading rapidly (Sedano and Randerson 2014). These facts suggest that increased resources for suppressing recently ignited fires would be an effective way to combat intensifying fire regimes.

Current Alaskan fire management is largely aimed at protecting human life and assets. Thus, fires in populated areas almost always stay very small, while there are large areas in the state where suppression is not the first response or primary goal. With additional resources, it could be possible to expand the areas where suppression is a management goal. One possibility is to increase the area covered under Full protection using a simple buffer around the existing areas as in Schultz et al. (2019); another possibility is to expand protection to include areas that are particularly carbon dense in order to more efficiently target emissions reductions. The best approach would likely be informed by the management community, which has intimate knowledge of conditions and constraints on the ground. However, any discussion of which areas to target for expanded protection will remain purely theoretical until the resources available for fire management in Alaska are increased through state and/or federal budgetary appropriations. More broadly, our results show that the social costs of emissions are several orders of magnitude larger than management costs. This general result is relevant for fire management policy both inside and outside Alaska. While it is becoming more common for states to quantify wildfire carbon emissions (e.g. CARB 2020), assigning a price to these emissions is only beginning to be explored in the academic literature, and is not standard in policy. Our results illustrate how substantial this oversight could be.

2.5 Conclusion

Overall, we find that the social cost of fire emissions greatly outweighs the cost of management when burned area exceeds historical levels, such that a large increase in management resources is justified. While our study focuses on Alaska, this conclusion likely translates to many boreal regions in Canada that contain similar forest characteristics, fire dynamics, carbon stocks, and fire management frameworks. Our results are less applicable to systems with different fire and management characteristics, particularly ecosystems adapted to high-frequency and low-severity fire regimes where continued suppression results in high fuel loads and flammability, for example in the western United States. However, we urge researchers and policymakers across regions to consider the social cost of carbon emissions when creating and evaluating fire management policy. When the alternative is skyrocketing carbon emissions with a social cost of billions of dollars per year, increased fire management may be a prudent and essential investment in the years to come.

Data and material availability: Historical fire data are available from the Alaska Interagency Coordination Center (https://fire.ak.blm.gov/aicc.php). LANDFIRE vegetation data are available from the US Geological Service (https://landfire.gov/). Code and other input data are available at DOI: 10.5281/zenodo.6395301

Chapter 3: The Social Cost of Wildfire Carbon Emissions[6]

Molly Elder

Abstract

During high-severity forest fires, organic matter burns and releases CO_2 into the atmosphere, which makes a significant contribution to climate change. However, ecosystems often continue to be net emitters of CO_2 for several decades after high-severity fires, and then absorb carbon for decades to centuries as forests regrow. These long-term carbon dynamics are often excluded from analyses of wildfire costs. This paper quantifies the social cost of the entire wildfire emissions profile, including post-combustion emissions and regrowth, by incorporating a representative fire emissions vector into the exogenous emissions vector in the DICE model, and then comparing welfare and consumption outcomes. The analysis finds that the social costs of fire emissions are approximately $27 million per 1000 hectares (ha) burned (in 2020 dollars), or an annual cost of $39 billion for all burned area across the United States. Furthermore, estimates of social costs are similar whether costs are quantified by comparing welfare or consumption, or by simply multiplying emissions in each period by the Social Cost of Carbon and discounting, provided the same discount rate is used.

[6] In addition to those thanked in the general acknowledgements, I received computing resources and support from Delilah Maloney and the Tufts Computing Cluster, and de-bugging help from Micah Nacht.

3.1 Introduction

When wildfires burn, they impose steep economic costs by pumping CO_2 and other greenhouse gasses into the atmosphere, contributing to global climate change. For example, in 2015, Alaska experienced a record fire year, with 1.6 million hectares (ha) burned in the interior and an estimated 0.055 Gigatons (Gt) C released (equivalent to approximately 0.2 Gt CO_2; Veraverbeke et al. 2017) and in 2020, wildfires in Californian forests released over 0.09 Gt CO_2 (CARB 2020), equivalent to approximately one quarter of the state's average annual pre-COVID anthropogenic emissions (CARB 2021). On average from 2002-2006, US fire emissions were equivalent to 4-6% of US anthropogenic emissions (Wiedinmyer and Neff 2007). Burned area has already increased across boreal, temperate, and tropical regions due to the influence of climate change (IPCC 2022). Fire regimes are expected to continue to evolve in the face of climate change, and fire regimes at high latitudes (Young et al. 2017) are expected to continue to intensify if global temperatures continue to rise. Due to their significant and increasing emissions, wildfires pose a serious threat to the global climate, so it is imperative that fire research explore the climate implications of current management policy.

There is limited research on the social costs of fires that includes costs from emissions, and research that does take the emissions costs of fires into account tends to consider only emissions during combustion (eg. Ning and Sun 2017, Thomas et al. 2017, Sánchez et al. 2021, Elder et al. 2022). However, the emissions impacts of a high severity (stand replacing) fire do not end once the fire is extinguished. After combustion, decomposing vegetation and dried-out soil continue to emit CO_2 into the atmosphere, and these continuing positive emissions are only overtaken by negative emissions (leading to net carbon uptake) several decades later, when vegetation regrowth becomes more substantial (Law et al. 2003, Amiro et al. 2010, Goulden et al. 2011). These post-combustion carbon dynamics have received negligible attention in the fire management literature. To fully understand the climate cost of wildfires, we must consider the entire cycle of fire and forest regrowth, rather than just the emissions during combustion.

In this paper, I quantify the social costs of wildfire emissions, including positive emissions both during and after combustion, and negative emissions which occur during regrowth. I compare baseline economic results to results with the additional emissions from fire added into the existing exogenous projection of land emissions. I find that the total social cost of emissions from combustion, post-combustion, and regrowth from a 1000 hectare (ha) high-severity fire is approximately $27 million (in 2020 dollars), and the emissions cost from total annual burned area across the entire United States, accounting for differences in fire severity, is $39 billion. The social cost of combustion alone is very similar to the social cost of the full emissions curve for the representative emissions curve in this analysis, because the additional costs imposed by post-combustion emissions are offset by the benefits from subsequent regrowth, but since the benefits of regrowth occur further into the future, they are discounted and the cumulative social cost remains positive. However, the social cost of all positive combustion and post-combustion emissions, which occur over approximately the next 30 years, are nearly 20% higher than the cost of combustion alone. This time frame is highly relevant since the Intergovernmental Panel on Climate Change (IPCC) has determined that achieving net zero emissions in the 2050s is necessary for the world to remain under 1.5 degrees of average warming (IPCC 2018).

This paper is built around the Dynamic Integrated Climate and Economy (DICE) model, which is widely used to combine economic and climate outcomes to estimate the damages from climate change. Originally developed by William Nordhaus in the early 1990s, DICE is now a widely used model for calculating the Social Cost of Carbon (SCC), since its open framework allows for a range of modifications (i.e., Ackerman et al. 2010, Lemoine and Rudik 2017). DICE is a welfare-maximizing model that combines exogenous projections of population and land emissions with endogenous per capita consumption and utility, GDP, industrial emissions, and climate damages (Nordhaus 1993, Nordhaus 2017, Nordhaus 2020).

I extend the existing DICE model by adding a vector of positive and negative emissions, representing carbon dynamics from combustion to forest maturity of a 1000 ha fire in Eastern Oregon, to the existing vector of exogenous land emissions in DICE. The purpose of the land emissions vector in DICE is to include separately the emissions from deforestation and non-CO_2 GHGs, which are assumed to be exogenous and declining at a constant rate[7] (Nordhaus and Sztorc 2013), while the majority of annual emissions come from human activity and are determined endogenously. Adding the additional fire emissions alters the exogenous emissions path slightly in each period until forest regrowth is complete, at which point the fire event becomes carbon neutral. Altering the exogenous emissions causes small shifts in the endogenous economic outcomes in subsequent periods. By altering the length of the fire vector, I can compare different scenarios including the impact of the entire vector of positive and negative emissions, the impact of combustion emissions alone, or the impact of only positive emissions.

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[7] While there has been recent work (e.g., Azar and Johansson 2021) to bring the carbon cycle module in DICE up to date, I have not found any papers that attempt to update the land emissions vector. According to Nordhaus and Sztorc (2013), the level and trend of exogenous land emissions is based on results from the Fifth IPCC Assessment. I think there is, at minimum, cause to update this vector based on the Sixth Assessment. However, we know that deforestation rates are affected by economic conditions (e.g., Alix-Garcia et al. 2013, Antonarakis et al. 2022, although the sign of the estimated effect is not the same across the two papers), that fire regimes globally are being altered by climate change (IPCC 2022), and that some increases in burned area are driven by intentional burning (for example, in Brazil; Pivello et al. 2021). Future research should allow both deforestation and burned area to be determined endogenously, in response to climate and economic outcomes.

I then compare the total welfare with and without this augmented emissions curve, and convert the welfare difference to a dollar amount to quantify the social cost of the fire (SCF). This difference-in-welfare approach is the one taken in DICE to estimate the SCC, and accounts for changes in both consumption and the marginal utility of additional consumption. Using this approach, I find that both the total SCF and the social cost of combustion emissions are approximately $27 million per 1000 ha (2022 dollars), but that the cost is up to 20% higher if only positive emissions during and after combustion are included. I also include two alternative methods for calculating the SCF, to test the robustness of the results to changes in methodology. I find that both alternative methods yield very similar results, provided a discount rate is applied that is consistent with the discount rate applied in the primary specification. However, I also find that the results are sensitive to the choice of discount rate, and that changing the discount rate does not have the same effect across the two alternative methods. In addition, I highlight a concern about imposing fixed discount rates on DICE results, which are built assuming a flexible growth framework.

The rest of the paper is organized as follows: Section 3.2 describes the fire emissions vector, DICE model framework, and model extension in detail. Section 3.3 presents the results from this framework. Section 3.4 presents the two alternative approaches and their results. Section 3.5 discusses the additional costs imposed by fires and the implications of climate-driven fire regime changes for the results. Section 3.6 concludes.

3.2 Model Framework

This paper integrates existing models from the ecological and Social Cost of Carbon literatures to estimate the social cost of combustion and post-combustion positive and negative emissions.

From the ecology literature, I selected a representative vector of the emissions from fire during and after combustion, scaled to reflect a 1000 ha fire. I include this vector of positive and negative emissions in a modified version of the DICE model[8]. The social cost of the fire is then the total change in welfare with and without the addition of the fire vector, converted into dollars, as described below.

