Optimizing forest management in consideration of environmental regulations, economic constraints, and ecosystem services

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Abstract

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Forest management is a multi-objective problem. Managers must plan treatments and harvests while accounting for multiple factors such as environmental regulations, economic and silvicultural constraints, and ecosystem services. Decision support in forestry management can benefit from a comprehensive approach to develop strategies that meet all the constraints while also accounting for their effects on ecosystem services. My research provides practical methods for decision support in forest management that can enhance forest stewardship and facilitate planning and communicating proposed actions and their effects to the public.

Optimization can be effectively applied to solving such management problems, and can serve as a basis for an integrated approach. Mathematical models have the benefit of low cost associated with multiple reruns of an existing model, and often the addition of new conditions and/or data does not require changing the model's entire structure. Changes to the model's input and conditions are reflected in its output values, and these results can be conveniently summarized and interpreted for use in forest management. These qualities can be used to analyze how the provision of ecosystem services changes depending on different management decisions, or strategies that are subject to multiple environmental regulations and constraints.

My research resulted in a number of practically applicable methods and conclusions to facilitate planning and decision making in forest management. Approaches proposed in this dissertation demonstrated the potential to 1) evaluate environmental regulations with regard to their unintended

effects on land management as well as their effectiveness in achieving their intended goals; 2) identify the guiding factor for allocating fuel treatments to reduce fire hazard landscape-wide; 3) estimating environmental risks associated with fuel treatments; and 4) quantify tradeoffs between management objectives. The mathematical programs described here produce management plans, allowing managers to analyze the effects and assess the tradeoffs of different strategies on ecosystem services and land management, rather than designing and evaluating such strategies manually. The practical results are useful not only to achieve the stated management goals but also for communicating the potential consequences of the proposed actions to stakeholders and evaluating the possibilities for providing ecosystem services for sale on the market.

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GLOSSARY [1]

ACTIVE CROWN FIRE: a fire in which a solid flame develops in the crowns of trees, but the surface and crown phases advance as a linked unit dependent on each other.

ARM: Area Restriction Model, a harvest scheduling model in which the maximum clearcut size is constrained.

BURN PROBABILITY: the probability that wildfire burns a specific area in a specific time period. Burn probability at each point on the landscape is calculated as the proportion of fires that reach this point to the total number of fires.

CLUSTER: a set of contiguous management units, with a combined area that just exceeds a given size restriction.

CONSTRAINTS: inequalities or equations in a mathematical model that define the set of all feasible combinations of decision variables by restricting their values.

CPLEX (IBM ILOG CPLEX): an optimization software package for solving integer and linear programming problems.

CROWN BULK DENSITY: the mass of available canopy fuel per unit crown volume.

CROWN FIRE: a fire that advances from top to top of trees or shrubs, more or less independent of surface fire.

[1]The terms defined in the glossary are applicable to all chapters in the dissertation. Terminology used in Chapter 1 is defined within the chapter and the definitions are applied within the chapter exclusively.

DECISION VARIABLE: a variable in a mathematical program that represents a management decision. The set of all possible values that decision variables can take is determined by the model's constraints.

ECOSYSTEM SERVICES: defined by the MEA as "benefits people obtain from ecosystems. These include provisioning services such as food, water, timber, and fiber; regulating services that affect climate, floods, disease, wastes, and water quality; cultural services that provide recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation, photosynthesis, and nutrient cycling" (MEA, 2005).

ENVIRONMENTAL REGULATION: a legal act that regulates human activities with the goal of reducing their negative impact on the environment.

FIRE HAZARD: defined by The National Wildfire Coordinating Group as "a fuel complex, defined by volume, type, condition, arrangement and location that determines the degree of ease of ignition and the resistance to control" (Glossary of Wildland Fire Terminology, 2012).

FIRE RISK: the chance of fire occurrence given the presence of fuels and weather conditions to create ignition.

FLAME LENGTH: the distance from the ground to the tip of the flame at the leading edge of the flame.

FUEL MODEL: a combination of fire behavior and fuel characteristics that are often used to describe the fire behavior in forests (Anderson, 1982).

MAT: the maximum total area that can be treated in each period.

MATHEMATICAL MODEL: a method to describe real world systems and processes using mathematical terms.

MATHEMATICAL PROGRAM: a set of functions that represent objectives and a set of constraints that represent restrictions on decision variables whose values are optimized.

MAXIMUM CONTIGUOUS TREATED AREA (MCTA): the maximum contiguous area treated according to a certain treatment scenario in the same planning period.

OBJECTIVE FUNCTION (OBJECTIVE): in optimization, a mathematical expression of the optimization criterion that uses values of the decision variables as input. Objective functions represent management goals in mathematical terms.

OPTIMIZATION: a mathematical method to find those values of decision variables that achieve the best value of the objective function(s) while still satisfying the model's constraints.

PASSIVE CROWN FIRE: a fire in the crowns of trees in which individual trees or groups of trees torch, ignited by the passing front of the fire.

PLANNING HORIZON: the time frame within which forest management activities are planned, divided into planning periods of equal length.

PLANNING PERIOD: the discrete time period for which forest treatments and/or other management activities are planned.

PRESCRIBED BURNING: application of prescribed fire, which is any fire ignited by management actions to meet specific objectives, e.g. surface fuel removal.

SURFACE FUELS: fuels lying on or near the surface of the ground, consisting of leaf and/or needle litter, dead branch material, downed logs, bark, tree cones, and low stature living plants.

TREATMENT: a silvicultural activity that is planned at the stand level according to the stand's characteristics with the goals of enhancing forest health and productivity, or to ensure stable financial returns. Examples of treatments are weeding, thinning, fuel piling and burning, and prescribed underburning.

TREATMENT ALLOCATION: an assignment of treatments to forest management units in the landscape.

TREATMENT SCENARIO: a set of management actions that are applied to a forest management unit to achieve certain management goals, e.g. fuel reduction.

TREATMENT (MANAGEMENT) UNIT: a unit of forest land that has similar silvicultural characteristics in terms of vegetation parameters, such as basal area, age, species, and forest structure.

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Chapter 1

INTRODUCTION

Human activities affect the environment to a great extent, causing ecological problems of concern at the international level (Crowley, 2000; MEA, 2005; Union of Concerned Scientists, 2012; World Wildlife Fund, 2013). Natural resource withdrawal and utilization to serve human purposes have resulted in destructive changes in marine and terrestrial ecosystems: water pollution from industrial sources causes not only loss of viable habitat and extinction of rare species, but also a decrease in potable water quality (Dudley and Stolton, 2003). The problem of global warming caused by the excessive emissions of green-house gases has become an internationally recognized issue (United Nations, 1998, 1992). National governments also recognize the need for environmental policies to mitigate the negative consequences of human activities (Clean Water Act, 1972; Clean Water Act Canada, 2006; Conservation Act, 1987; Threatened Special Protection Act, 1995).

Disrupted ecosystems cannot provide the ecosystem services essential for life on Earth. Ecosystem services are broadly defined as benefits that humans obtain from ecosystems (MEA, 2005). The MEA defines four groups of services: provisioning, supporting, regulation, and cultural. All living creatures are dependent on the sustainable provision of ecosystem services, which not only make life on Earth comfortable and enjoyable, but even possible due to the availability of clean air and potable water. Forest ecosystems provide services of all four types and a wide range of resources as well (Smith et al., 2011); thus, protection of forest ecosystems and their services is addresed in multiple environmental laws (Clinton and Gore, 1993; Dudley and Stolton, 2003).

Public awareness of environmental problems has been addressed in multiple environmental policies. In the United States, environmental laws and regulations guide forest management in both public and private forests to ensure that management activities do not adversely affect forest ecosystems (Endangered Species Act, 1973; Multiple-Use and Sustained-yield Act, 1960). Voluntary mechanisms, such as forest certification standards promote sustainable forest management that prioritizes the social, economic, and ecological values of forests (Forest Stewardship Council,

2010; Sustainable Forestry Initiative, 2010).

Public demand for ecosystem services has resulted in a change of management paradigm in the United States over the last couple of decades (Gilmour et al., 2007; Davidson-Hunt and Berkes, 2001; Petersen, 2011). National forest management prioritizes conservation and supports restoration programs to encourage and facilitate efforts to improve the health of forest ecosystems so that they provide a variety of services. The provision of ecosystem services is important not only in public forests: private forest owners are interested in increasing provision when they can be reimbursed for it (Deal et al., 2012; USDA Farm Service Agency, 2013). Land owners can be compensated for protecting, enhancing and restoring forests through conservation incentives, tax credits, and subsidies (USDA Forest Service, 2009).

Environmental regulations, economic constraints and the provision of ecosystem services influence decision making in forest management. Because a number of regulations guide forest management regardless of ownership, it might be unclear whether and how combinations of these regulations affect management and ecosystem services. Similarly, it might not be possible to achieve all management objectives simultaneously due to potential tradeoffs between them. For example, thinning to reduce canopy closure can effectively prevent crown fire and spread of insects and diseases, but it might increase the risk of windthrow (Castedo-Dorado et al., 2009; Hayes et al., 1997; Mikita et al., 2012; Pollet and Omi, 2002). Similarly, fuel treatments remove surface fuels and reduce fire hazard, but can increase sedimentation and water pollution (Cram et al., 2007; Elliot, 2005; Neary et al., 2003; Reid, 2010; Rhodes, 2007; USDA Forest Service, 2005; Wright et al., 1976). The tradeoffs depend not only on the treatment design but also on its frequency, intensity, and timing.

Forest and natural resource management, then, is complex and often includes multiple objectives. Forest managers face dilemmas about how to allocate resources in such a way that multiple objectives can be met with minimum adverse effects on the ecosystem. Mathematical programs can be effectively used to address this multi-objective problem and provide approaches for decision support and design of sustainable forest management plans (Kennedy et al., 2008; Tóth et al., 2006; Tóth and McDill, 2009). Mathematical programs are based on models representing ecosystem processes, and the design of both requires detailed data and is often time consuming. Mathematical models, however, can be reused for similar projects with no additional cost and refined using data

that become available later. Similar to field experiments and simulations that provide the actual and projected data under different management scenarios, respectively, mathematical models are used to study trends and predict the behavior of ecosystems under different pre-defined management scenarios, and to design such scenarios.

In my research, I focused on designing approaches to assist forest managers in planning management activities in consideration of both environmental regulations and the provision of ecosystem services. The integration of these factors allows a comprehensive analysis of the potential effects of management on the forest ecosystem, informing decision making for achieving broad conservation goals.

1.1 The key concept: Ecosystem services

Ecosystem services are broadly defined as benefits that can be obtained from forest ecosystems (MEA, 2005). There are four groups of ecosystem services according to MEA: provisioning, supporting, regulating, and cultural. Provisioning ecosystem services are represented by timber products, food and biochemical resources. Forested watersheds are valued for the water quality and wildlife habitat that they provide; these are supporting ecosystem services. The group of regulating ecosystem services includes soil formation and nutrient cycling. Finally, recreational opportunities and spiritual values of forests form the group of cultural ecosystem services.

The variety of ecosystem services that forests provide, and the growing demand for these services, makes decision making in forest management a complex problem. Provision of ecosystem services might compromise the achievement of other management goals. In addition, forest owners face dilemmas regarding the provision of ecosystem services that potentially compromise each other. There is a need to assess what services can be provided, and whether it is possible to maximize their provision simultaneously or if there are tradeoffs that should be accounted for. Having this information, forest managers and owners can consider stacking or bundling ecosystem services.

When multiple services can be provided as a joint product, or bundle, managers target just one service, while the provision of the others is complimentary. For the studies in my dissertation, the choice of ecosystem services was based on the key functions of the sites and primary public concerns regarding the areas. Multiple services could be provided jointly; thus, the choice was made in favor of the most representative one. For instance, the service of water quality represents not

only drinking water quality but also potential freshwater habitat. The requirements for habitat are more relaxed, so it did not need to be considered independently. Similarly, an increase of the area in native forest is broadly representative of a number of services, including recreational opportunities and wildlife habitat.

1.2 Contribution

My research has resulted in several methodological approaches that can be applied in practice for building sustainable forest management strategies. Broadly, the studies in my dissertation:

- provide **analytical and methodological approaches** to integrate ecosystem services to forest management and account for other factors that can affect the provision of the services and achievement of other management objectives.

- provide approaches for **planning** forest management actions under various constraints and objectives, as well as examples of their application. The management plans obtained using these approaches are spatially and temporally explicit allocation of treatments and/or harvests that meet a number of constraints and management goals, including the provision of ecosystem services.

- provide information about potential tradeoffs between management objectives to facilitate **decision making**. Such information is essential for choosing a management strategy that is associated with the least environmental risks and ensures the maximum provision of ecosystem services.

- generate information regarding the social and ecological benefits of each management plan and facilitate **communication among stakeholders**. Such data is important for presenting and advocating the choice of management strategy at stakeholder meetings.

- contribute to the development of **marketing strategies** and assess the opportunity of selling ecosystem services, alone or in bundles, at the market.

- contribute to **modeling**. Mathematical models built in the studies can be reused in similar projects to design management plans to satisfy multiple objectives, quantify tradeoffs between these goals and analyze the effects of environmental regulations.

1.3 Overview

Although each chapter of my dissertation stands alone, all of them are united by the common theme of forest management guided by the provision of ecosystem services.

Chapter 2 proposes a framework to analyze the unintended consequences of environmental regulations and to provide insights on whether the forest owner can provide ecosystem services above the level required by the regulations, perhaps offering these services on a market. The framework uses the structure of a mathematical program to express environmental regulations as constraints and management goals as objective functions. A case study in a forested catchment in New Zealand demonstrates the potential of the proposed approach to evaluate environmental regulations and policies.

Chapter 3 introduces and evaluates models that optimize fuel treatment allocations for fire hazard reduction. The models produce spatially and temporally explicit patterns of treatment allocations and allow analysis of the effects of these allocations on fire hazard reduction at the landscape scale. The results help identify key factors for planning fuel treatments to reduce fire hazard and evaluate the expected performance of treatments at the landscape scale.

In Chapter 4, I use results from Chapter 3 to build optimization models with multiple objectives in order to analyze the provision of ecosystem services under different fuel treatment scenarios. I compare the effects of the treatments on a forest ecosystem, both short and long term, and identify and quantify tradeoffs between the management objectives under different constraints on total area treated.

The studies in my dissertation use data with direct application to natural resource management, and the results I obtained are applicable to decision support and analysis. Forest managers define the constraints that must be met, whether economic, ecological, social or some combination. The models incorporate all the data and constraints and optimize forest management activities to increase the provisioning of ecosystem services. The studies provide approaches and data for well-informed decisions applicable to sustainable forest management and planning.

Chapter 2

AN OPTIMIZATION APPROACH TO EVALUATE THE IMPACT OF ENVIRONMENTAL REGULATIONS ON FOREST ECOSYSTEM SERVICES

2.1 Introduction

A common goal of many environmental regulations and policies is to increase the provision of forest ecosystem services; however, the combined effects of these regulations is unknown. While several authors (e.g., Hof and Bevers, 2000; Nalle et al., 2005; Zhu and Bettinger, 2008) have evaluated the economic and ecological implications of individual environmental regulations, such as green-up constraints, a comprehensive approach to assess interactions among these regulations is missing. To fill this gap, I propose an optimization approach that uses the structure of a mathematical program. Structural components of the proposed optimization approach, constraints and objective functions, represent environmental regulations and ecosystem services, respectively. The interaction of the constraints in a mathematical program results in changes to the values of the objective functions; this feature is used in the approach to capture the impact of combinations of environmental regulations on ecosystem services. I show how the approach works in a forested catchment in New Zealand, where it is unclear how the interaction between nitrogen emission regulations, deforestation taxes, and the economic reality of timber and dairy production affect water quality. This case study provides an example of the approach's application and demonstrates its potential to assess the effects of environmental regulations on the provision of ecosystem services.

In many countries, environmental regulations and certification standards aim to promote the provision of ecosystem services as a response to growing awareness of the environmental issues. The provision of forest ecosystem services, broadly defined as benefits obtained from forests (Brown et al., 2007; MEA, 2005) attracts public concern because many forests are being managed primarily for timber revenue, with little regard for other services (U.S. Environmetal Protection Agency, 2013). Two primary mechanisms exist that aim to resolve this conflict: one mandatory, such as environmental regulations, the other voluntary, such as forest certification standards. For example,

environmental authorities in New Zealand have responded to public concerns about water quality decline due to expansion of dairy farms by regulating land uses that increase nitrogen emissions (Hayward, 2008). Clean Water Act of the US and Clean Water Act of Ontario, Canada (Federal Water Pollution Control Amendments of 1972; Clean Water Act, 2006) are laws that control water quality by restricting the release of high quantities of pollutants into water. In the Russian Federation, over a hundred federal laws and directives govern utilization, water quality monitoring, restoration, and protection of water resources to ensure sustainability of these resources (ConsultantPlus, 2013). Worldwide, there has been an attempt to protect the remaining habitat of endangered and threatened species and increase their populations (e.g., Endangered Species Act (1973) in the US, Environment Protection and Biodiversity Conservation Act (1995) and "Protecting Our old-Growth Forest" Policy (2001) in Australia, Conservation Act (1987) in New Zealand). Forest certification schemes, such as those internationally administered by the Forest Stewardship Council (2010) or the Sustainable Forest Initiative (2010) promote the protection of a wide range of ecosystem functions through third-party monitoring of a set of management principles and criteria.

Environmental regulations may have unintended consequences on forest ecosystem services, especially if they are applied in combination with other regulations or in the presence of market forces. Old growth forests and wildlife habitat are promoted by environmental regulations (e.g., Endangered Species Act 1973; Multiple-Use Sustained-Yield Act 1960; Northwest Forest Plan, 1993; proposed Old Growth Protection Act (Canada)); however, such management may inadvertently affect plant and animal species, e.g. the bobcat (*Lynx rufus*) in Western North America, whose primary prey depends on young forests and edge habitat (Chamberlain et al., 2003). Regulation pressures might backfire if they do not address land use changes on private lands: forest landowners in the Northwest United States often sell their land in riparian areas to developers just to avoid environmental regulations (Bradley et al., 2009). Greater uncertainty is associated with the impacts of combinations of environmental regulations. Similar challenges regarding impact analysis arise in the context of forest certification, where a multitude of management criteria are applied simultaneously while a rigorous quantitative approach that can assess the combined effects on ecosystem services is lacking (Auld et al., 2008).

Models have been proposed for decision support in forest management that account for individual ecosystem services and environmental regulations to schedule management actions to optimize

the layout to best meet a set of objectives. Bettinger et al. (1998; 2003) and Loehle (2000) introduced models that ensure compatibility between timber production and wildlife habitat preservation. Hof et al. (2001; 2000) and Loehle (2000) proposed harvest scheduling models that account for water quantity and quality in terms of sediment level. On the other end of the spectrum are modeling efforts focused on specific environmental regulations, such as maximum clear-cut size restrictions, but these studies fall short of establishing a formal link between regulations and the provision of ecosystem services (e.g. Baskent et al., 2008; Constantino et al., 2008; Goycoolea et al., 2005; McDill et al., 2002; Murray, 1999). A few studies found that the effects of green-up constraints on forest structure and related ecosystem services may vary, and further assessment is needed (Boston and Bettinger, 2006; Nalle et al., 2005; Zhu and Bettinger, 2008). A formal link was established by Tóth et al. (2006) and Tóth and McDill (2009) who used multi-objective optimization to map out efficient frontiers between timber revenues and mature forest habitat area in large contiguous patches in the presence of maximum harvest opening size restrictions. Efficient frontiers, in this context, are sets of harvest schedules where neither timber revenues nor mature forest area can be further improved without compromising the other.

In this chapter, I extend this concept of Pareto-efficiency to analyze the impact of multiple, interacting regulations on the provision of forest ecosystem services using a mathematical programming approach. The approach is designed to simulate the effects of regulations on ecosystem services and analyze the consequences of different management strategies subject to environmental policies and economic reality. I illustrate the proposed method in a river catchment in New Zealand, part of which is managed as a radiata pine (*Pinus radiata*) plantation, and assess the net effect of nitrogen emissions regulations, deforestation taxes, and economics of timber vs. dairy production on water quality.

2.2 Concept

I propose a mathematical programming approach to evaluate the effectiveness of environmental regulations in promoting ecosystem services in the presence of economic goals and constraints. The objective functions of mathematical programs represent management goals and the inequalities act as constraints on these objectives. Management decisions, spatio-temporal assignments of operation to management units, are captured in the form of decision variables that tie the functions to the

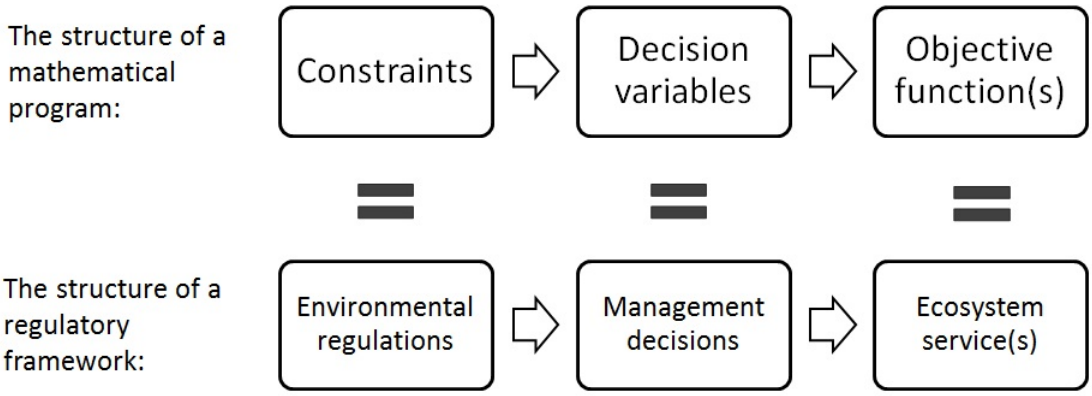


Figure 2.1: Concept of the proposed framework.

inequalities: the constraints imposed on the variables restrict the values of the objective functions. I make use of this structure by defining the provision of forest ecosystem services as objective functions and by using the constraints to represent regulations (Fig. 2.1). This structure can be used to study the impact of environmental regulations and economic policies on forest ecosystem services in the presence of logistical, budgetary or other constraints. I apply the proposed method to determine whether the current regulatory environment is sufficient to prevent additional nitrogen emissions, given expected land use patterns in the Tauranga-Taupo catchment in the central North Island of New Zealand.

2.3 Case study

The 230 km^2 Tauranga-Taupo catchment is located in the Waikato region of New Zealand's North Island (Fig. 2.2). Indigenous forest vegetation covers two thirds of the catchment. This area belongs to the Kaimanawa Forest Park and is managed by the Department of Conservation. The rest of the catchment (approx. 30%) is commercial radiata pine plantation owned by the Maori, the indigenous Polynesian people of New Zealand, and managed by New Zealand Forest Managers Ltd. (Turangi; 2005). In addition to timber production, I considered two ecosystem services that the commercial plantations in the Tauranga-Taupo catchment could provide: indigenous forest restoration and nitrogen emissions control. Native forests are valued for their ecological functions

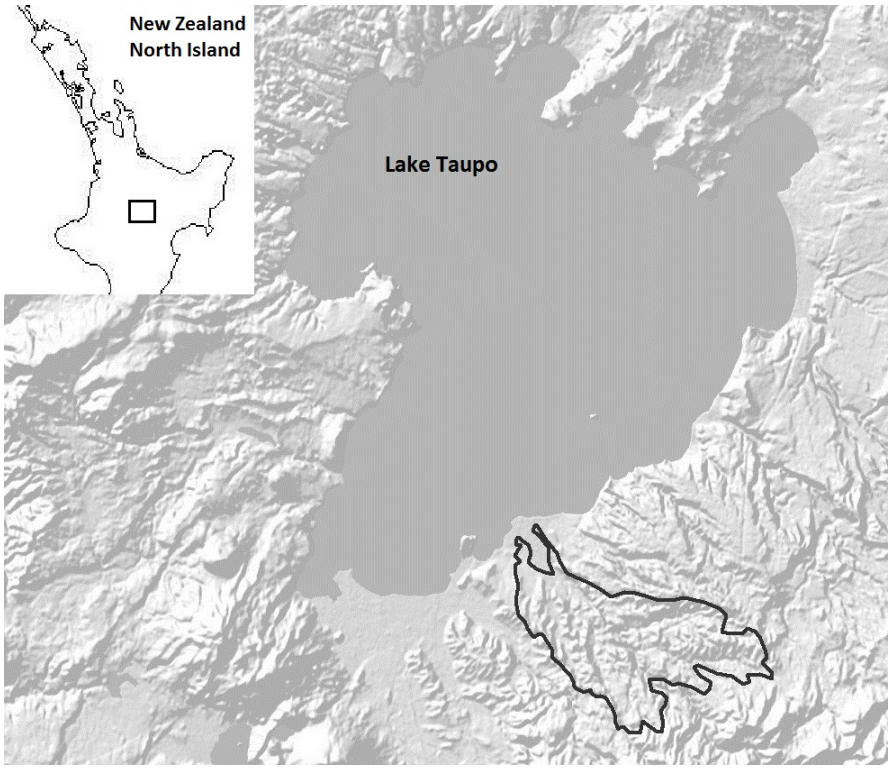


Figure 2.2: Tauranga-Taupo catchment.

and their aesthetic and recreational attributes. However, water quality is one of the major public concerns in the Lake Taupo basin, which contains the catchment. Lake Taupo is the largest lake in New Zealand, has exceptional recreational value, and provides the habitat for a variety of freshwater species, including native crayfish (*Paranephrops planifrons*) and introduced brown and rainbow trout (*Salmo trutta* and *Oncorhynchus mykiss*, respectively) (Armstrong, 1935; Dedual, 1996). Naturally low levels of nitrogen in the lake have increased in the last 30 – 40 years due to nitrogen-rich runoff from dairy farms established in the lake's catchments (Young et al., 2005), often in place of plantation forests that no longer provide satisfactory financial return. It is estimated that the conversion of land to pastoral farming contributed up to 92% of the total manageable nitrogen input to the lake (i.e., nitrogen from human generated sources that can be reduced by applying sustainable management practices or by avoiding conversion; Berry (2008)). Nitrogen emissions from other land uses, such as plantation forestry, are much lower (Hamilton, 2005).

Before I give a formal description of the models that were built to assess environmental regulations in this area, I define the terminology and describe the regulatory environment and data used in this study.

2.3.1 Terminology

A **management unit** is a unit of forest land with uniform topography, age class, and croptype (Fig. 2.3). **Croptypes** refer to areas of forests that are under the same silvicultural regime. Management units belong to one or more **blocks** which are larger sections of forest (Fig. 2.3); the areas and boundaries of the blocks are based on historical and current property lines.