3.2.1 Evolution of Fire Emissions During and After Combustion

To evaluate the social cost of fire emissions, the first required input is a vector of emissions during and after combustion. While the exact values vary by ecosystem and fire severity, the universal pattern is a large pulse of emissions during combustion, followed by one to several decades of positive emissions and then a long tail of negative emissions (carbon uptake).

In the ecological literature, the Net Ecosystem Production (NEP) of an ecosystem is a measure of its net carbon uptake or emissions in a particular state (Chapin et al. 2002), and NEP curves give the net emissions or uptake of an ecosystem over time. There are numerous studies which measure NEP for ecosystems at various lengths of time after fire events across both boreal (Bond-Lamberty et al. 2004, Amiro et al. 2010, Goulden et al. 2011) and temperate forests (Law et al. 2003, Campbell et al. 2004, Pregitzer and Euskirchen 2004, Magnani et al. 2007), providing a picture of how NEP evolves over time in different ecosystems. While there is variation in the magnitude of NEP and the estimated time for NEP to stabilize after a disturbance, there is a clear pattern in the shape of the NEP curve (see Figure 3.1 for the NEP curve used in this paper). In the first several years to decades (depending on the ecosystem) after a forest fire, the disturbed ecosystem continues to be a net source of carbon into the atmosphere

[8] I use the 2016 DICE model (specifics described in Nordhaus 2017), translated into Matlab by Derek Lemoine (https://github.com/dlemoine1/DICE-2016R-Matlab)

as emissions from the decomposition of newly-dead vegetation exceed uptake from regrowing vegetation (Chapin et al. 2002, North and Hurteau 2011). In the curve used in this paper, post-combustion emissions are equivalent to approximately 80% of combustion emissions. Then, over the next century or several centuries, the ecosystem exhibits a net uptake of carbon as vegetation regrows and soil organic matter reaccumulates, until eventually annual NEP stabilizes at zero. Of the papers that estimate full NEP curves, the curves for high-severity temperate ecosystems stabilize after 200-400 years with peak negative emissions of 200-500 $gCm^{-2}yr^{-1}$ (Law et al. 2003, Campbell et al. 2004, Pregitzer and Euskirchen 2004), while boreal ecosystems appear to stabilize after around 150 years with peak negative emissions of 100-300 $gCm^{-2}yr^{-1}$ (Bond-Lamberty et al. 2004, Goulden et al. 2011).

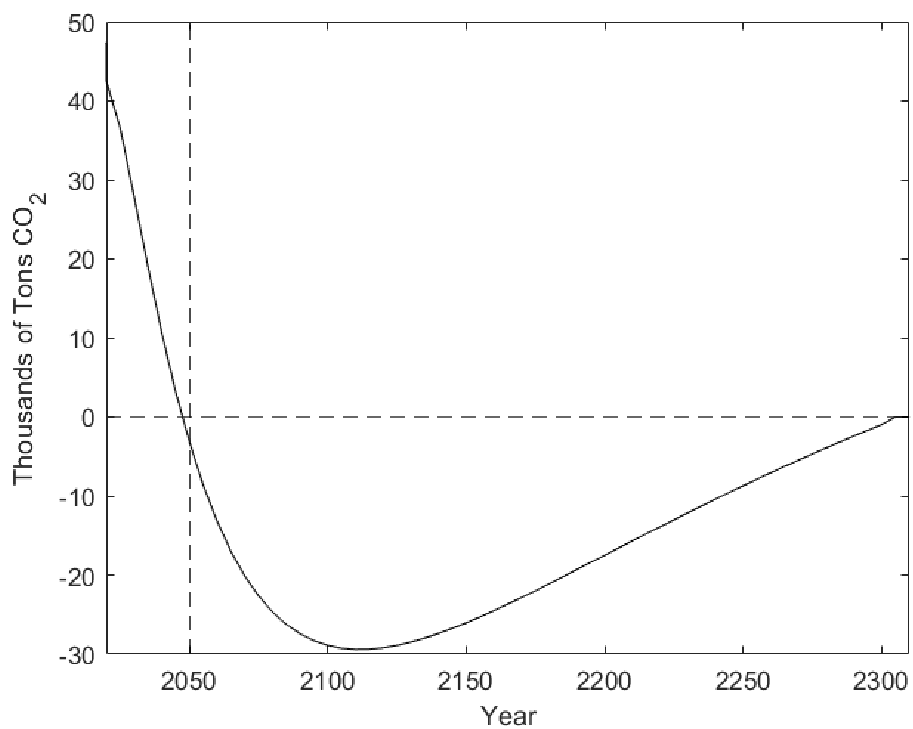

Figure 3.1: Projected CO_2 emissions from 1,000 ha of burned pine forest. Combustion emissions are not included in the figure, but are equal to approximately 738,500 metric tons.

For this study, I used the NEP curve defined in Law et al. (2003), which is estimated for high-severity fires in Ponderosa Pine stands in central Oregon[9] and represents a central estimate of both time-to-stabilization and peak negative emissions level from the temperate NEP literature cited above. I assumed that the sum of emissions during combustion and the integral of the post-combustion NEP curve is zero. This assumption, originally advanced by Odum in 1969, means that so long as the average time between stand-replacing fires is the same as the time for the forest to fully regrow, in this case 291 years, the ecosystem will be carbon-neutral in the long run. It will be instructive to shorten the tail of the NEP curve later in the analysis, since the influence of climate change means that the average time between fires is decreasing (eg. Mouillot et al. 2002, Flannigan et al. 2009, Guyette et al. 2014, Riley and Loehman 2016).

It is important to highlight here that the NEP curve presented in Law et al. is estimated for high-severity fires. In a high severity, stand-replacing fire, a large percentage (70-100%) of trees are killed, and the organic layer of the soil is burned through to expose the mineral layer below (Ghimire et al. 2012). In contrast, a low severity fire which burns through underbrush will not result in high overstory mortality (tree mortality at or below 25%), while medium severity forest fires result in tree mortality of typically 40-65%. NEP curves for lower-severity fires are characterized by smaller annual emissions (positive or negative), and by a shorter initial period of post-combustion emissions. In Ghimire et al. (2012), which synthesizes results from a variety of Western forests, negative emissions in moderate-severity fires peak at about 60% and 15-20 years sooner than in high-severity fires, while negative emissions in low-severity fires peak at

[9] This particular NEP curve is qualitatively a fairly central estimate of both the total length of the curve and the time from combustion until the switch from positive to negative emissions for high-severity, temperate forest fires. Furthermore, Law et al. helpfully provide the estimated equation in their paper.

35% and 30-40 years earlier. Since the paper does not provide parameter estimates for the different equations, I am not able to perfectly incorporate these differences, but I will scale the Law et al. emissions curve by these factors (0.6 and 0.35) to approximate costs for low- and medium-severity fires. On average, in the period 2010-2020, 12% of burned area in the contiguous United States was the result of a high-severity fire, and 23% was the result of a medium-severity fire (MBTS 2022). With this broad picture of fire severity across the United States, I will be able to apply the cost estimates derived from the Law et al. emissions curve to estimate costs across the country.

Law et al. estimate NEP as the grams of carbon absorbed per square meter per year, with negative values representing net emissions. For purposes of incorporating wildfire burns into DICE, I scale this curve up to reflect a burned area of 1000 hectares. In Oregon, the average burned area in the period 2010-2019 was 225,629 ha per year (nearly 1% of Oregon's total land area; NWCC 2021), and approximately 10% of Oregon's land area (approximately 2.5 million ha) is covered by Ponderosa Pine forest (Graham and Jain 2005). An additional 1000 ha of burned Ponderosa Pine forest thus is ecologically plausible based on land cover and represents a marginal increase relative to the total statewide burned area, while being large enough that the economic impact will not be lost to rounding.

3.2.2 Climate-Economic Framework

The damage imposed on society by the release of carbon or CO_2 into the atmosphere is typically calculated in several steps, using an Integrated Assessment Model (IAM) to combine climate and economic models. In general terms, a small pulse of CO_2 is added to the baseline emissions trajectory. Some of this pulse is sequestered by the land or ocean over time, while the remainder increases the atmospheric concentration of CO_2. This results in a small increase in global average temperatures, which in turn affects economic outcomes. The difference in economic outcomes with and without the additional emissions pulse is then compared in each period after the initial pulse, and the per-period difference is discounted and summed to calculate the total damages (National Academies of Sciences 2017). When the simulated pulse is a single ton of CO_2, the summed damages are called the Social Cost of Carbon (SCC). In general, SCC trends up over time because the increasing baseline concentration of CO_2 in the atmosphere increases the marginal damages of an additional pulse. This is both because positive economic growth implies higher per capita GDP and damages are calculated as a fraction of GDP, and because projected large future damages become closer in time.

To project how economic outcomes are impacted by emission changes, I use the Dynamic Integrated Model of Climate and the Environment (DICE) as a base. DICE was originally developed by William Nordhaus in the early 1990s (Nordhaus 1993, Nordhaus 2017, Nordhaus 2020), and is widely regarded as one of the premier IAMs for calculating SCC (Interagency Working Group 2010, National Academies of Sciences 2017). The DICE model[10] uses a Ramsey-Cass-Koopmans framework for optimal economic growth, which means that the social planner chooses policies to maximize total welfare (Equation 3.1), defined as the discounted sum of per-period utility U (Nordhaus and Sztorc 2013). In the baseline model, where climate mitigation is held at a low level, welfare is optimized by adjusting the savings rate in each period to alter current and future levels of consumption and investment. The utility function defines the benefits to consumption, and is a function of per capita consumption, $c(t)$ scaled by population, $L(t)$. Utility is discounted at rate ρ, the pure rate of time preference.

[10] This section provides a highly condensed overview of the DICE model's economic framework. For a much more thorough treatment, see Nordhaus and Sztorc 2013.

(3.1) $$W = \sum_{t=1}^{T} \frac{U[c(t), L(t)]}{(1+\rho)^{t-1}}$$

The utility function takes a familiar form, assuming constant elasticity of consumption (Equation 3.2), where η is the elasticity of marginal utility of consumption and measures the societal aversion to inequality.

(3.2) $$U[c(t), L(t)] = L(t) \cdot \frac{c(t)^{1-\eta}}{1-\eta}$$

Values of η greater than 1 indicate declining marginal utility of consumption, so increased consumption going to the rich produces less utility, while $\eta = 1$ indicates that society is neutral to inequality[11] and values all consumption gains equally. In this formulation, marginal utility declines toward zero as per capita consumption increases, but is always positive.