Forest treatments are silvicultural activities that enhance and/or recover forest growth or merchantable yield. Examples are site preparation, planting, thinning and fertilization. **Conversions** are transformations of management units to dairy farms, whereas **restoration** refers to transition from plantations to native forestland. A **prescription** is a sequence of treatments, conversion or restoration that can be assigned to a management unit during the planning horizon.

A **viable dairy unit (VDU)** is a contiguous set of management units whose combined area exceeds a threshold, called **minimum viable dairy unit size**. Finally, a **cluster** is a spatially connected set of management units, the combined area of which is no smaller than the minimum

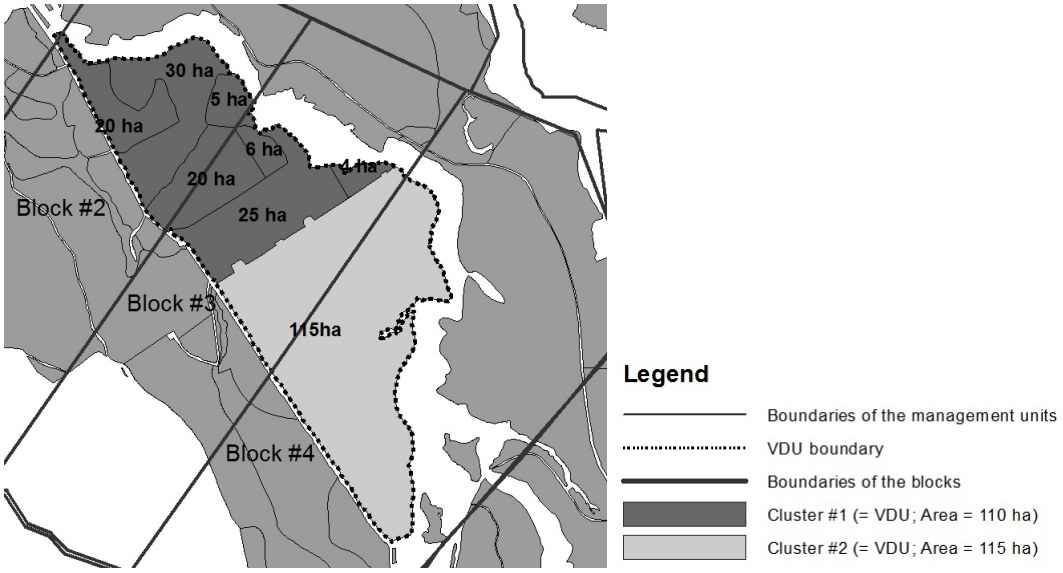


Figure 2.3: Spatial terminology: management units, land blocks, VDU, and clusters.

VDU size (Fig. 2.3). Multiple, adjacent or overlapping clusters can form VDUs.

2.3.2 Environmental regulations

There are two key environmental regulations that affect forest management and land use choices in the catchment: a deforestation tax and limits on average nitrogen emissions into Lake Taupo's water. The deforestation tax is a component of the New Zealand Emissions Trading Scheme (ETS), a price based mechanism developed by the New Zealand Parliament to reduce the net greenhouse gas emissions and comply with the Kyoto Protocol and the United Nations Framework Convention on Climate Change (United Nations, 1998, 1992). Landowners must pay the deforestation tax if they convert forests that had been established before 1990 to farmlands. Post-1989 forests can be converted to dairy farms or other uses tax free, as long as the landowner does not sell the carbon stored on the land. If any carbon has been sold, the owner has to pay the shortfall at the current market price (MAF, 2010). Thus, in pre-1990 forests the deforestation tax is simply the product of carbon price and the volume of carbon stored in the forest at the moment of conversion. This calculation applies to forests older than nine years. Otherwise, the deforestation penalty is calculated based on the full rotation carbon liability, which is equal to the carbon stored in a 28

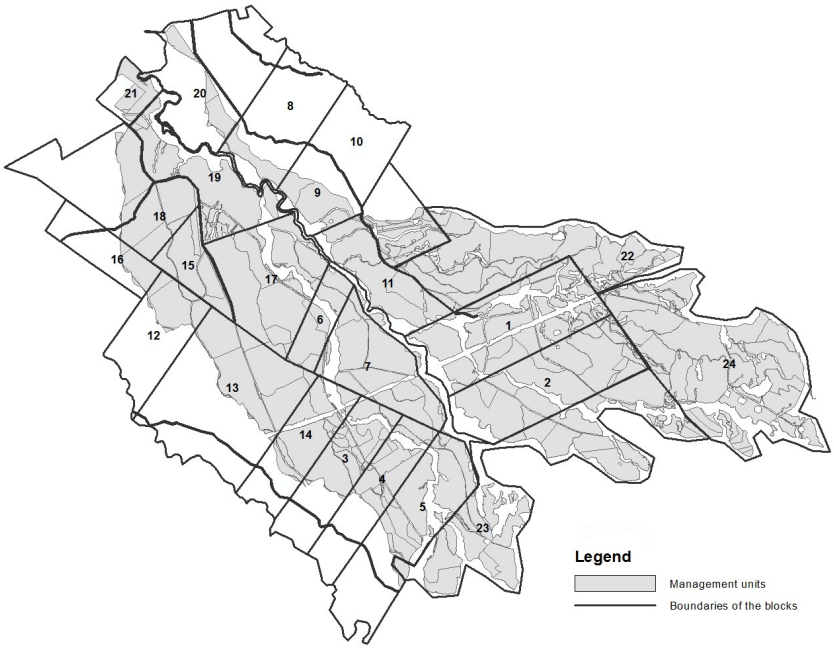


Figure 2.4: Land blocks in Tauranga-Taupo catchment.

year old stand times the carbon price.

Nitrogen emissions in the Lake Taupo basin are regulated by Waikato Regional Council (WRC). In collaboration with other local policy groups, WRC has developed a strategy, "Protecting Lake Taupo" to guide environmental legislation in the area (Environment Waikato 2003, 2011). This strategy regulates nitrogen emissions on a block-by-block basis, where the block boundaries are based on historical property lines. Landowners are allowed to increase average nitrogen emission as long as it does not exceed 5 kg/ha per year from any block (average emission from native and plantation forests is 3 kg/ha per year.). WRC uses a nutrient budgeting model, called $Overseer^{TM}$ (Environment Waikato, 2011), which estimates the nutrient flows in a productive farm and identifies the risk of environmental impacts through calculation of nutrient runoff and leaching. The input parameters for the model include detailed data about the fertilizers, crop production, type of livestock, herd size, and soil types on the farm. For the purpose of this study, I used an average nitrogen emissions rate of 50 kg/ha per year, given that nitrogen emissions range between 30 and 70 kg/ha per year on pastoral farms (Hamilton, 2005).

To see how the nitrogen emission regulation works, consider a 100 ha block of radiata pine plantation. The total nitrogen emission from this block is assumed to be 300 kg per year given the 3 kg/ha per year baseline nitrogen emission from forests. If a 4 ha section is converted to a dairy farm and the rest of the block remains forested, the total nitrogen emissions will rise to 488 kg per year, assuming the average nitrogen emission rate at 50 kg/ha per year from dairy farms. At 4.88 kg/ha per year, this land-use scenario does not violate the allowable maximum of 5 kg/ha per year. However, a land-use change where 5 ha are converted to a dairy farm is infeasible because the total nitrogen emission from the block goes up to 535 kg per year, which exceeds the maximum allowed. If the minimum VDU size exceeds 4 ha, the only way to establish large enough farms on 100 ha blocks without violating the nitrogen emission regulation is to convert adjacent sections of neighboring blocks.

2.3.3 Data

Spatial, biophysical and economic data for the study area, which comprised the commercial plantations of the Tauranga-Taupo Catchment, were provided by the Lake Taupo Forest Trust and the New Zealand Forest Managers Ltd. The former organization represents the Maori owners of the land while the latter group are the land managers. The data included information about management unit and block boundaries (Fig. 2.4), initial ages, areas and croptypes. Cost, revenue and timber volume data was available for each croptype and for each age in one year increments. Restoration cost estimates for planting a mixture of indigenous species in equal proportions was taken from Davis et al. (2009), in addition to a fencing cost that increases by 33% every time the restoration area is doubled. All cash-flows were discounted using an 8% real interest rate. Carbon data was provided by the New Zealand Forest Research Institute (Scion) in ton/ha as a function of the age of the management units.

For simplicity, I excluded all management units whose area were less than one hectare (13 ha in total). Radiata pine covers 99.73% of the study area, 93.5% of which is in second rotation with an age younger than 25 years, the minimum rotation age (Fig. 2.5). Based on stakeholder input, I considered conversion as an option only if certain conditions with regard to soil productivity and slope were met. About a quarter of the study area was too steep for dairy farming and, as a

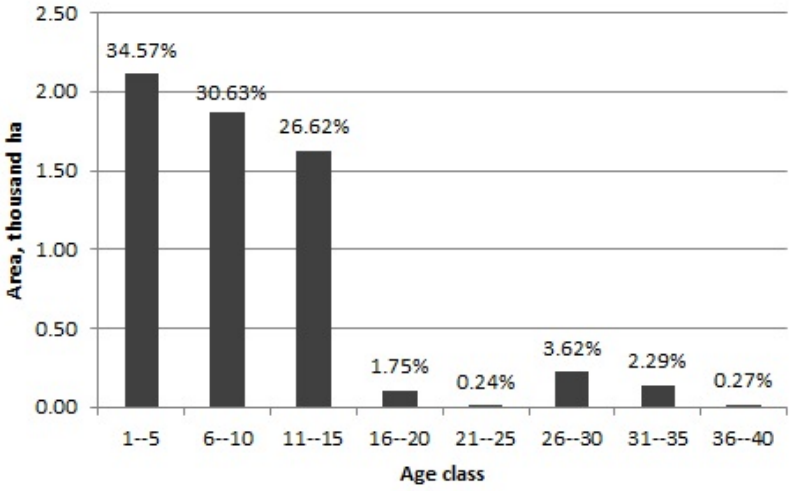


Figure 2.5: Age class distribution of the radiata pine plantations in the Tauranga-Taupo catchment.

consequence, was not associated with any prescriptions that included dairy conversions. The entire study area was eligible for timber harvesting and restoration.

2.3.4 Design of the experiment and model formulation

To evaluate impacts of the regional environmental regulations in the Tauranga-Taupo catchment, I formulated two models: E^- and E^+. E^+ incorporated the regulations described above, whereas E^- did not. Both models included harvest flow, ending inventory and minimum VDU size constraints.

I assumed that the forest is managed by a profit-maximizing decision maker who rationally responds to environmental regulations, developing strategies for subsequent years. The planning horizon consists of ten 5-year long periods. Given that sawmills do not accept timber from a radiata plantation younger than 25 years, a minimum rotation age of 25 years was used, thereby allowing the units to be cut at most twice during the planning horizon. Harvests, conversions and restorations take place at the mid points of the periods, and all costs directly associated with them occur within the period. Timber harvests and restorations do not change nitrogen emissions rates. If a management unit is converted to dairy, nitrogen emissions increase in each affected block in proportion to the overlapping area of the converted unit. Partial cuts, conversions or restorations

are economically infeasible. Lastly, the harvested units regenerate as the same croptype unless they are converted to dairy farms or restored to native forests.

The decision variables represent different prescriptions — treatments, conversions and/or restorations — assigned in one or several planning periods. Consider $x_{73(3,9,10,0)}$ as an example. This binary variable represents the decision whether management unit 73 should be assigned prescription (3,9,10,0) or not: should the unit be harvested in the third and ninth planning periods, and then converted to a dairy farm in period ten. The numbers in parentheses represent the periods of the first and second harvests, the periods of dairy conversion and native forest restoration, respectively. The last "0" in prescription (3,9,10,0) indicates that restoration did not take place, here because the unit has been converted to a dairy farm. Once a unit is converted to dairy or restored to native forest, it is no longer available for any other management activities. If variable $x_{73(0,0,0,0)}$ takes the value of one that means that unit 73 should not be cut, converted or restored during the planning horizon.

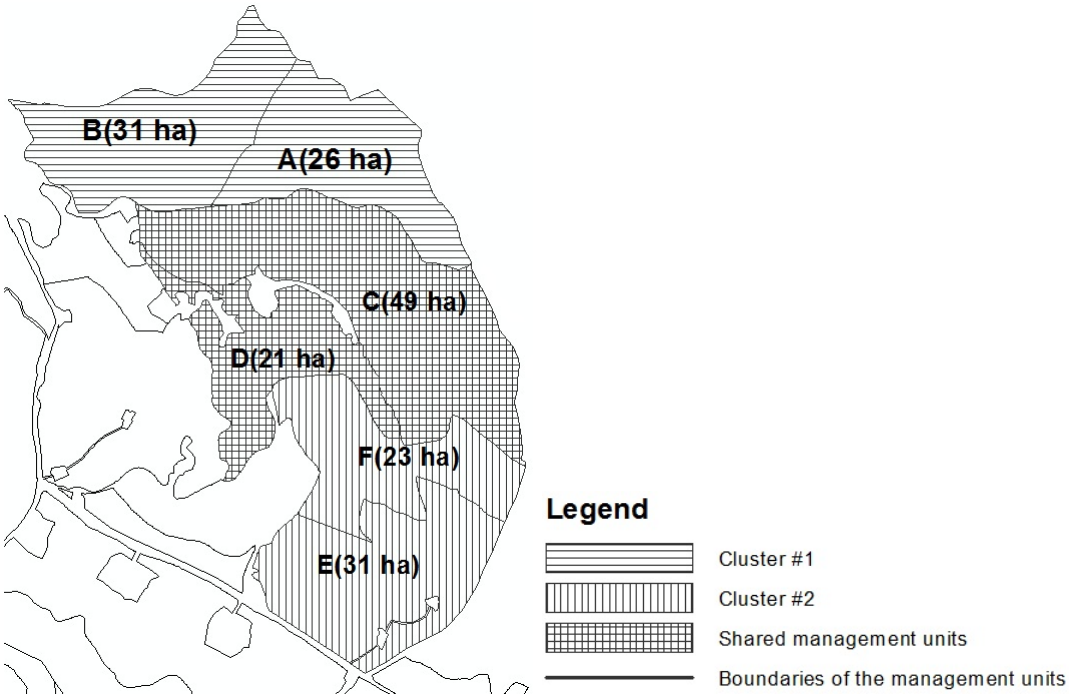


Figure 2.6: An example of cluster enumeration: algorithm ensures that each cluster satisfies minimum VDU size requirement but avoids super-clusters.

Returns from dairy farms are obtained annually starting from the second year after conversion. For minimum VDU size, I used 110 ha based on reports of the New Zealand Ministry of Agriculture and Forestry Ministry of Agriculture and Forestry (2013). To meet this requirement, a management unit may be converted to a dairy farm only if it is part of a cluster that is to be converted to dairy in its entirety in the same planning period (Fig. 2.6). Clusters were enumerated using the algorithm described in Rebain and McDill (2003), and the corresponding constraints ensured the minimum VDU size. The algorithm ignores super-clusters that are unions of other clusters. For example, the cross-hatched area (units A, B, C, D, E, F) on Fig. 2.6 is a VDU that was not enumerated because it can be expressed as the union of Cluster 1 (units A, B, C, and D) and Cluster 2 (units B, D, F, and E), each of which alone can form an independent VDU.

I define Model E^+ as follows:

$$\max \sum_{i \in I} \sum_{j \in E_i} g_{ij} a_i x_{ij} \tag{2.1}$$

$$\max \sum_{i \in I} \sum_{j \in S_i} r_{ij} x_{ij} \tag{2.2}$$

$$\min \sum_{i \in I} \sum_{j \in S_i} n_{ij} x_{ij} \tag{2.3}$$

subject to:

$$\sum_{j \in S_i} x_{ij} \leq 1 \quad \text{for all } i \in I \tag{2.4}$$

$$A_p \sum_{i \in I} \sum_{j \in S_i} a_i v'_{ij} x_{ij} - v_0 \sum_{i \in I} \sum_{j \in S_i} a_i x_{ij} \geq 0 \tag{2.5}$$

$$\sum_{i \in I} \sum_{j \in S_{it}} a_i v_{tij} x_{ij} - h_t = 0 \quad \text{for all } t \in T \tag{2.6}$$

$$b_l h_t - h_{t+1} \leq 0 \quad \text{for all } t \in T \tag{2.7}$$

$$-b_u h_t + h_{t+1} \leq 0 \quad \text{for all } t \in T \tag{2.8}$$

$$\frac{1}{a_b} \sum_{i \in I_b} \sum_{j \in S_i} n_{ij} x_{ij} \leq 5 \quad \text{for all } b \in B, \tag{2.9}$$

$$\sum_{c \in C_i} y_{ct} - \sum_{j \in D_{it}} x_{ij} \geq 0 \quad \text{for all } t \in T,\ i \in I \tag{2.10}$$

$$\sum_{i \in I_c} \sum_{j \in D_{it}} x_{ij} - |I_c| y_{ct} \geq 0 \quad \text{for all } t \in T,\ c \in C \tag{2.11}$$

$$\sum_{i \in I_c} \sum_{j \in D_{it}} x_{ij} - y_{ct} \leq |I_c| - 1 \quad \text{for all } t \in T,\, c \in C \tag{2.12}$$

$$x_{ij}, y_{ct} \in \{0, 1\}, \tag{2.13}$$

where the variables are:

x_{ij} = one if prescription j is assigned to management unit i, and zero, otherwise;

y_{ct} = one if the entire cluster c is to be converted to dairy in period t, and zero, otherwise;

h_t = an accounting variable for the total timber volume harvested in period t.

The parameters are:

I = the set of management units (indexed by i);

T = the set of planning periods (indexed by t);

S_i = the set of prescriptions for unit i (indexed by j);

S_{it} = the set of prescriptions for unit i that involve harvest in period t;

E_i = the set of all prescriptions for unit i that involve native forest restoration;

D_{it} = the set of prescriptions for unit i involving conversion to dairy in period t;

B = the set of blocks (indexed by b);

I_b = the set of units in block b;

C = the set of clusters (indexed by c);

C_i = the set of clusters that contain unit i;

I_c = the set of units that comprise cluster c;

$|I_c|$ = the number of units that form cluster c;

g_{ij} = the age in years of native forest in unit i following prescription j;

a_i = the area in ha of unit i;

n_{ij} = nitrogen emission in kg/ha per year from unit i under prescription j;

A_p = the total area of pine plantations in ha at the beginning of the planning horizon;

a_b = the area of block b in ha;

v_{tij} = timber volume in unit i in m^3/ha in period t given prescription j;

v'_{ij} = timber volume in unit i at the end of the planning horizon given prescription j;

v_0 = timber volume in the entire catchment at the beginning of the planning horizon;

b_l, b_u = the lower and upper bounds on harvest volume fluctuations in adjacent periods;

r_{ij} = the discounted net revenue in NZ$ associated with prescription j for unit i.

Assuming a general prescription of $j = (h_1, h_2, d, f)$, the formula for calculating the revenue coefficients, r_{ij}, was the following:

$$r_{ij} = r_{ij}^1 + r_{ij}^2 + r_{ij}^d - r_{ij}^f$$

where

$$r_{ij}^{h1} = \begin{cases} a_i \left(\frac{r_{ij}^{h_1} - q_i^a - q_i^h}{(1+\alpha)^{5h_1 - 2.5}} - \sum_{k=1}^{5h_1-3} \frac{q_i^{s_k}}{(1+\alpha)^k} \right) & \text{if } h_1 \geq 1 \\ 0 & \text{otherwise,} \end{cases} \tag{2.14}$$

$$r_{ij}^{h2} = \begin{cases} a_i \left(\frac{r_{ij}^{h_2} - q_i^a - q_i^h}{(1+\alpha)^{5h_2 - 2.5}} - \sum_{k=5h_1-2}^{5h_2-3} \frac{q_i^{s_k}}{(1+\alpha)^k} \right) & \text{if } h_2 \geq 1 \\ 0 & \text{otherwise,} \end{cases} \tag{2.15}$$

$$r_{ij}^{d} = \begin{cases} a_i \left(r_d \frac{(1+\alpha)^{51.5-5d} - 1}{\alpha(1+\alpha)^{51.5-5d}} - q_e - p_c w_{ij} \right) (1+\alpha)^{2.5-5d} & \text{if } d \geq 1 \\ 0 & \text{otherwise,} \end{cases} \tag{2.16}$$

and

$$r_{ij}^{f} = \begin{cases} a_i q_r (1+\alpha)^{2.5-5f} & \text{if } f \geq 1 \\ 0 & \text{otherwise,} \end{cases} \tag{2.17}$$

where

$r_{ij}^{h_1}$, $r_{ij}^{h_2}$ = harvest revenues from unit i given prescription j in period h_1 and h_2, respectively;

q_i^a, a_i^h = the timber access and harvesting costs for unit i, respectively;

$q_i^{s_k}$ = the silvicultural cost that occurs in the kth year after the last harvest;

r_{ij}^d = the dairy revenues (NZ$1,000/ha per year) starting in period d;

q_e = the one time dairy establishment cost (NZ$6,000/ha);

p_c = real carbon price (NZ$20/ha);

w_{ij} = the carbon liability in tons in unit i if prescription j is applied;

r_{ij}^f = the cost of native forest restoration in period f;

q_r = the one time native forest restoration cost;

α is the real discount rate.

Function (2.1) maximizes the total area restored to native forest, weighted by the length of the time span between restoration and the end of the planning horizon to prioritize mature forest.

Function (2.2) maximizes the discounted harvest and dairy revenues minus the harvest, conversion, restoration and silvicultural costs.

Function (2.3) minimizes the total nitrogen emissions at the end of the planning horizon.

Constraint set (2.4) is a logical restriction, which allows only one prescription to be assigned to a management unit in the planning horizon.

Ending inventory constraint (2.5) ensures that the average per hectare timber volume left by the end of the planning horizon is not less than that at the beginning of the planning horizon, to prevent the depletion of the commercial timber resource in the catchment.

Constraints (2.6)–(2.8) are flow constraints that work in concert (McDill et al., 2002). Constraint (2.6) calculates the timber volume harvested in each planning period (h_t). The two inequalities ensure that the volume in each period, except the first, does not decrease (increase) by more than a factor of b_l (b_u) relative to the volume in the previous period.

Constraint (2.9) captures WRC's nitrogen emission regulation and prevents the emissions from exceeding 5 kg/ha per year within a block.

Inequalities (2.10)–(2.12), adopted from Tóth et al. (2009), ensure that the requirement of the minimum VDU size is met when management units are converted to dairy. Constraints (2.10) ensure that a management unit is converted to dairy only if it is a part of a cluster which is to be converted in its entirety in the same planning period. Constraint set (2.11) allows, and (2.12) forces, cluster c to be converted if, for every unit in c, at least one of the prescriptions that involve a dairy conversion in period t is selected.

Constraint sets (2.13) define variables x_{ij} and y_{ct} as binary.

Model E^- can be obtained from this formulation by omitting constraint (2.9). Additionally, the coefficients of objective function (2.2) are recalculated so that the deforestation tax is not accounted for in these coefficients: p_c is set to zero in Eq. (2.16).

Both models were solved as multi-objective programs leading to sets of Pareto-optimal solutions (management plans), rather than to a pair of unique solutions. A management plan is Pareto-optimal if none of the associated environmental outputs (native forest restoration and nitrogen emissions) or revenues can be improved without compromising another output.

The two multi-objective programs were solved using the Alpha-Delta Algorithm (Tóth and McDill, 2009).The algorithm starts by finding the optimal solution, i.e., the set of optimal prescrip-

tions for each management unit, with respect to each of the three objective functions, one at a time without regard to another objective. Pareto-efficient solutions, solutions where none of the associated objective function values can be improved without compromising another objective value, were found iteratively using a slightly sloped composite objective function. For further algorithmic details, see (Tóth and McDill, 2009).

After finding the set of Pareto-efficient solutions for both Model E^+ and E^-, I performed sensitivity analyses on the minimum VDU size and nitrogen emissions to further explore conversion options. I set the minimum VDU size to 50, 80 and 110 ha, where 50 ha represented the size of the smallest, and 80 ha represented the size of the average dairy farm in New Zealand. Two nitrogen emissions rates for dairies were tested, 30 vs. 50 kg/ha per year, resulting in a total of six scenarios. I assumed that dairy revenues were independent of the operational changes required to reduce emission from 50 to 30 kg/ha per year.

2.3.5 Results and Discussion

The results of the modeling experiment suggest that the current combination of deforestation taxes and nitrogen emission regulations adopted in the Waikato region has been effective in preventing further increases in the emissions in the Tauranga-Taupo catchment. Mapped sets of Pareto optimal solutions, both in the presence and absence of environmental regulations, demonstrate tradeoffs between the costs of providing ecosystem services in different combinations (Fig. 2.7 and 2.8). In the absence of regulations, the values of the function that represent nitrogen emissions were up to 11.4 times greater than under the current regulatory environment. Dairy conversions were the choice of the profit-maximizing landowner on an overwhelming 75% of the area of the plantation (Fig. 2.7). An increase in nitrogen emissions following such a conversion could initiate irreversible degradation processes (Young et al., 2005). Note that points 29 – 39 on the frontier in Fig. 2.7 correspond to scenarios where the entire area of the Tauranga-Taupo catchment is restored to native forest; therefore, nitrogen emissions are constant and this part of the frontier is two dimensional, as the entire plot of the solutions of model E^+ is (Fig. 2.8).