In DICE, economic damages, the wedge between economic outcomes with and without climate influence, are a quadratic function of temperature change, so larger changes in temperature result in increasingly severe economic impacts. Temperature change is dictated by the atmospheric concentration of CO_2, which is defined using earth system model equations that determine how emitted CO_2 is absorbed by the land, ocean, and atmosphere. The SCC (Equation 3.3) is given by the partial derivative of welfare with respect to emissions, converted from utils to dollars using the partial of welfare with respect to consumption.

(3.3) $$SCC_t = \frac{dW}{dE_t} \cdot \left(\frac{\partial W}{\partial C_t}\right)^{-1}$$

[11] When $\eta = 1$, per-capita utility is calculated as simply the natural log of per-capita consumption. Utility still becomes asymptotically large/small as η approaches 1, but this avoids division by zero.

SCC is thus the sum of the changes in utility in each period starting from an initial pulse of emissions, discounted at the utility discount rate, and converted to a consumption equivalent. Note that the calculated SCC values depend on the selected parameter values of both the pure rate of time preference ρ and the elasticity of marginal utility of consumption η.

To calculate the social cost of fire emissions (SCF), I add a vector of positive and negative emissions pulses to the existing vector of exogenous land-use emissions in DICE. I then calculate the SCF as the difference in welfare with and without the fire emissions vector included. I call this the DICE approach, since in the original model a difference in welfare is used to calculate the SCC.

In the DICE approach, the damages from emissions are calculated as the difference in welfare when exogenous emissions include the fire vector (W^F) and welfare in the baseline model run (W^0), converted into dollars. I add the fire vector to the existing vector of exogenous emissions, and then calculate the subsequent welfare difference, allowing consumption, industrial emissions, and other endogenously determined variables to adjust based on the changed exogenous emissions vector. The DICE Approach is summarized in Equation 3.4.

(3.4)
$$SCF_D = (W^F - W^0) \cdot \left(\frac{\partial W}{\partial C_1}\right)^{-1}$$

The comparison of outcomes encompasses the full IAM projection period, because the effects of an exogenous emissions change reverberate through the economy for decades after the initial emissions pulses from fire have ceased.

To calculate both W^0 and W^F, I run a baseline version of DICE, which means that emissions reductions are being implemented in a business-as-usual model (i.e. no climate policy in place), although welfare is still maximized conditional on business-as-usual climate policy.

3.3 Results

Figure 3.2 shows that the social cost of the positive and negative CO_2 emissions from the hypothetical 1000 ha, high-severity fire event is approximately $27 million (2020 dollars). The social costs for medium- and low-severity fires respectively are $16.2 million and $9.5 million per 1000 ha, respectively. Over the period 2012-2021, the average annual burned area across the entire United States was 2.99 million ha (Congressional Research Service 2022). If we assume that the emissions curve used in this analysis is broadly representative of all fires in US forests[12] and multiply the cost per 1000 ha fire by the total average annual burned area broken down by severity, the annual social cost of all carbon emissions from fires in the United States is approximately $39 billion each year[13].

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[12] This is a very strong assumption. To incorporate slightly more regional heterogeneity, we can differentiate between burned area in the continental United States (represented by the temperate forest estimates from Law et al.) and burned area in Alaska. Alaskan forests are primarily boreal, a different ecosystem, and represent an average of 15% of national burned area (NPS 2022). If we perform the same analysis as above but use the NEP curve originally estimated in Goulden et al. (2011) from study sites in Manitoba, and parameterized in Phillips et al. (2022), the SCF for a 1000 ha burn in boreal forest is $3.15 million dollars, and the total US costs from fire are approximately $35 billion. The assumption is still fairly strong, since some fires occur in grasslands or other non-forest ecosystems. Introducing this increased heterogeneity will be the subject of future work.

[13] See Appendix 3 for calculation.

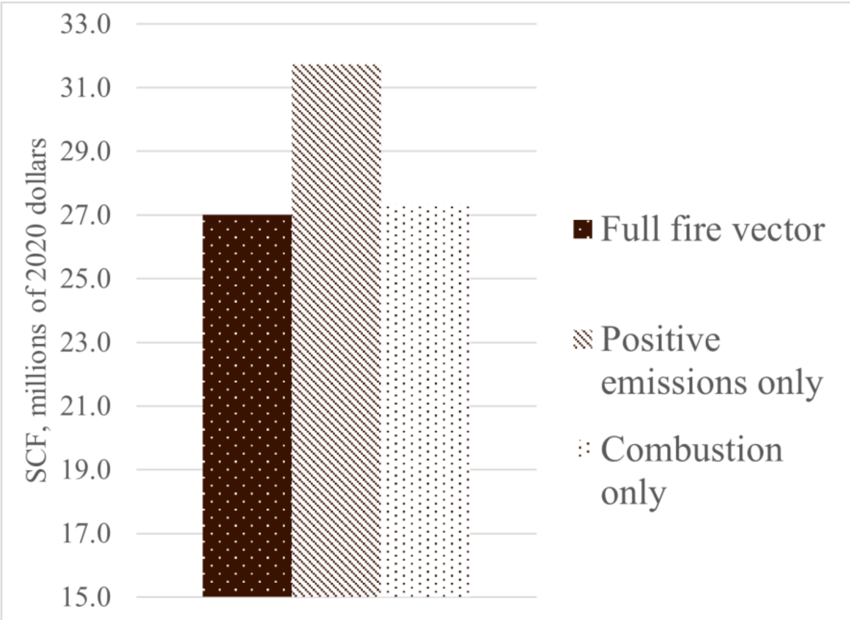

Figure 3.2: Social cost of fire emissions by fire vector length, in millions of 2020 dollars.

Using the emissions curve from Law et al., I find that the social cost of the full emissions curve is very similar to the social cost of combustion emissions alone, $27.3 million. However, the cost of all positive emissions, without benefits from regrowth, is 18.5% higher than the cost of the complete emissions vector. The intuition for this result is made clear in Figure 3.3a, which shows the evolution of consumption over the projection period. Consumption initially declines in all three cases, but the decline is larger when post-combustion positive emissions are included. For the combustion only case and the positive emissions only case, consumption declines continue to increase throughout the period, with a larger decrease in the case with additional positive post-combustion emissions. This explains why the estimated SCF is larger in the positive emissions case – more fire emissions result in larger damages. In contrast, while the drop in consumption is the same for the full emissions case and the positive emissions case for the first 31 years, the two diverge once regrowth begins to yield negative emissions. By year 100 the benefits from regrowth have counteracted initial consumption declines enough that the annual difference from baseline becomes smaller than the difference in the combustion-only case (Figure 3.3b). Eventually, when the fire emissions vector reaches its end and the sum of emissions reaches zero, consumption returns to its baseline level.

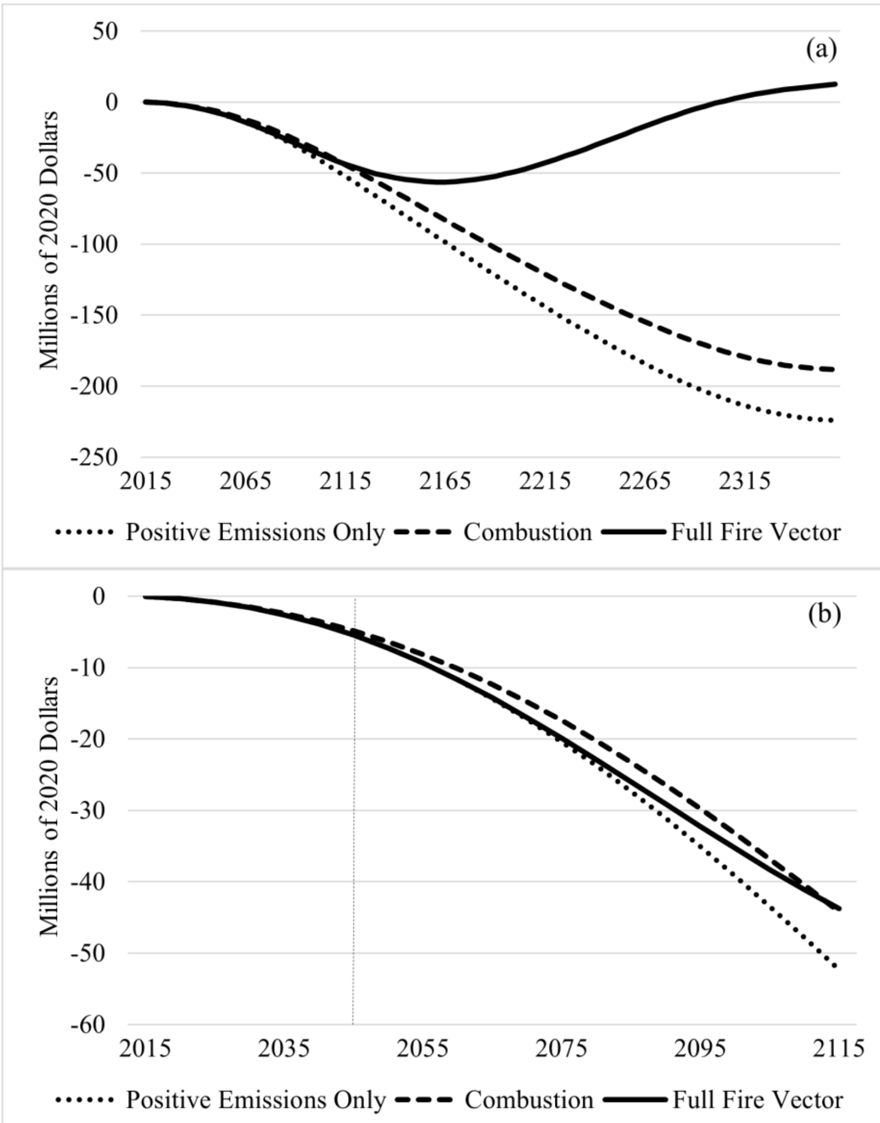

Figure 3.3: Annual change in consumption under three versions of fire emissions, relative to baseline, in millions of 2020 dollars. Notably, this figure shows consumption outcomes valued in the year in which they occur. The SCF_D is calculated using the difference in welfare (the present value of the summed difference in utility), so economic shifts that occur earlier in the projection period are given more weight in the welfare calculation. Panel a shows how consumption evolves over the entire projection period, while panel b zooms in on the first 100 years to provide a clearer image. The dotted blue line in panel b marks the point at which negative emissions begin.