An unexpected finding in this study was that dairy conversions were not possible in the Tauranga-Taupo catchment under model E^+. A combination of factors led to this result. First, no more than

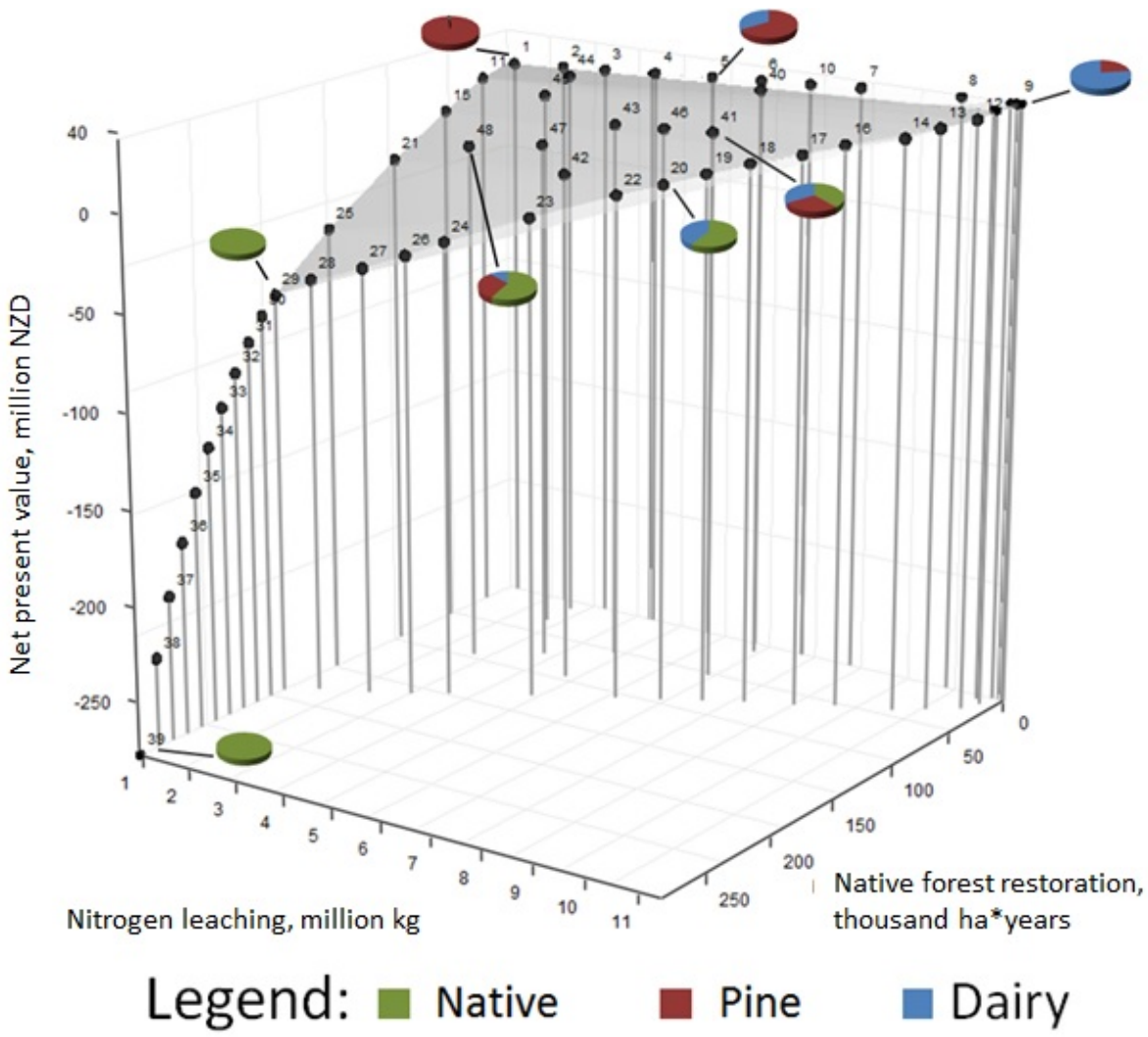


Figure 2.7: Pareto optimal solutions to model E^{-}.

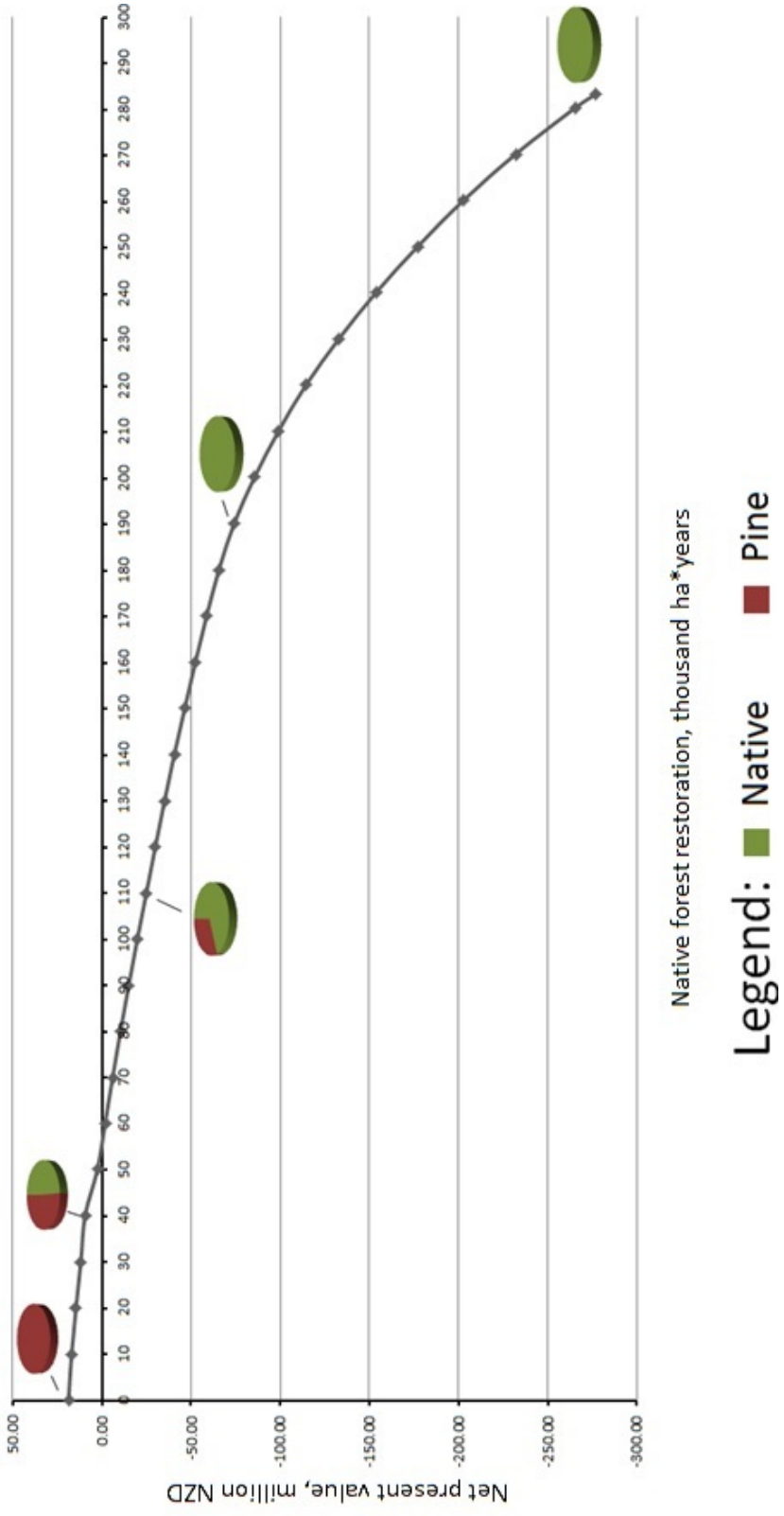


Figure 2.8: Pareto optimal solutions to model E^+.

35 ha of pine plantation can be converted from any one block in the catchment, given the relatively small block size (less than 830 ha). Since average nitrogen emissions cannot exceed 5 kg/ha per year per block to comply with WRC regulations, the area of forestland that can be converted to dairy in any one block is small relative to the 110 ha minimum VDU size. Additionally, management units greater than 35 ha are not eligible for conversion, as partial conversions were not considered for economic and silvicultural reasons. Finally, many of the management units in the catchment are spatially disjoint, making it difficult to construct large enough clusters, even across block boundaries. Nitrogen emissions were assumed to be the same in both plantations and native forests; thus, the value of the corresponding objective function (Eq. 2.3) was constant in model E^+. In model E^-, however, tradeoffs exist among revenues, nitrogen emissions and native forest restoration (Fig. 2.7), precisely because dairy conversions are feasible. Overall, the results imply that the nitrogen emissions regulations and deforestation taxes make a significant difference in land-use patterns. If these restrictions were not in place, most of the catchment would be converted to dairy and landowner revenues would be 51% greater (Fig. 2.7).

As far as native forest restoration is concerned, the findings suggest that this would be an expensive endeavor, costing about NZ$75 million to convert the entire plantation over the 50-year long planning horizon (Fig. 2.7 and 2.8). The price tag is much steeper, nearing NZ$270 million in net present value (Fig. 2.7 and 2.8), if restoration were to take place as early in the horizon as possible. The question, of course, is whether there is enough willingness to pay for native forest restoration services.

The results of the sensitivity analyses imply that dairy conversions would be possible under current regulations if the minimum VDU size were in the range of 50–60 ha and if nitrogen emissions could be lowered to 30 kg/ha per year. The optimal land-use management plan derived by solving model E^+ for this scenario suggests that as much as 169.67 ha of radiata pine plantation could be converted to dairy, forming three separate units of about 56 ha. Revenues, however, would be lower (2.7% less) than what would be possible if no conversions were to take place, because the deforestation tax exceeded the value of the resulting dairy revenue.

These specific results cannot be generalized to areas beyond the Tauranga-Taupo catchment. Differences in the size and the spatial configuration of the land-use blocks and the forest management units can lead to different results. Nonetheless, the proposed spatial model can be applied to other

locations.

2.4 Conclusion

In this chapter, I introduced a general approach to evaluate the effects of environmental regulations and other policy mechanisms on forest ecosystem services in the presence of economic drivers. Mathematical programs are excellent tools for this task, because the provision of ecosystem services can be modeled as objective functions while the regulations can be captured with constraints and the management decisions with variables. I illustrated the approach with a case study in New Zealand, where it was unclear how effective current regulations were in preserving water quality and whether complementary market mechanisms were needed. Using mathematical programming, I showed that the current regulations are effective in the particular area, and additional instruments, e.g. market, might be helpful only if the provision of other ecosystem services, such as native forest restoration, was desired.

The general approach can be applied for the analysis of other regulations and/or certification standards alone or in combination. For example, this method can be applied to analyze whether and how the Clean Water Act (Federal Water Pollution Control Amendments of 1972) affects ecosystem services other than water quality control. Model constraints would limit the total maximum daily load of each pollutant, and the objective functions would represent the provision of other ecosystem services. Similarly, key differences in the criteria of forest certification programs, such as Forest Stewardship Council (2010) and Sustainable Forestry Initiative (2010), can be incorporated into the models to analyze to what extent the two standards promote or compromise the provision of ecosystem services. Knowing the effects of regulations and certification standards on ecosystem services, and land management in general, would help forest managers plan operations that comply with the regulations and certification criteria while still maximizing revenue and the provision of ecosystem services.

Chapter 3

OPTIMAL FUEL TREATMENT ALLOCATION FOR REDUCING WILDFIRE HAZARD

3.1 Introduction

Forest wildfires pose a substantial threat to forest ecosystems and exurban buildings (Martell, 2007); forest managers plan fuel treatments to reduce both fire risk and fire hazard (defined in the Glossary; Glossary of Forest Fire Management Terms (2003); Glossary of Wildland Fire Terminology (2012)). Extensive modeling efforts have resulted in a number of methods for allocating fuel treatments to reduce the risk of large fires, but none of them have explored fire *hazard* reduction at the landscape scale as a result of the treatments. To address this question, I propose and test optimization models for allocating fuel treatments, to identify the model that mitigates fire hazard best.

The primary goal of fire management is to avert significant threats to public safety and resources caused by wildfires by reducing fire hazard (Forests and Rangelands, 2012). To achieve this goal, managers design treatment strategies that reduce fire hazard to ensure effective fire containment with minimal risk to firefighters. Fire hazard is an important characteristic used in planning firefighting efforts (Hardy, 2005). Crowning index (Fiedler et al., 2003) and the potential of crown fire (Moghaddas et al., 2010) have been used as proxies for fire hazard, but such definitions might not reflect the intended property—that fire managers base their strategies on (Andrews and Rothermel, 1982).

Extensive literature has explored treatment effectiveness at the stand level. Many studies report that fire hazard is reduced at the stand level following treatments (Agee and Skinner, 2005; Christensen et al., 2002; Fernandes and Botelho, 2003; Fiedler et al., 2003; Kalabokidis and Omi, 1998; Moghaddas et al., 2010; Pollet and Omi, 2002; Reinhardt et al., 2010; Schmidt et al., 2008; Skinner et al., 2004; Wilson and Baker, 1998), but is it still an open question how treatment placement affects fire hazard at the landscape scale (Moghaddas et al., 2010). Wildfires usually burn areas that exceed the average size of treatment units, so strategies for fire hazard mitigation must

plan treatments across the landscape. Fuel removal from the entire landscape might be infeasible because of economic, topographic, silvicultural, or conservation reasons; thus, the treatments must be strategically placed. Placement of fuel treatments subject to constraints on the total area and location can be effectively achieved using an optimization technique such as mathematical programming (Finney, 2007).

Optimization based methods have been used to identify patterns of fuel treatments that reduce the spread of large wildfires. Finney (2007) described an algorithm which iteratively identifies the areas where treatments most effectively retard fire growth. The algorithm is used in the Treatment Optimization Model (TOM) which allocates treatments in the landscape to achieve desired landscape conditions as input by the user (Finney, 2006). Finney et al. (2007) analyzed the effects of random vs. optimal fuel treatment allocations on fire growth and burn probability at the landscape level in the short (Finney, 2007) and long term (Finney et al., 2007) and reported overlapping patterns to be more effective at reducing fire spread rate compared to a random allocation. Kim et al. (2009) provided approaches for coordinating both operational and fuel reduction goals in a large landscape and reported that the effects of different treatment patterns (dispersed, clumped, random, and regular) on simulated wildfires did not vary significantly.

A number of optimization models have been proposed to address different management goals related to fire hazard mitigation. Calkin et al. (2005) optimized fuel treatments to reduce fire threat, defined as a combination of flame length, crowning and torching indices. Jones et al. (1999) combined simulation and optimization to define fuel treatment scenarios that minimize total area treated and the risk of undesired natural processes, such as fire or insect infestation. A few studies described optimization models that minimize the potential loss of public and private property in case of fire (Chung and Jones, 2011; Omi et al., 1981; Rytwinski and Crowe, 2010; Wei et al., 2008). A few revenue maximization models were proposed that accounted for fire hazard or fire risk (Daugherty and Fried, 2007; Konoshima et al., 2010).

The common shortcomings of these studies are that they lack a clear definition of fire hazard, and they do not analyze how fuel treatment allocation affects fire hazard reduction at the landscape level. In this chapter, I build and compare eleven single-objective optimization models for allocating of fuel treatments in a fire-prone landscape. I define a fire hazard measure that uses those fire characteristics that directly correspond to fire's resistance to control. Proxy measures defined

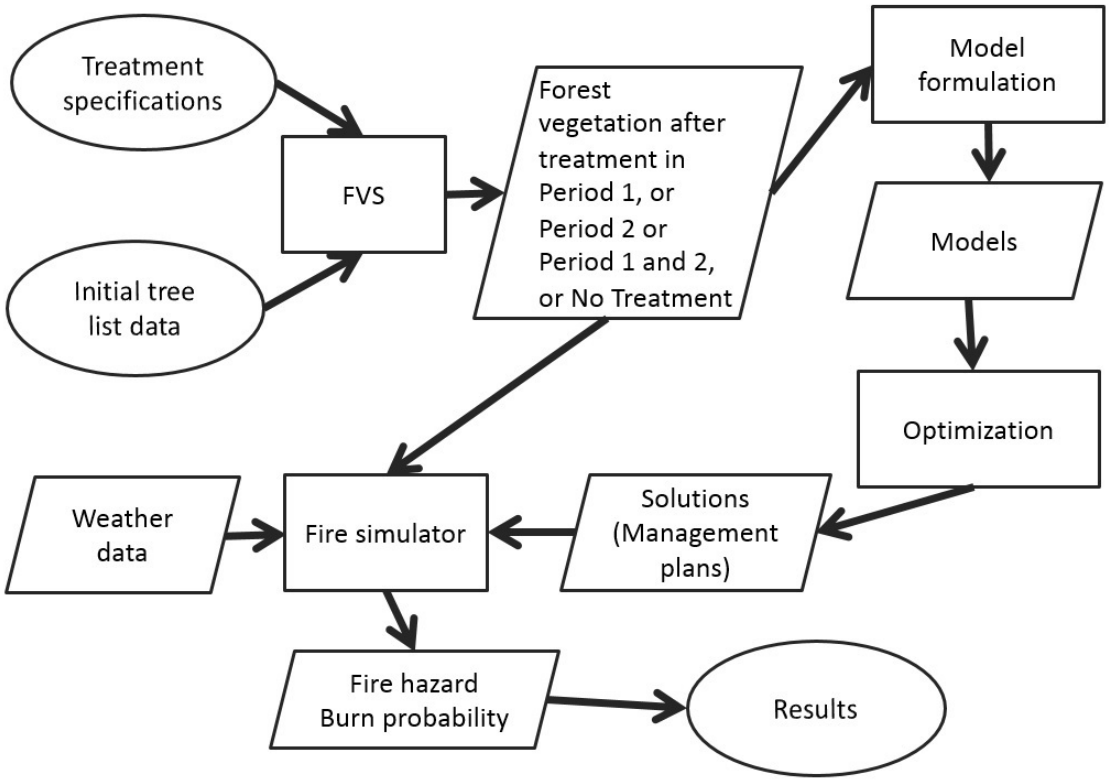


Figure 3.1: The schema of the approach to analyze the effects of fuel treatment allocations.

for fire hazard provide quantitative evaluations across the landscape, making them suitable for optimization models. Using vegetation projections from an area in Central Oregon, I analyze fire hazard changes on the landscape as the result of different spatially and temporally explicit fuel treatment assignments. The goal of the study is to find a computationally tractable model that effectively reduces landscape-wide fire hazard.

3.2 Methods

The general approach for the analysis of the effects of spatial treatment allocations (Fig. 3.1) includes several steps. The data for each of these steps and the processes are detailed in the sections following the study area decription.

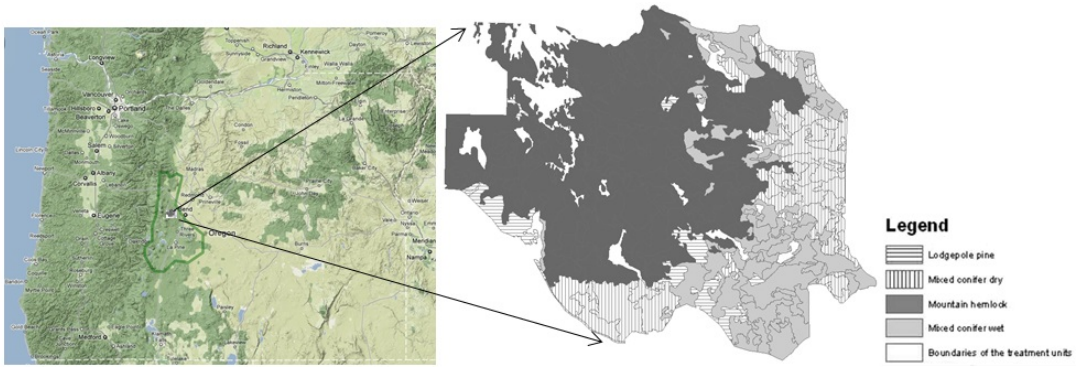


Figure 3.2: Drink Planning Area and plant association groups in the area.

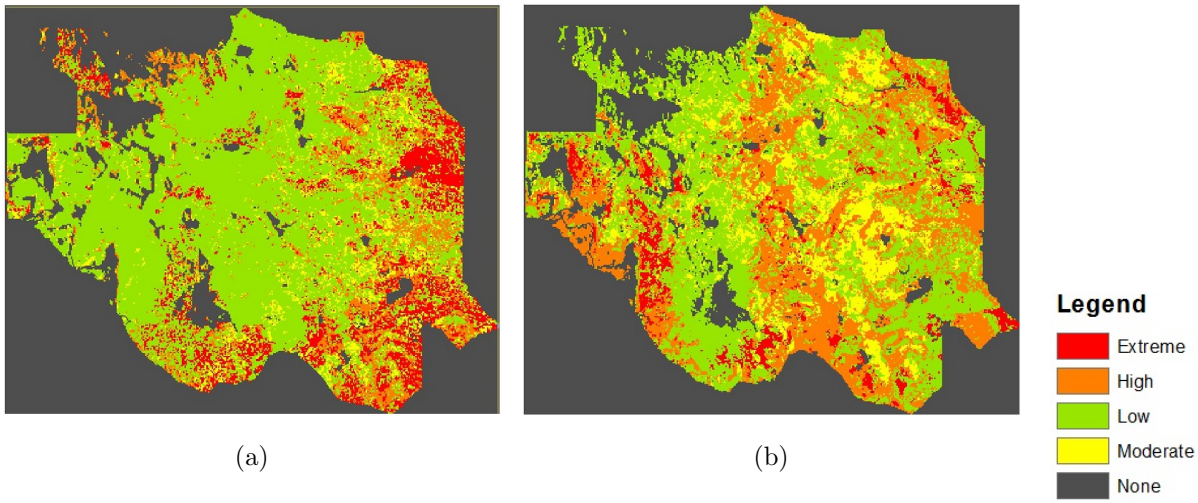
3.2.1 Study area

The data for this study are from the Drink Planning Area, a 7,056 ha area located on the eastern slopes of the Cascade Range in the Deschutes National Forest (Fig. 3.2). One third of the Drink area is the municipal watershed for the city of Bend, OR, so high severity fire poses a significant threat to the city's water supply. Wildfires can increase soil's hydrophobicity, causing surface and mass erosion, and leading to a significant sedimentation, debris flow and potential channel reorganization (Meyer et al., 2001; Rieman and Clayton, 1997; Wondzella and King, 2003). Therefore, protection of this area from fire is a high priority objective for the US Forest Service.

The study area is dominated by mature stands with high fuel accumulation and ladder fuels; so the fire hazard is high (Fig. 3.3a). Projections showed that, with time, fire hazard would increase in most of the area without fuel treatments (Fig. 3.3b); however, fire hazard would decrease on the east side of the Drink area, because of canopy density decrease and changes in forest structure. The US Forest Service is interested in treating the fuels in the area, so that fire is more manageable not if, but when, it occurs.

3.2.2 Data

The study area was divided into treatment units, each of which was associated with one or more tree lists that contain inventory data. Using a division into units based on tree lists alone would



(a) (b)

Figure 3.3: Fire hazard in the Drink area: initial in 2011 (a) and projected in 2051 (b).

not be beneficial, because often contiguous groups of the same tree list were too small to treat each one as a single unit. The Forest Service provided tree list data, the imputed inventory data for the study area (LEMMA; Landscape Ecology, Modeling, Mapping, and Analysis (2013)). Estimates for the initial values of the surface fuels of different size classes was obtained from fuel model data (Yanez, 2013).

Treatments specified for the Drink area aim to reduce crown bulk density and remove surface fuels. However, because the Drink area is roadless, the fuel treatments must be performed manually, and only trees smaller than 18 cm (for a few stands 30.5 cm) can be cut. (Treatment specifications are given in Appendix A.) For each unit, there was only one treatment defined, based on its vegetation characteristics and plant association group. Thus, the only decision regarding a given unit was whether to treat it in a chosen period, or not.

I used the Forest Vegetation Simulator (FVS; Dixon, 2013) SORNEC variant (C.E. Keyser, 2012) and its Fire and Fuel Extension (FFE; S.A. Rebain, 2010) to obtain projections of vegetation growth and fire characteristics given different treatment scenarios. FVS variants account for local peculiarities of vegetation and fuel types in different forests throughout the US; thus, these models are widely used to project forest characteristics. This software simulates vegetation and fuel parameters in cycle-by-cycle lists, where each cycle represents a period of time for which increments

of tree characteristics are predicted.

FVS has its limitations, and I followed the advice of area specialists by making two modifications to obtain more realistic output for the Drink area. The decay rates in the simulations were adjusted using data provided by a forester from the Forest Management Service Center, USDA-FS (Rebain, 2012). Canopy closure was adjusted to approximate recent LiDAR data for the area (Yanez, 2013). An adjustment factor was calculated from the average canopy cover for each unit and tree list by normalizing the unadjusted canopy closure taken from the FVS projections against the LiDAR data. Finally, crown base height projections were decreased by a factor of two (Yanez, 2013).

I considered a 40-year planning horizon, divided into two 20-year planning periods. This period length was chosen according to practical and theoretical assumptions regarding the longevity of the fuel treatments and vegetation growth in the study area (Agee and Skinner, 2005; Fernandes and Botelho, 2003). Vegetation growth is slow, thus, projections for shorter planning periods would not differ significantly in terms of vegetation and fuel characteristics.

Vegetation and fire parameters accounted for regeneration in the area following treatment. Simulations were run four times to obtain projections under four scenarios: "No treatment", "Treatment in the first period", "Treatment in the second period" and "Treatment in both periods", using the corresponding treatment specifications for each unit. The output was used in building the models and evaluating their performance.

3.2.3 Fire hazard metric

Before I provide the model formulations, I introduce the fire hazard metric used in the study. The metric is similar to the one for wildfire risk presented in Vaillant and Ager (2013), with crown fire potential substituted for burn probability. Fire hazard was defined as a combination of flame length and crown fire potential (Table 3.1). Flame length is a key measurement used in practice for planning the equipment for fire containment (Andrews and Rothermel, 1982). Crown fire potential quantifies the probability of crown fire, the containment of which requires extreme measures involving aerial water bombing and retardant chemicals. Both of these characteristics, flame length and crown fire potential, relate directly to fire's resistance to control.

Table 3.1: Fire hazard metric

	Flame length, ft			
Crown fire potential	<4	4 - 8	8 - 11	>11
1 = surface fire	Low	Moderate	High	Extreme
2 = passive crown fire	Moderate	High	Extreme	Extreme
3 = active crown fire	High	Extreme	Extreme	Extreme

3.2.4 Models

In this study, I compared two types of mixed-integer programs for fire hazard minimization: spatial and non-spatial. This distinction allows analysis of the importance of the spatial location of treatments. The variables in the models represent decisions about whether or not to assign a particular treatment to a certain unit in either or both planning periods. Therefore, each solution represents a spatio-temporal allocation of treatments in the landscape.

Non-spatial models

Non-spatial models allocate treatments to improve fire behavior parameters in the treated areas but do not account for their location. The proxies quantify the changes in the fire hazard, a qualitative characteristic, depending on different treatment assignments.

The general mathematical formulation of the non-spatial models is as follows:

$$\max \sum_{i \in I \setminus U, r \in R \setminus 0} C_{ir} x_{ir} \tag{3.1}$$

subject to:

$$\sum_{i \in I \setminus U} a_i (x_{ir} + x_{i12}) = H_r \quad \text{for all } r \in \{1, 2\} \tag{3.2}$$

$$H_r \leq A \quad \text{for all } r \in \{1, 2\} \tag{3.3}$$

$$l H_1 - H_2 \leq 0 \tag{3.4}$$

$$-u H_1 + H_2 \leq 0 \tag{3.5}$$

Table 3.2: Summary of the objective functions in the proposed optimization models

Model	Objective function description
	Non-spatial models
NSA	Maximize fuel model reduction on the landscape.
NSB	Maximize the reduction of fuel model rating.
NSC	Minimize a combined coefficient that includes flame length and crown fire potential.
	Spatial models
S1	Maximize spread of treated units (dispersion) across the landscape and avoid clustering.
S2	Maximize dispersion of the treated units while accounting for the prevailing west wind direction.
S3	Maximize dispersion of the treated units while prioritizing creation of contiguous barriers perpendicular to the prevailing wind direction.
S4	Maximize dispersion of the treated units using an area restriction model approach.
	Combined models
S1B, S2B, S3B, S4B	These models are combinations of the corresponding spatial model and model NSB.

$$\sum_{r \in R} x_{ir} = 1 \quad \text{for all } i \in I \setminus U \tag{3.6}$$

$$x_{ir} = 1 \quad \text{for all } i \in U, r \in R \setminus \{0\} \tag{3.7}$$

$$x_{i0} = 0 \quad \text{for all } i \in U \tag{3.8}$$

$$x_{ir} \in \{0, 1\} \quad \text{for all } i \in I, r \in R \tag{3.9}$$

where the variables are

x_{ir} = one if management unit i is assigned a prescription r, and zero otherwise;

H_r = accounting variables for total area treated in periods 1 and 2, respectively.