Relying on a single NEP curve means that the analysis will be based on a post-combustion curve of a particular shape. However, while the Law et al. curve provides a middle-of-the-road estimate, the regrowth curve could be different lengths or shapes. For example, a flatter curve would have regrowth benefits concentrated at a greater temporal distance from combustion. In Figure 3.4, I concentrate all the regrowth benefits in a single year and estimate the SCF as that year is moved further out. As regrowth benefits are concentrated further in the future, the estimated SCF increases to approach its upper bound, the SCF for only positive emissions. While the SCFs from combustion and from the full fire vector are very similar in this analysis, Figure 3.4 shows that the SCF depends on the shape of the NEP curve. The NEP curve used in this analysis represents a central estimate of both the duration and height of NEP curves in temperate forests, so I conclude that the results should represent a central estimate of SCF as well.

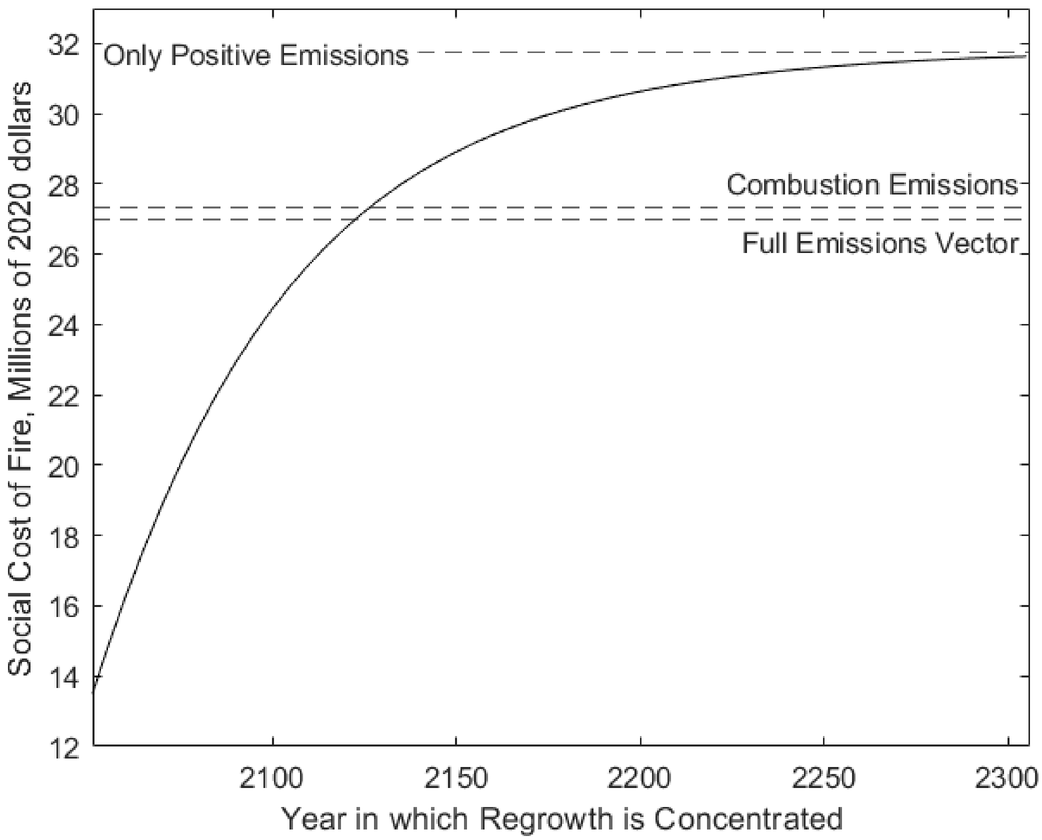

Figure 3.4: Social cost of fire emissions when regrowth benefits are concentrated in a single year, in millions of 2020 dollars. The dashed lines, from top to bottom, show the estimated SCF with all positive emissions, combustion emissions only, and the full fire vector.

3.4 Alternative Approaches

While the primary specification in this paper estimates costs using a comparison of welfare outcomes, there are two other plausible ways to estimate costs: a comparison of consumption, and a price-times-quantity approach. All three approaches yield similar results when the discount rate applied is consistent with the underlying modelling assumptions, but results vary significantly under other discount rates.

3.4.1 Methods

The first alternative approach I consider draws on the methodology of the United States Interagency Working Group on the Social Cost of Greenhouse Gasses (IWG) and computes social costs as the difference in per-capita consumption. The second alternative approach simply multiples the quantity of emissions in each period by the price per unit of emissions, which is the SCC.

For US policy, the IWG combines results from DICE and two other IAMs, PAGE and FUND, to produce recommended SCC values for official use (Interagency Working Group 2010, Interagency Working Group 2021). For the purposes of this analysis, the most important difference in approaches between DICE and the IWG is that while DICE compares welfare and then converts to consumption equivalent units, the IWG compares consumption directly in each period. As in the DICE Approach, the difference in consumption outcomes in each period with the additional fire vector (C_t^F), and baseline consumption C_t^0, is the result of both the direct changes in consumption due to a change in exogenous emissions and the changes in consumption due to endogenous variables adjusting to the new emissions scenario in previous periods. The IWG Approach can be summarized in Equation 3.5.

$$SCF_I = \sum_{t=1}^{T} \frac{C_t^F - C_t^0}{(1+r)^{t-1}} \tag{3.5}$$

Note that, because the IWG Approach considers consumption changes, the appropriate discount rate is the consumption rate r, not the utility rate ρ used above.

The most straightforward way to calculate the SCF is to multiply the fire emissions vector $\boldsymbol{E}^F$ by a vector of SCC values in each year **SCC**, and then discount back to the present and sum the results. I used SCC values calculated in the baseline DICE model[14], assuming business-as-usual mitigation levels without including the fire emissions curve. I refer to this as the Straightforward Approach, summarized in Equation 3.6, since it relies on the very basic economic intuition that the total cost of a group of items (in this case, emissions pulses) is the number of items times their unit price, while not taking into account that the value of later emissions pulses may be altered by the lingering effects of earlier pulses.

$$SCF_S = \sum_{t=1}^{T} \frac{E_t^F \cdot SCC_t}{(1+r)^{t-1}} \tag{3.6}$$

Again, the appropriate discount rate for the Straightforward Approach is the consumption discount rate, *r*, since in multiplying emissions we are combining dollar-valued impacts from each period. Note that in the Straightforward Approach, the summation starts at combustion and ends in the last year of the emissions vector, rather than at the end of the model's projection period.

3.4.2 Choice of Discount Rate

The fire emissions vector I used for this paper extends for nearly 300 years, with positive near-term emissions and negative emissions (uptake) further out. Because of this long time horizon, it is important to consider the time value of money, the discount rate. While discounting utility involves only the choice of a single parameter, there are two main schools of thought for choosing consumption discount rates: the prescriptive approach, implicitly present in DICE

[14] The DICE SCC values are reasonably similar to the IWG (2021) values, which makes sense since DICE is one of the three input models in the IWG calculation. The DICE SCC is $44 in 2020, increasing to $107 in 2050. The IWG SCC is $51 in 2020, increasing to $85 in 2050. I use the DICE values, because IWG only estimates SCC through 2050, but the length of the fire emissions curve requires SCC values through at least 2310.

(Nordhaus and Boyer 2000, Interagency Working Group 2010), and the descriptive approach, favored by the IWG.

Since DICE is concerned with maximizing welfare, rather than consumption, intertemporal tradeoffs are assessed using the value of marginal consumption η and the pure rate of time preference ρ, the welfare discount rate. However, by applying the Ramsey equation (Equation 3.7), the choice of values for these parameters combined with the per-capita consumption growth rate g implies a consumption discount rate r which varies with the growth rate (National Academies of Sciences 2017).

$$r = \rho + \eta g \tag{3.7}$$

This is called the prescriptive approach to discounting. In DICE, the consumption interest rate is calculated only as an output, and is not employed in the welfare optimization. However, it accurately reflects the intertemporal tradeoffs built into the model's optimization, so it is useful to have it on hand when working with the model's dollar-denominated output.

Scenarios with long time horizons tend to be highly sensitive to the choice of discount rate, and in the prescriptive approach this means that results are sensitive to the choice of parameter values for ρ and η. Ethicists (and ethically-inclined economists) tend to prefer a pure rate of time preference of very nearly zero, because there does not seem to be any a priori reason that future generations should receive less consideration than current generations, other than the small possibility that humans will go extinct and there will be only a limited number of future generations (Stern et al. 2006).

There is more disagreement about η. The parameter measures society's aversion to inequality, with higher values corresponding to higher aversion. However, η must carry our aversion to both intragenerational and intergenerational inequality (Dasgupta 2008), and those considerations can pull in different directions if we think that in future generations wealth will be both higher on average and more evenly distributed than in the present[15]. In DICE, the default parameter values are $\rho = 0.015$ and $\eta = 1.45$, and because consumption growth is endogenously determined by other outcomes including climate damages, the implied consumption discount rate varies. In the baseline DICE run, consumption grows at an average of 1.1% annually over the first 350 years, implying an average consumption discount rate of 3%. However, the consumption growth rate declines over time, with higher growth and discount rates at the beginning of the projection period, so an average discount rate of 3% across the whole period should not be taken to mean that a 3% discount rate is appropriate to apply in all time periods.

The IWG prefers the descriptive approach to discounting, in which the interest rate is determined by observing historical rates (National Academies of Sciences 2017). For regulations that will primarily affect consumption decisions, or have effects measured in consumption-equivalent units, the US federal government recommends using a discount rate of 3%. This represents the social rate of time preference, proxied using the average return to long-term government debt (OMB 2003). In addition to this central rate, the IWG applies two other fixed annual rates, 5% and 2.5%. The higher rate reflects the possibility that changes in consumption are positively correlated with climate damages (Interagency Working Group 2010). If damages are larger when consumption is high, the payouts from investment are higher in boom times and reduced in bust times, making them less valuable than receiving consistent payments and necessitating a higher rate of interest to compensate. The lower rate of 2.5% has two rationales. The first is the converse of the rationale for the high interest rate, namely that climate damages could be higher when consumption is lower, so the benefits from mitigation investment are

[15] For a good summary of the arguments around discounting, see Posner and Weisbach (2010).

especially large and desirable when the economy is doing poorly. This countercyclicality would make mitigation investment more attractive, and would mean that a lower interest rate is needed to sway investors (National Academies of Sciences 2017). The second rationale is that future discount rates are uncertain, and the certainty-equivalent rate is lower than the mean rate of 3% (Newell and Pizer 2003, Interagency Working Group 2010).

Both the Straightforward and IWG Approaches measure differences in consumption or consumption-equivalent units, so r is the appropriate discount rate to apply. I compare results using four rates: the IWG's preferred fixed rates of 2.5%, 3%, and 5%, and the variable rate implied by the combination of ρ, η, and the per-period modeled growth rate of consumption.

3.4.3 Results

I will demonstrate in this section that the choice of discount rate for the IWG and Straightforward Approaches is extremely consequential for the ultimate size of the SCF. For both approaches, using the implied Ramsey discount rate yields a nearly identical result to the DICE approach. However, when fixed discount rates are imposed, the results differ substantially from DICE. Furthermore, changing the value of the discount rate has an opposite effect on the size of the calculated SCF using the IWG and Straightforward Approaches. The results are summarized in Figure 3.5.

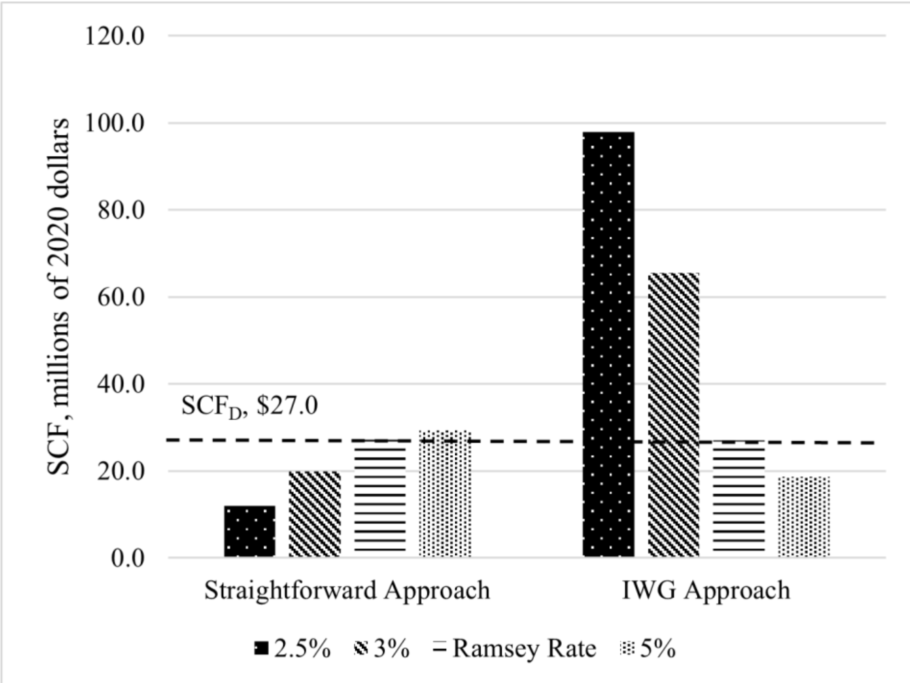

Figure 3.5: Social cost of fire emissions for the full fire emissions curve under different discount rates. SCF_D is displayed as the black dashed line for comparison.

To understand these results, it is useful to remember what outcome each approach measures. The IWG approach uses the present value of the difference in consumption (Figure 3.6a). Consumption is lower than baseline for nearly the entire projection period, so increasing the impact of future events by decreasing the discount rate leads to an increase in the total cost. In contrast, the Straightforward approach uses the present value of the quantity of positive or negative emissions (Figure 3.6b). Negative emissions confer benefits, so a lower discount rate that puts more emphasis on future events leads to a decline in the estimated SCF. The variable Ramsey rate starts around 4.5% and declines to 2% by 2375, with a certainty equivalent of 4.3%[16], putting it in between the high and medium fixed rates. Overall, we see that the disparity between the Straightforward approach and the IWG approach results is greater for low discount

[16] See Appendix 3 for calculation

rates, and that the impact of changing the discount rate has a different sign for the two alternative approaches. Typically, in the SCC literature, decreasing the discount rate increases the cost estimate by increasing the weight given to future damages, so it is novel that the Straightforward approach shows the opposite pattern.

In addition to yielding highly variable results, imposing a fixed discount rate on output from the DICE model creates inherent structural tension. The DICE model is designed to maximize social welfare over numerous generations by adjusting the per-period savings rate to determine flows of consumption and investment[17] (Nordhaus and Sztorc 2013). Total welfare is the sum of per-period utility, a function of population, per-capita consumption, and the marginal utility of consumption, discounted using the utility discount rate, the pure rate of time preference. These factors are captured in the Ramsey equation. In a very fundamental way, the DICE model's optimization is built on the Ramsey optimal growth structure. Thus, when we discount consumption changes for the IWG Approach using the Ramsey discount rate, calculated based on the DICE model output, we are using a discount rate consistent with the assumptions about intertemporal tradeoffs used in the model optimization. In contrast, discounting consumption changes using one of IWG's preferred fixed rates is inconsistent with the optimization assumptions, and yields results that diverge significantly from the DICE Approach result.

[17] In the climate-optimized version of the model, the emissions reductions rate is also a choice variable. I use the business-as-usual specification, where climate mitigation is fixed.

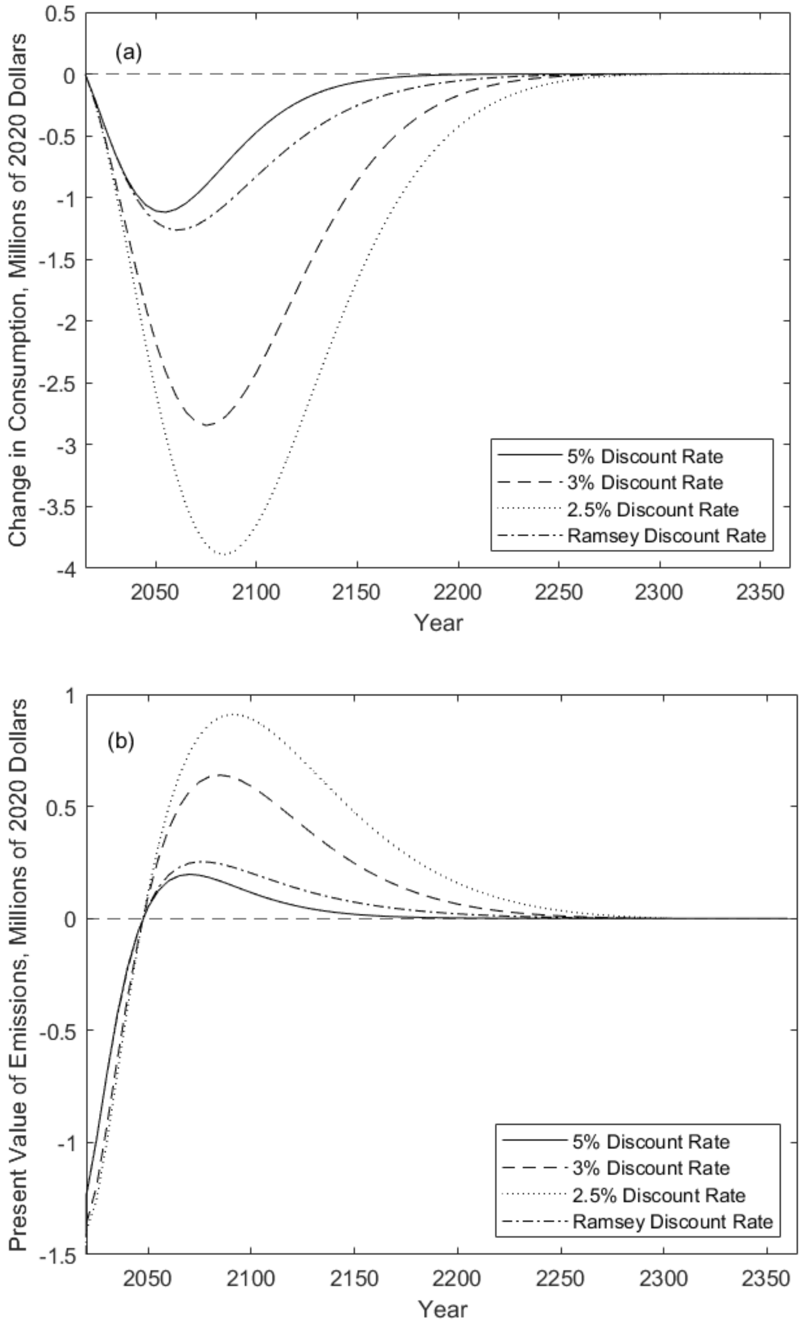

Figure 3.6: The present value of the annual difference in global consumption (a) after the fire emissions vector is added to exogenous emissions, relative to baseline consumption, in millions of 2020 dollars, is negative or approximately zero throughout the projection period. The present value of the annual positive or negative emissions (b) is negative when emissions are positive, indicating a cost, and positive during the period of regrowth, indicating a benefit. The first year is not included, because combustion generates a much larger pulse of emissions and would make subsequent variation difficult to see. The value of combustion emissions is approximately $27.3 million.

This issue is magnified because of the long time horizon in the current analysis, but it is not unique to this analysis. So long as the economic growth rate, marginal utility of consumption, and pure rate of time preference are inconsistent with the fixed discount rate, this internal inconsistency will be present. Moreover, the IWG SCC calculation is based on an average of model output from PAGE, FUND, and DICE, harmonized to have the same population and economic growth rates (Interagency Working Group 2021). In this paper, I highlight the importance of incorporating post-combustion emissions into valuations of fire damages, but researchers who attempt to monetize the carbon storage or emission impacts of policies by using the IWG's SCC values and discount rates will inadvertently introduce the inconsistency (eg. Mills et al. 2015, Haight et al. 2020, Sánchez et al. 2021). It is clear from comparing the IWG and Straightforward Approach results that the impact of applying inconsistent discount rates is difficult to predict, and may be large.