The model parameters are the following:

I = the set of treatment units (indexed by i);

U = the set of treatment units that are non-forested;

R = {1, 2, 12, 0}, indicating treatment in period 1, 2, both or none, respectively; indexed by r;

a_i = the area of unit i;

A = the maximum total area that can be treated in each planning period (MAT);

l, u = the lower and upper bounds on area fluctuations treated in periods 1 and 2;

C_{ir} = fire hazard reduction coefficient, particular to each model.

Function (3.1) minimizes fire hazard. Constraints (3.2–3.3) restrict total area treated in each period to lie within the recommended range of 30%–40% of the total area (Finney, 2007). Constraints (3.4) and (3.5) are balance constraints, ensuring that the treated area does not increase (decrease) by more than a factor of u (l) compared to the previous period. Constraint (3.6) ensures that only one prescription is assigned to a given management unit in the planning horizon. Equations (3.7) and (3.8) ensure that non-forested units are considered as treated during the planning horizon. Finally, the last constraint set (3.9) defines variables x_{ir} as binary.

Model NSA

Model NSA maximizes fuel model reduction, weighted by area, across the landscape. In this model, fuel model reduction is used as the proxy for fire hazard minimization. Fuel models described by Anderson (1982) can be used to identify areas where fire hazard is the highest. Reducing the fuel model, from 10 to 8 for example, improves fire behavior parameters: in this case flame length and rate of spread will decrease (Table 3.3). Therefore, a decrease in the fuel model is expected to

Table 3.3: Fuel model examples

Fuel model	Group	Flame length, ft	Rate of spread, chains/hour	Total fuel load, tons/ac
5	Shrub	4	18	3.5
8	Timber	1	1.6	5
10	Timber	4.8	7.9	12
12	Logging Slash	3.5	6	34.6
13	Logging Slash	10.5	13.5	58

reflect a positive dynamic in fire hazard reduction. Thus, in model NSA:

$$C_{ir} = FM_{ir} \tag{3.10}$$

where FM_{ir} is the fuel model reduction in unit i at the end of the planning horizon after prescription r. I weighted fuel model for each unit by area and, for each treatment scenario, calculated a coefficient representing its reduction compared to the No Treatment scenario.

Model NSB

Similar to model NSA, model NSB accounts for the changes in the fire behavior after treatments; this model maximizes the reduction of the fuel model rating across the landscape. A reduction in fuel model, however, does not always imply an improvement in fire behavior; specifically, fuel models with a low index are associated with high flame length potentials (Table 3.3). From a fire control perspective, such conditions would not be satisfactory, because fires would be difficult to contain. To account for this fact, I assigned each fuel model a rating based on flame length potential, rate of spread, and associated fuel loads. Fuel models with high values of these parameters had a poorer (i.e., higher) rating. Model NSB is similar to model NSA, but measures the reduction of fuel model rating instead of fuel model:

$$C_{ir} = FR_{ir} \tag{3.11}$$

where FR_{ir} is the reduction of fuel model rating in unit i at the end of the planning horizon after prescription r.

Model NSC

Model NSC uses flame length and crown fire potential directly as a metric for fire hazard reduction. The proxy for fire hazard reduction in model NSC is a combination of flame length and crown fire type (1=surface, 2=passive, 3=conditional, 4=active):

$$C_{ir} = Fl_{ir} Ft_{ir} \tag{3.12}$$

where Fl_{ir} is flame length, and Ft_{ir} is the crown fire type in unit i at the end of the planning period after prescription r was applied. Both parameters should be minimized because, by definition, fire hazard is lowest when both flame length and crown fire potential have low values (Table 3.1).

Spatial models

The four spatial models assign treatments to units on the landscape in dispersed patterns. Dispersion is defined as the sum of the physical distances between treated units, as in (Kim et al., 2009). Finney (2007) showed that regularly dispersed treatments, where slow burning units are interspersed with fast burning ones, minimize fire growth. Both flame length and crown fire potential are correlated with spread rate; thus, treatments in such a pattern might reduce fire hazard at the landscape level.

The mathematical formulation of models S1–S3 is the following:

$$\max \sum_{r \in R \setminus 0} \sum_{i \in I} \sum_{j \in I \setminus i} D_{ij} z_{ijr}, \quad \text{for all } i, j \notin U \tag{3.13}$$

subject to:

$$z_{ijr} - x_{ir} - x_{i12} + z_{ij12} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i, r \in \{1, 2\} \tag{3.14}$$

$$z_{ij12} - x_{i12} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i \tag{3.15}$$

$$z_{ijr} - x_{jr} - x_{j12} + z_{ij12} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i, r \in \{1, 2\} \tag{3.16}$$

$$z_{ij12} - x_{j12} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i \tag{3.17}$$

$$x_{ir_i} + x_{jr_j} - z_{ijr} \leq 1 \tag{3.18}$$

$$\text{for all } i \in I, j \in I, i, j \notin U, j > i, r_i, r_j, r \in R \setminus \{0\}, r = \min\{r_i, r_j\}, |r_i - r_j| \neq 1$$

$$x_{ir} = 1 \quad \text{for all } i \in U, r \in R \setminus \{0\} \tag{3.19}$$

$$x_{i0} = 0 \quad \text{for all } i \in U \tag{3.20}$$

$$\sum_{i \in I \setminus U} a_i (x_{ir} + x_{i12}) \leq A \quad \text{for all } r \in \{1, 2\} \tag{3.21}$$

$$\sum_{r \in R} x_{ir} = 1 \quad \text{for all } i \in I \setminus U \tag{3.22}$$

$$x_{ir} \in \{0, 1\}, z_{ijr} \in \{0, 1\} \tag{3.23}$$

Variable x_{ir}, parameters a_i, A, and sets I, P, U, R are defined as in the non-spatial models. z_{ijr} = one if both units i and j are treated in the same period, and zero otherwise;

r_i = treatment prescription for unit i;

D_{ij} = the distance coefficient for management units i and j, particular to each model.

Function (3.13) maximizes the dispersion of management units treated in the same planning period. Constraints (3.14–3.18), adapted from Tóth et al. (2011), work together to avoid nonlinearity, to ensure that distances are computed only between units treated in the same period and that a given distance is counted only once. Constraints (3.14–3.17) allow and constraints (3.18) ensure that z_{ijr} is one if both x_{ir_i} and x_{jr_j} are equal to one. If either of the units is non-forested, the corresponding constraints (3.14–3.18) can be omitted: the decision variables for non-forested units are always equal to one and, therefore, these constraints do not affect variables z_{ijr}. Constraints (3.19) and (3.20) require all non-forested units to be accounted for as treated during the planning horizon. Area constraints (3.21) do not allow the total area treated to exceed a given limit. Equations (3.22) ensure that there is only one prescription assigned to each unit in the planning horizon. Finally, constraints (3.23) define variables x_{ir} and z_{ijr} as binary.

In model S1, the distance coefficients are defined using distances between the units' centroids (Fig. 3.4):

$$D_{ij} = \begin{cases} d^c_{ij} & \text{if } i \text{ and } j \text{ are not adjacent} \\ -M & \text{otherwise,} \end{cases} \tag{3.24}$$

where d^c_{ij} is the distance between the centroids of units i and j, and M is a constant ($M > \max_{i,j}(d^c_{ij})$). The distances between units' centroids are positive even if the units are adjacent; and if the units are large, the distance might be larger than between smaller non-adjacent units.

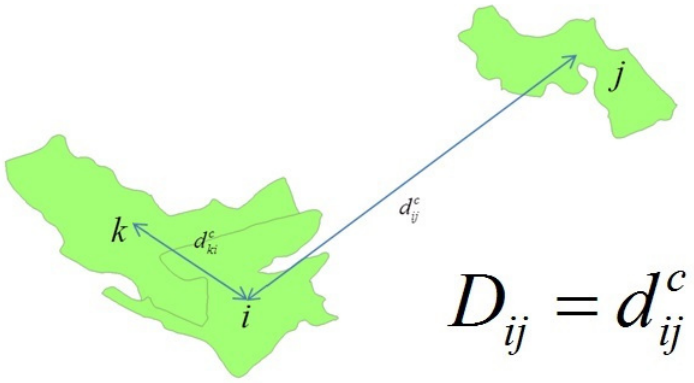


Figure 3.4: Definition of distance coefficients in model S1.

Adjacent units, however, were not considered to be dispersed. To avoid clumping, the distance coefficients between adjacent units were assigned large negative values.

Model S2 maximizes the spread of treated units along the prevailing wind direction. The coefficients in this model are the east-west components of the distances between the units' centroids (Fig. 3.5):

$$D_{ij} = \begin{cases} d^c_{ij} \cos \alpha^c_{ij} & \text{if units } i \text{ and } j \text{ are not adjacent} \\ -M & \text{otherwise,} \end{cases} \tag{3.25}$$

where α^c_{ij} = the angle between the line that connects the centroids of units i and j and the horizontal. Parameters d^c_{ij} and M are as defined above.

In model S3, the units to the north or south of a treated one are prioritized for treatment because together they form contiguous barriers perpendicular to the prevailing westerly wind (Fig. 3.6). In this model, a unit has a greater distance coefficient if (1) it is located to the north or south relative to a treated unit or (2) it is farther from a treated unit. The sets of units that could form barriers were identified up front. The distance coefficients in model S3 are as follows:

$$D_{ij} = \begin{cases} d^c_{ij} & \text{if units } i \text{ and } j \text{ share common boundary on} \\ & \text{the north or south or are not adjacent} \\ -M & \text{otherwise,} \end{cases} \tag{3.26}$$

where d^c_{ij} and M as defined above.

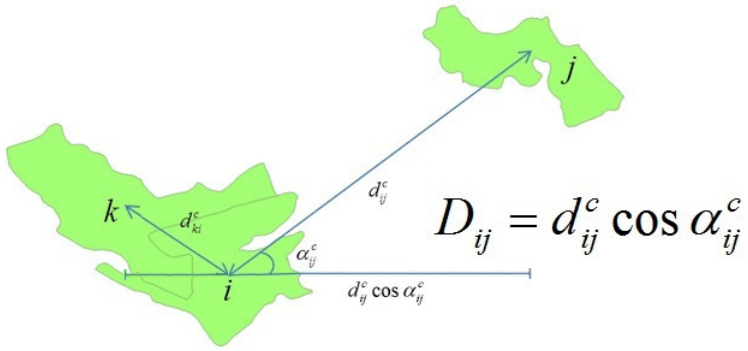


Figure 3.5: Definition of distance coefficients in model S2.

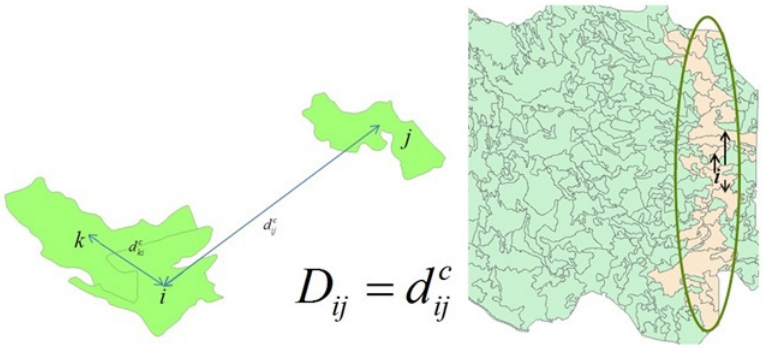


Figure 3.6: Definition of distance coefficients in model S3.

Models S1B–S3B were obtained from models S1–S3 by adding fuel model rating to the coefficients in the objective functions. This addition necessitated the construction of different decision variables and, consequently, different constraints to avoid non-linearity. Constraints (3.28–3.30) differ from the corresponding constraints (3.14–3.18) in models S1–S3 in that they account not only for all possible combinations of treatments but also for changes in fuel model ratings depending on the treatment prescriptions in each unit. The general formulation for models S1B–S3B is the following:

$$\max \sum_{r_i, r_j \in R \setminus 0} \sum_{i \in I} \sum_{j \in I \setminus i} D^f_{ijr_ir_j} y_{ijr_ir_j} \quad \text{for all } |r_i - r_j| \neq 1, i, j \notin U \tag{3.27}$$

subject to:

$$y_{ijr_ir_j} - x_{ir_i} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i, r_i, r_j \in R \setminus \{0\}, |r_i - r_j| \neq 1 \tag{3.28}$$

$$y_{ijr_ir_j} - x_{jr_j} \leq 0 \quad \text{for all } i \in I, j \in I, i, j \notin U, j > i, r_i, r_j \in R \setminus \{0\}, |r_i - r_j| \neq 1 \tag{3.29}$$

$$x_{ir_i} + x_{jr_j} - y_{ijr_ir_j} \leq 1 \tag{3.30}$$

$$\text{for all } i \in I, j \in I, i, j \notin U, j > i, r_i, r_j \in R \setminus \{0\}, |r_i - r_j| \neq 1$$

$$x_{ir} = 1 \quad \text{for all } i \in U, r \in R \setminus \{0\} \tag{3.31}$$

$$x_{i0} = 0 \quad \text{for all } i \in U \tag{3.32}$$

$$\sum_{i \in I \setminus U} a_i (x_{ir} + x_{i12}) \leq A \quad \text{for all } r \in \{1, 2\} \tag{3.33}$$

$$\sum_{r \in R} x_{ir} = 1 \quad \text{for all } i \in I \setminus U \tag{3.34}$$

$$x_{ir} \in \{0, 1\}, y_{ijr_ir_j} \in \{0, 1\} \quad \text{for all } i \in I, r \in R, r_i, r_j \in R \setminus \{0\} \tag{3.35}$$

Coefficients $D^f_{ijr_ir_j} = D_{ij}(FR_{ir_i} + FR_{jr_j})$, where FR_{ir_i} is defined as in model NSB. Constraints (3.28)–(3.30) are included to avoid non-linearity in the model: inequalities (3.28)–(3.29) allow, and inequalities (3.30) require that $y_{ijr_ir_j}$ is one if variables x_{jr_j} and x_{jr_j} are one. The other variables and constraints are as defined for models S1–S3.