The National Academies recommend that the IWG adopt a Ramsey discounting framework rather than a fixed-rate framework, while choosing values of ρ and η such that the implied Ramsey rate would be consistent with the preferred fixed rates in the short run (National Academies of Sciences 2017). The Academies are motivated by a desire to incorporate sensitivity analysis around uncertain discount and growth rates, and not by the DICE/IWG rate inconsistency highlighted in this paper, but the proposed approach would resolve this inconsistency. Newell et al. (in press) estimate Ramsey parameters that are consistent with both observed near-term market interest rates of 1.5%, 2%, 3%, and 5%, and modeled long-run declining term structures[18]. I performed a sensitivity analysis of the SCF using the four sets of Ramsey parameters (Table 3.1).

Table 3.1: Ramsey parameters from Newell et al. with resulting SCF

	Initial Consumption Discount Rate			
	1.5%	2%	3%	5%
ρ	0.0%	0.1%	0.8%	2.4%
η	0.99	1.25	1.53	1.86
Social Cost of Fire, millions of 2020 dollars	254.73	128.92	39.83	11.38

As the discount rate decreases, the SCF increases, since the present value of far-out damages is larger. What may be more surprising is the magnitude of the effect. The estimated SCF for a 1000 ha fire ranges from \$255 million for an initial discount rate of 1.5% to \$11.4 million for an initial discount rate of 5%, and the central estimate at a 3% initial rate is more than \$10 million more than the central estimate using the DICE approach, at nearly \$40 million. The SCF using the IWG Approach with a 2.5% discount rate is less than half the SCF using the parameters for a 2% initial consumption discount rate. This can be attributed to the declining rate structure (Figure 3.7), in which the increase in the present value "penalty" declines over time. The results in Table 3.1 show that, while it is possible to use both a Ramsey growth model framework and a set of preferred initial consumption interest rate values, addressing the inconsistency problems laid out in this paper, doing so may increase the sensitivity to parameter

[18] There is a significant body of literature on the effect of uncertainty in the discount rate or underlying factors upon the certainty-equivalent discount rate. If discount rates are uncertain, as the time horizon increases the expected value of the discount factor, E(exp(-rt)), declines toward the lowest of the possible discount factors (see proof in Weitzman 1998). As t increases, larger values of r yield vanishingly small discount factors and only the discount factors associated with smaller values of r remain significant. Thus, as t increases, the per-period certainty-equivalent discount rate declines.

selection and make confidence intervals even wider. Hence, switching to this combined approach is not necessarily a panacea for all the discounting considerations IAMs have faced, although it does resolve the immediate concerns about inconsistency with the internal model framework.

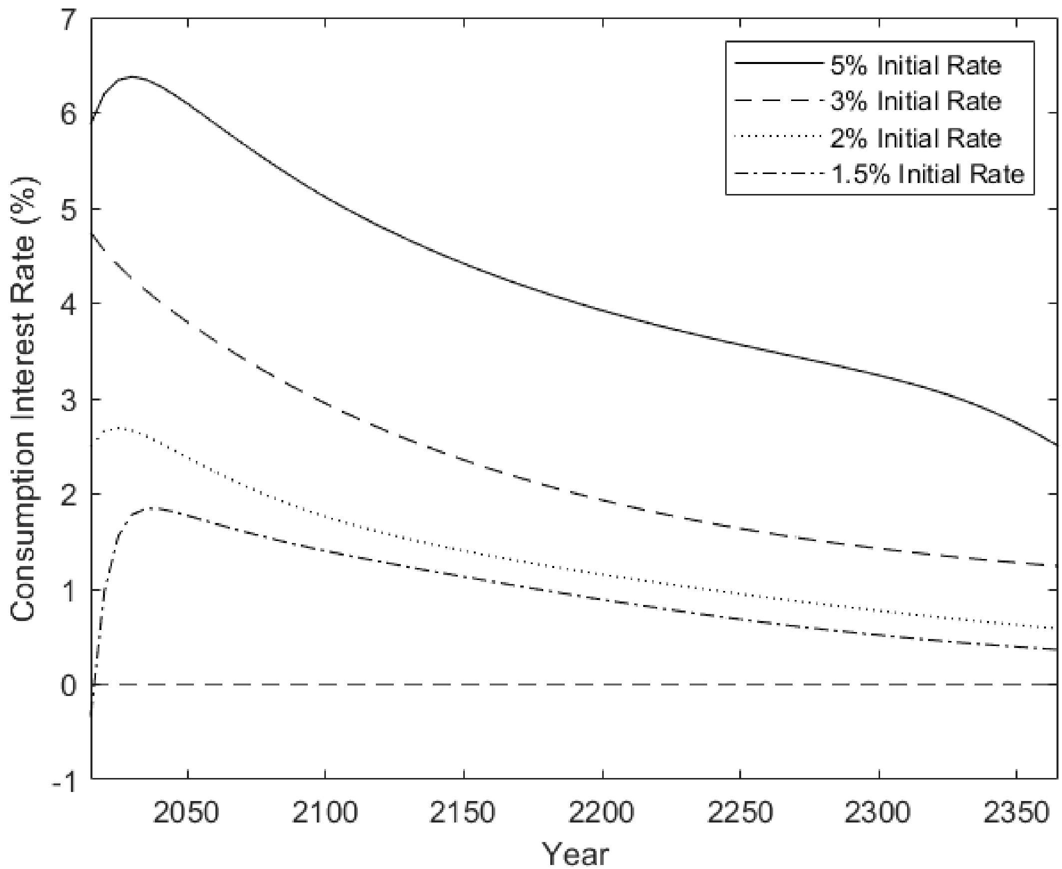

Figure 3.7: Consumption interest rates implied by the η and ρ values for different initial interest rates suggested by Newell, Pizer et al., and the projected growth rate.

3.5 **Discussion**

Research and policy analysis routinely fails to consider post-combustion emissions, or often even combustion emissions alone, when estimating the costs imposed by wildfires and the resources that should be allocated to address fires. Excluding the social costs of these emissions can result in a severe underestimate of the benefits of fire management. I found that the social cost of the full emissions curve is similar to the social cost of combustion emissions, at $27 million per 1000 ha fire, but 20% lower than the social cost of all positive emissions. However, there are several factors that suggest this is still an underestimate of the true costs of fires, including the non-CO_2 climate impacts from fires and the direct damages imposed, which were outside the scope of this paper. I also address the sensitivity of these results to climate-driven decreases in the average interval between fires, and how updated IAMs and damage functions may alter the results.

3.5.1 Additional Costs from Fires

While this analysis attempts to give a full account of the social costs of CO_2 emissions during and after a fire, it does not include many other ways in which fire events can impose social costs, through both climate and non-climate channels. Fire can impact climate on a local scale since many ecosystems have decreased surface albedo after burning (eg. Jin and Roy 2005, Dintwe et al. 2017), which means that more light is absorbed, and local temperatures increase. In areas with snow cover, albedo typically rises after fire, since snow-covered ground is exposed in the absence of vegetation and snow is more reflective than vegetation (Lyons et al. 2008). However, black carbon (soot) deposits on snow and ice can decrease albedo significantly (Flanner et al. 2007).

On a global scale, fires emit a number of greenhouse gasses in addition to CO_2 during combustion, including methane, nitrous oxide, ozone, and ozone precursors (Ward et al. 2012, Urbanski 2014), all of which affect global temperatures. These non-CO_2 gasses may significantly increase the radiative forcing (the energy kept in the atmosphere) from fire emissions (eg. Huang et al. 2016, Manojkumar and Srimuruganandam 2019). More radiative forcing results in higher temperatures, so an increase in radiative forcing implies higher social costs imposed by increasing temperatures. Furthermore, fire events can lead indirectly to an increase in methane emissions by degrading permafrost in Arctic and sub-Arctic regions (Jafarov et al. 2013, Holloway et al. 2020) and degrading peat in tropical (Akhtar et al. 2020) and boreal (Bourgeau-Chavez et al. 2020) regions. Because of these factors, wildfires may impose larger damages through the channel of climate change than estimated in this paper.

There are also numerous economic impacts imposed by fire directly, rather than through the channel of climate, which are included in neither DICE nor in this paper's analysis of the SCF. Among other impacts, wildfire smoke exposure can have significant health impacts across large areas (Cascio 2018, Burke et al. 2021, Burke et al. 2022), as well as direct economic impacts like infrastructure loss, depressed housing markets, and declines in revenue from recreation and tourism (Thomas et al. 2017). Thomas et al. estimate that the total economic burden of wildfires in the United States is between $76.7 and $375 billion[19] without including the social costs of emissions. Based on the national results above, the social costs of all non-climate fire impacts are only about double the often-excluded costs of emissions. Including emissions in fire cost estimates could significantly increase the estimated costs of fire. Put another way, including the social cost of emissions increases the estimated benefit of increasing fire management resources to reduce burned area.

3.5.2 Impact of Climate Change on Fire Regimes

Climate change is driving global increases in both forest area burned and emissions from forest fires, which could affect this analysis both by shortening the time between ignitions and by altering the amount of carbon released. There is already evidence that climate change is driving a decrease in the interval between fires across diverse forest ecosystems (eg. Mouillot et al. 2002,

[19] Originally 2016 dollars; converted to 2020 dollars for direct comparison

Flannigan et al. 2009, Guyette et al. 2014, Riley and Loehman 2016). We can think of this as cutting off the tail end of the fire emissions curve, reducing carbon uptake during regrowth. In theory, this should decrease the benefits of regrowth and increase the total social costs of fire emissions. In practice, as has already been shown, the early emissions and their economic effects drive much of the cost. Eliminating the last 41 years of uptake, so that the fire emissions vector is 250 years long instead of 291 years, makes little difference in any of the results presented in Table 3.1 (Table 3.2). Thus, the results presented here are reasonably robust to climate-driven changes in fire frequency, holding carbon emissions in each year of the post-combustion emissions curve constant.

Table 3.2: Total cost of first 250 years of fire emissions vector, millions of 2020 dollars

	Consumption Discount Rate			
	Ramsey Rate	2.50%	3%	5%
Straightforward Approach	27.3	12.9	20.2	29.3
IWG Approach	27.1	98.5	65.7	18.7
DICE Approach	27.1			

Another way to think about this robustness questions is to hold fixed the original 291 year study period, rather than holding fixed the fire and shortening the study period. If we assume that after 250 years another fire occurs with exactly the same emissions curve as the first fire, the estimated SCF_D imposed by land carbon emissions and uptake over the 291 year study period is $27.267, an increase of approximately 1% over the main SCF_D estimate. Temporally distant changes to fire regimes do not have a large effect on the SCF estimate, regardless of whether the original single fire or the original time period is considered.

Increasingly frequent fires may decrease available above and below-ground carbon over time (Foster et al. 2020), so future fires may have a lower climate impact and a lower social cost. However, there is evidence that soil carbon may return to baseline levels within 60 years post fire, and possibly earlier (Li et al. 2021). In peat forests like boreal Alaska and Canada, where the vast majority of fire emissions come from soil carbon, repeated fires may release soil carbon that was previously too wet or frozen to burn (Brown and Johnstone 2011, Turetsky et al. 2015). Thus, while frequent fires may reduce average per-acre emissions over time in some ecosystems, in others the emissions per acre could remain at currently measured levels or even increase if fires intensify and make more previously sequestered carbon accessible. How these ecosystem-specific effects would add up over time to alter national or global annual fire emission cost estimates is a promising area for future research.

Notably, climate-fire feedbacks have not been incorporated into DICE, or any other IAM of which the author is aware. There have been some recent attempts to incorporate positive climate feedbacks involving permafrost methane into DICE and PAGE (the second IAM used by IWG; the third is FUND), but neither model incorporates all climate feedbacks comprehensively (Interagency Working Group 2021). Since climate-fire feedback are not currently included in DICE's estimates of climate damages, calculating the SCF by either applying the SCC or incorporating a fire emissions vector into DICE directly will not be "double counting" the climate damages from fires.

3.5.3 Impact of Higher SCC Estimates

Recent work suggests that DICE may be underestimating the social costs of climate change in at least two important ways. For one, DICE assumes a quadratic damage function, so higher temperatures lead to increasingly high damages (Nordhaus and Sztorc 2013). However, the damage function parameters have not been adjusted to keep pace with recent estimates of the impacts of climate change, and likely underestimate the damages from temperature increases (e.g., Burke et al. 2015, Howard and Sterner 2017). In addition to a damage function calibration that lags behind the current literature, there are no tipping points built into DICE (Nordhaus and Sztorc 2013). Tipping points are levels of temperature increase after which damages increase significantly, for example because of permafrost thaw releasing large quantities of methane or the collapse of the Amazon rainforest ecosystem (National Academies of Sciences 2017). In the face of these small-probability catastrophic events, the expected social cost of temperature increase rises substantially (e.g., Weitzman 2011).

While some of the work cited above attempts to adjust components of the DICE model to better fit the current state of the literature, some recent scholarship moves beyond DICE. Rennert et al. (2022) have developed an entirely new IAM which establishes SCC levels over three times higher than DICE in the near term, while addressing many of the criticisms faced by DICE. Taken together, the consensus appears to be that the true social cost of emissions is significantly higher than that estimated in DICE.

All of this scholarship introduces a question – if the true social cost of emissions is much higher than DICE estimates, how should we view the results presented here? I expect there would be first-order and second-order effects. The first-order effect is that, if the true social costs of emissions are much higher, the social cost of *fire* emissions will also be higher. The second-order effect comes from changes in the convexity of the damage function, for example if tipping points are priced into the model. If new estimates of marginal damages increase at a much higher rate than in the DICE model, the benefits of regrowth (which occur in the future, when the stock of atmospheric carbon is expected to be higher) will be larger than the estimates presented in this paper, although with a non-zero discount rate its influence will still be muted. Combining these two effects, I expect that the SCF estimate would be increased in light of recent research, but not as much as estimates of present-day SCC. This supports the point in section 3.5.1, that the estimates presented in this paper are a lower bound of the full climate impacts of fire emissions.

Taken together, the results presented here suggest that the annual social cost of fire emissions is being underestimated by at minimum $39 billion, and possibly by significantly more. This means that the benefits of fire management, both suppression efforts which keep small fires from growing, and fuels treatment and other efforts prior to ignition that decrease the likelihood of fire starting or spreading, are being undervalued and underfunded. Fire management can be an effective tool to reduce burned area (Phillips et al. 2022), and the costs of inaction, which include billions of dollars in emissions costs, should be fully recognized.

3.6 Conclusion

Wildfires release large amounts of CO_2 into the atmosphere during combustion, but the costs imposed by these emissions are often not included in fire policy or research. This analysis found that the social cost of the emissions from a 1000 ha fire is approximately $27 million. The annual social costs of the emissions from all fires in the United States are $39 billion, equivalent to up to half of all other non-climate wildfire costs combined (Thomas et al. 2017) and significantly larger than the $2.4 billion spent annually by the federal government on fire suppression over the last ten years (NIFC 2022). Furthermore, since positive post-combustion emissions occur for 30 years after the initial fire, incorporating emission costs into policy is especially important, since the world needs to become carbon neutral by the 2050s in order to remain under a 1.5 degree temperature increase (IPCC 2018). The social cost of fire emissions from the present through 2050 is nearly 20% higher than the combustion costs alone.

In addition to providing a concrete estimate of social costs from emissions, this paper makes two methodological contributions. First, it has shown that using a simple price-times-quantity approach to calculating the social cost of an emissions event with a long time horizon yields a comparable result to more complex approaches, provided the appropriate discount rate is applied. I have shown that the social costs imposed by fire emissions are large, so excluding them from policymaking means that the benefits of fire management to limit burned area are being significantly undervalued. The finding that the straightforward approach can yield fairly accurate estimates of social costs should encourage researchers and policymakers to incorporate emissions costs in future work, since the methodology for doing so need not be overly specialized.

Second, any work using DICE as its basis must take care to use discount rates consistent with the growth rates which occur during the model's optimization. This paper points out that not only can the use of fixed discount rates lead to a wide range of outcomes, which has been previously documented, but it will also produce results inconsistent with the model's optimization. The application of fixed discount rates to SCC estimates which wholly or partially from DICE is an ongoing problem in government and academic research.

While this paper focuses on wildfire emissions, the methods presented here could be applied to other land processes with long emissions curves, which should be subject to the same pitfalls if post-event emissions are not considered, or the wrong discount rate is applied. For example, emissions are positive for several years after forests are clear cut (Howard et al. 2004, Davis et al. 2009) and then may slowly become negative if forests are allowed to grow back. This paper provides a methodological roadmap so that the social costs of complex emissions curves from fire and other land disturbance events can be valued and incorporated into policy. When these emissions costs are excluded, the total costs of inaction and the benefits of proactive policy are significantly undervalued. The climate costs of wildfires are very large, and it is imperative that they be incorporated into policy decision making, both consistently and accurately.

Appendices

Appendix 1: Chapter 1 Supplementary Tables and Figures

Table S1.1: Fire Weather Variable Definitions
(Source: Natural Resources Canada)

Abbreviation	Description
DC	Drought Code - rates average moisture content of deep, compact organic soil layers
DMC	Duff Moisture Code - rates average moisture content of loosely packed organic soil layers of moderate depth
FFMC	Fine Fuel Moisture Code - rates the moisture content of litter
ISI	Initial Spread Index - combines wind and FFMC to rate the expected rate of fire spread
BUI	Buildup Index - combines DC and DMC to rate the amount of available fuel
FWI	Fire Weather Index - combines ISI and BUI to rate fire danger and intensity
DSR	Daily Severity Rating - modifies FWI to rate the difficulty of suppressing fires
T	Temperature
RH	Relative Humidity - the ratio of the water vapor pressure in the air relative to the pressure of water at a given temperature
VPD	Vapor Pressure Deficit - the difference between the measured amount of moisture in the air and its maximum capacity, based on temperature

Table S1.2: Conversion Dates for the Modified FMZ
(Source: PC, Robert Schmoll)

Year	Conversion Date	Notes
2007	10-Jul	With the exception of Kanuti Refuge
	20-Jul	Kanuti Refuge
2008	10-Jul	With the exception of Kanuti Refuge
2009	21-Jul	
2010	10-Jul	
2011	6-Jul	
2012	10-Jul	
2013	10-Jul	
2014	10-Jul	
2015	17-Jul	

Table S1.3: Summary Statistics

variable	Mean	Standard Deviation	Median	Minimum	Maximum
Total Acres	9081.814	32829.740	51	0	517078
Estimated Total Cost (2015 dollars)	1158814	21900000	4275.513	0	594000000
BLM Cost (2015 dollars)	116999	524105	4206	0	8048410
Pct Burned Area Under BLM Protection	0.724	0.442	1	0	1
Cause = Lightning	0.917	0.276	1	0	1
Pct White Spruce	13.946	21.732	2.228	0	100
Pct Black Spruce	25.089	26.238	15.889	0	100
Pct Deciduous	14.407	21.390	5.882	0	100
Pct Grassland	41.029	35.244	31.063	0	100
Pct Burnable	95.473	16.463	100.000	0	100
Mean DC	195.223	92.051	189.547	4.557	528.324
Mean DMC	21.293	14.590	17.419	0.060	88.061
Mean FFMC	69.371	11.608	69.703	5.749	92.575
Mean ISI	2.219	1.600	1.885	0	10.256
Mean BUI	30.261	18.003	26.565	0.116	113.987
Mean FWI	4.955	4.487	3.748	0	26.327
Mean DSR	0.903	1.268	0.449	0	9.010
Mean Temperature	16.041	3.033	16.017	6.794	26.758
Mean RH	62.484	9.320	63.438	28.915	97.020
Mean Windspeed	10.222	3.749	9.964	2.007	27.215
Mean Snow Depth	0.004	0.011	0	0	0.073
Mean VPD	761.771	300.955	713.314	33.370	2416.391

Table S1.4: Balance Table

	Critical	Full	Modified	Limited
Burned acres	2580.143	4813.971	4465.421	12064.380
	(9331.655)	(18198.690)	(16412.490)	(39639.930)
Pct Black Spruce	21.195	26.419	21.356	25.705
	(27.646)	(27.958)	(23.960)	(26.015)
Pct Deciduous	13.405	21.311	10.210	12.969
	(26.755)	(28.902)	(18.103)	(17.835)
Pct Grassland	21.798	27.708	52.872	44.059
	(31.798)	(30.820)	(36.233)	(34.973)
Pct Burnable	91.428	92.238	96.677	96.572
	(26.032)	(22.238)	(11.176)	(14.120)
Mean DC	147.976	177.727	173.989	209.109
	(89.435)	(85.590)	(88.186)	(92.693)
Mean DMC	20.711	23.697	23.106	20.029
	(13.622)	(17.249)	(13.474)	(13.712)
Mean FFMC	73.678	71.883	72.768	67.428
	(12.295)	(12.252)	(11.172)	(11.031)
Mean ISI	2.481	2.648	2.524	1.977
	(1.645)	(2.037)	(1.620)	(1.353)
Mean BUI	28.759	32.127	32.418	29.158
	(17.860)	(19.638)	(17.833)	(17.374)
Mean FWI	5.221	5.938	5.592	4.436
	(4.637)	(5.634)	(4.493)	(3.909)
Mean DSR	0.867	1.199	0.984	0.779
	(1.144)	(1.688)	(1.354)	(1.042)
Mean Temp	16.102	16.796	16.655	15.620
	(3.840)	(3.331)	(2.822)	(2.846)
Mean Relative Humidity	58.404	59.617	61.772	63.906
	(10.710)	(11.038)	(8.741)	(8.356)
Mean Wind Speed	8.743	10.118	11.112	10.128
	(2.801)	(4.340)	(3.618)	(3.557)
Mean Snow Depth	0.00229	0.00299	0.00238	0.00486
	(0.00718)	(0.00953)	(0.00862)	(0.01161)
Mean VPD	833.213	856.166	797.257	715.496
	(390.185)	(362.851)	(307.065)	(257.525)

Appendix 2.1: Chapter 2 Supplementary Methods

Adjustment for Low FRI

Recently burned stands are generally less flammable than mature forests due to fuel limitations and, in some cases, less flammable deciduous vegetation (Rogers, Randerson et al. 2013, Bernier, Gauthier et al. 2016, Beverly 2017). In Alaska, mature forests are typically dominated by evergreen conifers such as black spruce (*Picea mariana*) and white spruce (*Picea glauca*), whereas early successional forests after fire can be fuel limited and have a higher fraction of less-flammable deciduous tree species, such as trembling aspen (*Populus tremuloides*) and birch (*Betula spp.*). Rogers et al. (2013) estimate the probability of a stand burning as a function of stand age. Since our model only estimates fire return interval (FRI) and not average stand age for each grid cell, we ran a linear regression using simulated stand age and FRI results to estimate the relationship between FRI and stand age. Using these regression results (Table S2.1 in Appendix 2.2), we transformed the results from Rogers et al. (2013) to derive a fire probability for FRIs up to 85 years. No adjustment is applied after 85 years.

The results in Rogers et al. (2013) give the probability of a stand burning each year given its age. We normalize the fire probabilities from Rogers et al. such that FRIs > 85 years have an adjustment factor of 1, which declines toward 0 as FRI decreases (Figure S2.1 in Appendix 2.2). For grid cells with short FRIs across the last 30 historical/simulated burning years, we select the fire size for each ignition as described in the primary methods section, and then decrease that fire size by the adjustment factor to account for the depressing effect of frequent burning on subsequent fires.

Literature Projecting Alaska's Burned Area Trends

We initially searched in Google Scholar using the terms "fire" or "wildfire," "Alaska," and "project" or "model" or "predict," following Phillips et al. (2022). We included only papers that project burned area for the whole state through the end of the century and provide time series of burned area changes. The seven papers we represent a wide range of climate scenarios and changes in burned area, summarized in Table S2.3. The great majority of climate scenarios within these papers project modest to large increases in burned area between the historical period or present and the end of the century.

Since each paper provides a different metric of burned area changes in the text, and some only give qualitative descriptions, we extracted data points from the provided figures and ran a simple regression of burned area on year to determine each paper's projected burned area increase for our projection period. Using the results, we calculated expected percent increase in burned area from 2020 to 2050 and 2100. Average percent increases using different weighting options can be found in Table S2.4. The preferred option is to weight each paper equally, so as not to overrepresent papers with more projections. Giving each observation instead of each paper equal weight yields similar results. Weighting each fire model equally suggests a much larger increase in burned area, but we find this approach less defensible since 19 of the 23 observations use the same fire model, ALFRESCO.

Summary of Methods in Phillips et al. (2022) for Estimating the Causal Effect of Management Spending on Fire Size

It is generally difficult to obtain a causal estimate of the effectiveness of wildfire management on reducing fire size, in part because larger fires typically cost more to manage. This leads standard

OLS estimation to be plagued by reverse causality and yield coefficient estimates with the wrong sign. In Phillips et al., the authors use a 2SLS Instrumental Variables (IV) approach to avoid this issue. In the first stage, they estimate the independent component of fire cost using Fire Management Zone (FMZ) as the instrument, while controlling for additional variables including measures of fire weather, temperature, relative humidity, windspeed, elevation, vegetation type, month, and year. FMZ is strongly correlated with cost due to the different standard responses across zones (relevance restriction), but FMZ boundaries are established before the fire season begins and do not change significantly between seasons, so they should be independent of fire size (exclusion restriction). In the second stage, the authors regress burned area on the estimated values of cost from the first stage as well as the control variables and find that a 1% increase in management expenditure leads to a 0.2063% decrease in burned area. We include this summary as useful background for understanding the source of one very important model parameter, management effectiveness. More thorough treatment of the methods can be found in the body and supplement of Phillips et al (2022).

Rationale for the Use of DICE for SCC Values

Since this research concerns US policy, our first choice was to use the SCC values published by the Interagency Working Group (IWG) on the Social Cost of Greenhouse Gasses for use in policy. However, currently the IWG only publishes SCC estimates through 2050 while our analysis extends through 2100, so we turn to the best available alternative, DICE. IWG averages the SCC results of DICE and two other IAMs, so relying solely on the DICE values is reasonably consistent with IWG results. DICE has two modelling options: the optimized version assumes that significant mitigation is undertaken in order to maximize economic wellbeing, while the non-optimized version assumes that mitigation is undertaken only at a very low level and that temperature rise by the end of the century will be in line with BAU projections. We use the non-optimized SCC values, since a BAU climate trajectory is consistent with burned area assumptions elsewhere in our model. Code to run DICE 16 (the most updated version) is publicly available at https://williamnordhaus.com/dicerice-models

Caveat about Dynamic Optimization

We faced a limited tradeoff between optimization and tractability. Ideally, management would be chosen not to minimize costs in each year separately, but rather to minimize costs across the entire projection period, since including feedbacks for low FRI means that management choices may affect burned area outcomes in subsequent years. To account for this, we considered a dynamic optimization approach, which would run the 80-year projection, adjust management in every year, and continue to iterate until an optimal solution for the whole period was reached. Our model structure, however, includes significant stochasticity, and does not tightly converge within model iterations. Thus, dynamic optimization across the entire period is infeasible, and we chose instead to optimize in each year.

Appendix 2.2: Chapter 2 Supplementary Tables and Figures

Figure S2.1: Fire probability adjustment given FRI

Table S2.1: Stand Age to FRI

	FRI
Stand_age	0.9947
	(0.0200)**
_cons	25.3266
	(7.4436)**
R^2	0.97
N	75

* p<0.05; ** p<0.01

Table S2.2: Historical fire size time trends

	ln(burned area)	ln(burned area)
Year	-0.0043	0.0331
	(0.0075)	(0.0031)**
_cons	3.8140	-71.5721
	(14.8868)	(6.2968)**
R^2	0.678e-5	0.0067
N	6989	16382
Annual increase in fire size	-0.43%	3.37%
Year range	1970-1989	1990-2019

* p<0.05; ** p<0.01

Table S2.3: Papers evaluated to estimate projected burned area increases, 2020-2100

Paper	Historical data range	Projection year range	Climate model and scenario	Implied temperature increase (Deg C)	Extracted percent change in burned area, 2020-2100	Fire modeling approach
Bachelet et al, 2005	1950-1999	2000-2099	CGCM1	5.8	270	MC1
			Hadley CM2 SUL	2.8	530	
Genet et al, 2017	1950-2009	2010-2100	CCCMA A1B	3.4	21	ALFRESCO, DOS-TEM
			CCCMA A2	4	124	
			CCCMAB1	1	37	
			ECHAM5 A1b	3.47	39	
			ECHAM5 A2	4.1	74	
			ECHAM5B1	2.2	9	
Euskirchen et al, 2009	1961-1990	2003-2100	Hadley CM3 A2	4.8	51	ALFRESCO, TEM
			Hadley CM3 B2	3.3	42	
			PCM A2	3	114	
			PCM B2	2.2	67	
Pastick et al, 2017	1961-1990	2010-2100	CCCMA A1B	3.4	58	ALFRESCO, DOS-TEM
			ECHAM5 A1B	3.47	9	
Balshi et al, 2009	1960-2002	2006-2100	CGCM2 A2	4.4	139	MARS
			CGCM2 B1	2.2	86	
Rupp et al, 2016	1950-2009	2000-2099	CGCM3.1 AIB	2.6	32	ALFRESCO
			CGCM3.1 A2	3.5	140	
			CGCM3.1 B1	1.7	71	
			ECHAM5 A1B	3.47	78	
			ECHAM5 A2	4.1	90	
			ECHAM5 B1	2.2	22	
Schultz et al, 2019	1950-2013	2020-2099	RCP6	4	55	ALFRESCO

Table S2.4: Estimated Percent Burned Area and Temperature Increase from Literature Review

Weighting	Temperature Increase 2100	Burned Area Pct Increase 2050	Burned Area Pct Increase 2100
Weight each observation equally	3.266	35.171	93.789
	(1.079)	(41.605)	(110.946)
Weight each paper equally	3.474	42.433	113.155
	(1.076)	(50.507)	(134.684)
Weight ALFRESCO, MARS, MC1 equally	4.110	75.212	200.565
	(1.338)	(71.390)	(190.373)
Median	3.4	25.158	67.088
25th Percentile	2.2	13.751	36.669
75th Percentile	4.0	42.763	114.035

Appendix 3: Chapter 3 Calculations

Conversion from variable discount rate to equivalent fixed rate:

$$\sum_{t=1}^{T} Z(t) \;/\; \Pi_{t=1}^{t}(1 + r_t) = \sum_{t=1}^{T} Z(t) \;/\; (1 + R)^t$$

Z(t) is the stream of payments

r_t is the period-specific interest rate

R is the equivalent fixed rate

Calculation of national fire emissions cost:

$$C = \sum_{i=1}^{3} BA * C_i * F_i$$

C is the estimated national cost

i is fire severity, which can be three values: low, moderate, and high

BA is annual national burned area, in thousands of hectares

C_i is the emissions cost per thousand hectare fire of severity i

F_i is average fraction of national burned area of severity i, 2010-2020

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