For model S4, I used an area restriction model formulation (ARM; Constantino et al., 2008; Goycoolea et al., 2005; McDill et al., 2002; Murray, 1999) to achieve dispersion. ARMs are harvest

scheduling models with maximum clearcut size constraints. Previous research (Boston and Bettinger, 2006; Franklin and Forman, 1987) has shown that a small maximum clear cut size results in a higher dispersion of harvested units. By substituting maximum contiguous area of treated units (MCTA) for maximum clearcut size, this feature of ARM can be used to create a dispersed pattern of fuel treatment allocation, similar to model S1. Models S4(50), S4(100) and S4(200) were formulated with the MCTAs of 20.23, 40.46, and 80.93 ha, respectively: The model formulation for S4 is as follows:

$$\max \sum_{r \in R \setminus 0} \sum_{i \in I \setminus U} a_i x_{ir} \tag{3.36}$$

subject to:

$$\sum_{i \in I \setminus U} (a_i (x_{ir} + x_{ir}) = H_r \quad \text{for all } r \in \{1, 2\} \tag{3.37}$$

$$H_r \leq A \quad \text{for all } r \in \{1, 2\} \tag{3.38}$$

$$lH_1 - H_2 \leq 0 \tag{3.39}$$

$$-uH_1 + H_2 \leq 0 \tag{3.40}$$

$$\sum_{i \in C_j} x_{ir} \leq |C_j| - 1 \quad \text{for all } j \in J, r \in \{1, 2, 12\} \tag{3.41}$$

$$\sum_{r \in R} x_{ir} = 1 \quad \text{for all } i \in I \setminus U \tag{3.42}$$

$$x_{ir} = 1 \quad \text{for all } i \in U, r \in \{1, 2, 12\} \tag{3.43}$$

$$x_{i0} = 0 \quad \text{for all } i \in U \tag{3.44}$$

$$x_{ir} \in \{0, 1\} \quad \text{for all } i \in I, r \in R \tag{3.45}$$

where variables x_{ir}, H_r, parameters a_i, u, l, and sets I, U, R are defined the same as for the non-spatial models;

C_j = the set of units that comprise cluster j, a set of contiguous units with a combined area just exceeding the MCTA;

$|C_j|$ = the cardinality of set C_j;

J = the set of clusters.

Function (3.36) maximizes the total area of treated units in the planning horizon. Constraints (3.37)–(3.40) are included to balance the total area treated in the two periods. Constraints (3.38) restrict the area treated in each period. Constraints (3.39) and (3.40) ensure that the treated area does not increase (decrease) by a factor of u (l) compared to the previous period. Constraint (3.41), adapted from McDill et al. (2002), prevents the treatment of adjacent units if their combined area exceeds the MCTA. Logic constraint (3.42) ensures that only one prescription is assigned to a given unit in the planning horizon. Constraints (3.43)–(3.44) assign values to non-forested units to indicate that they are considered treated during the planning horizon. Constraints (3.45) ensure that the decision variables are binary.

For model S4B, the only change that needs to be made is the substitution of fuel model rating reduction FR_{ir} for area a_i (3.11).

The models were solved using CPLEX 12.4 on a Dell PE2950 with a 3GHz Intel 5160 dual-core CPU and 16 GB RAM.

3.2.5 Model evaluation

I compared the solutions of the models using FlamMap, a fire behavior simulator that has proven to be effective in projecting fire behavior on large landscapes (Ager et al., 2011; Finney, 2006). An advantage of this simulator is that it calculates fire characteristics using topography and fuel data on the landscape. In this study, the input for FlamMap incorporated information about the treatment allocation from the optimization models, allowing comparison of fire hazard reduction due to treatments in different patterns.

Weather conditions were defined to approximate weather during the fire season in the study area. The weather input for FlamMap (fuel moisture, weather and wind files) was created in FireFamily Plus 4 (USDA Forest Service, 2002) using RAWS data from years 2002 to 2012 (KCPFast, 2013) for the Tumalo Ridge station. According to the weather data, the predominant wind direction was from west to east. To evaluate the solution's robustness, I ran a few simulations with an easterly wind as well. FlamMap's azimuth parameter was set to 270 degrees to simulate a west wind, and to 90 degrees for an east wind. Wind speed in the simulations was set to 20 mph, the gust speed taken from the records, because gust speed is more likely to affect fire growth, intensity and crown

fire initiation than the average wind speed (Stratton, 2006). Foliar moisture content was set to 90%, as the literature recommends (Agee et al., 2002; Alexander, 2010; Keyes, 2006). FlamMap simulations were run for a few days, 8/12–8/17, chosen arbitrarily within the fire season.

The input for the burn probability simulation included a list of 2000 random ignitions, compiled in RandIg (Missoula Fire Sciences Laboratory, 2011), a software package used to create lists of random ignitions based on spatial, vegetation and weather input. The maximum simulation time per ignition was arbitrarily set to 4200 min per simulated ignition. A longer simulation time would give more accurate results, but would take longer to run (at 4200 minutes, the simulation for each model took more than 8 hours). The list of ignition locations was the same for each treatment allocation.

FlamMap output was used to build fire hazard and burn probability maps to compare the performance of the optimization models. Fire hazard maps were produced by combining flame length and crown fire potential on a per-pixel basis using the fire hazard classes defined in Table 3.1. In addition to the graphical information, I obtained summary statistics for the area in each fire hazard class, calculated as a percentage of the total area.

In addition to fire hazard and burn probability, I recorded solution times and optimality gaps for the models. Optimality gap characterizes the quality of the solutions: the lower the gap, the better the solution. These data allow the selection of models based on time to solution and their optimality. Some of the models were computationally hard and could not be solved to optimality; in these cases I interrupted the process after 14 hours and recorded the optimality gap achieved at that point.

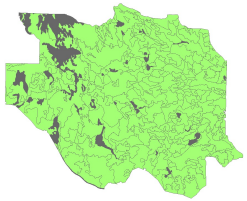
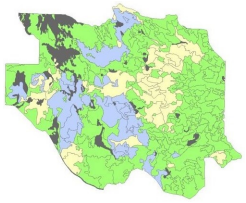
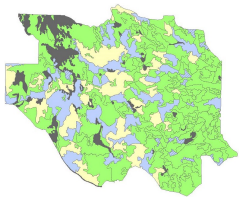
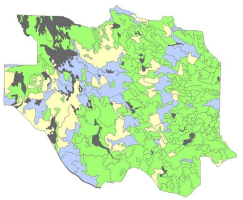
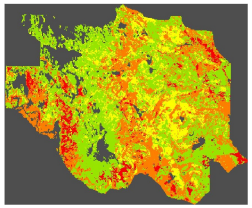
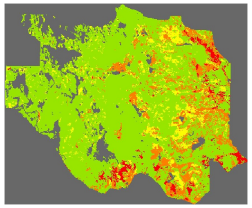
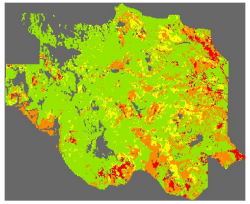
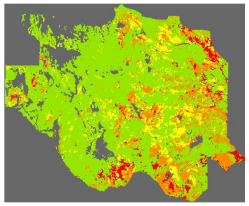
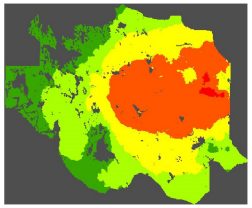
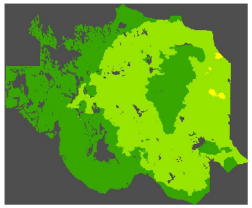
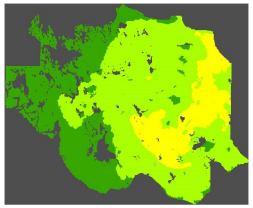
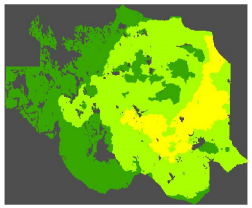
3.3 Results

The models produced different treatment patterns (Table 3.4)[1]. The spatial models produced patterns where treatments were dispersed (S1, S1B, S2, S2B) or aggregated into barriers perpendicular to the prevailing wind direction (S3, S3B). The non-spatial models, as well as spatial models S4 and S4B, allocated treatments in a clumped pattern.

Fire hazard was most effectively reduced in models NSB, S3B, S1B, and S4B(200) (Table 3.5).

[1]Results produced by the other models can be found in Appendix B.

Table 3.4: Comparative analysis of the models for fuel treatment allocation

Parameter	No treatment	NSB	S1B	S3B
Fuel treatment allocation Treated in both periods Treated in period 1 Treated in period 2 Treatment units Non-forested				
Fire hazard Extreme High Low Moderate None				
Burn probability 0 0 - 0.1 0.1 - 0.2 0.2 - 0.3 0.3 - 0.4 0.4 - 0.5 0.5 - 0.6				

Model NSC, with a proxy of flame length and crown fire potential, showed worse results than model NSB. However, model NSC outperformed NSA, which had the most area in the extreme fire hazard class. The area in high and moderate fire hazard classes was similar for all the non-spatial models.

Spatial models S1B–S4B outperformed models S1–S4: the area in the extreme fire hazard class was higher for the latter models. All the spatial models were able to reduce the area in high and moderate fire hazard classes compared to the No Treatment scenario. Models S1B–S4B were able to reduce the area in the extreme fire hazard class compared to the No Treatment scenario, whereas models S1–S4 did not. Of the combined models, model S3B performed the best; and of all the models, S3B was second best.

The models that effectively reduced fire hazard also showed the best results in terms of burn probability (Table 3.4). The lowest fire hazard and burn probabilities were achieved in models NSB, S1B, S3B, and S4B(200).

The correlation between MAT increase and fire hazard reduction was the same for the spatial and non-spatial models, and showed that better fire hazard reduction is achieved with higher MAT values (Table 3.6). However, the area in the high and extreme fire hazard classes did not decrease in proportion with increases of MAT value. Specifically, area in the extreme fire hazard class decreased the least with the increase in MAT: the range of values was less than 0.72%. The greatest variation occured in the low and high fire hazard classes.

The reversed wind direction (west vs. east) did not affect the general conclusion about which models were most effective. Models that performed well under west wind conditions outperformed the others when the wind direction was reversed. Under east wind conditions, less area was classified as low fire hazard and more was classified as high and extreme fire hazard, in the solutions of all the models. A change in the wind direction affected the burn probability distribution more dramatically, however (Table 3.7). When the wind direction was changed to east, the area projected to burn with probability greater than 0.2 increased in all solutions. Model NSB was the only model that produced a solution where burn probability remained uniformly below 0.3 after the wind direction change.

In terms of solution times and optimality, the non-spatial models and models S4 and S4B performed the best: these models solved fast, in less than a minute (Table 3.8). The spatial and combined models, aside from models S4 and S4B, were computationally hard: their solution times

Table 3.5: Comparison of areas in different fire hazard classes as a result of different fuel treatment allocations. The MAT was 1,618.7 ha, except for models S4 and S4B, in which the largest area that could be treated in one period was 1,335.5 ha, with an MCTA of 80.93 ha.

	Area in fire hazard classes, % of the total area			
Model	Low	Moderate	High	Extreme
NSB	74.68	9.11	12.64	3.57
S3B	70.2	10.57	15.11	4.12
S1B	67.53	10.27	17.82	4.38
S2B	68.84	10.61	15.71	4.84
S4B (200)	65.43	10.22	18.65	5.7
S4B (100)	60.51	11.15	22.38	5.96
S4B (50)	58.46	11.56	23.66	6.32
No-Treat	45.98	16.3	30.98	6.74
S2	57.14	12.05	22.58	8.23
S1	61.21	11.53	18.35	8.92
S3	60.75	10.93	18.28	10.03
NSC	64.28	10.3	14.68	10.74
S4 (50)	55.16	12.12	21.42	11.3
S4 (100)	68.39	2.66	17.36	11.59
S4 (200)	63.9	9.97	13.26	12.87
NSA	63.96	9.04	13.3	13.7

Table 3.6: Effects of MAT on fire hazard class distribution (model NSB)

Area treated	Area in fire hazard classes, % of the total area			
limit, ha	Low	Moderate	High	Extreme
2,428.12	79.54	6.51	10.29	3.66
1,618.70	74.68	9.11	12.64	3.57
1,214.06	68.15	10.38	17.63	3.84
809.37	60.77	12.57	22.37	4.29

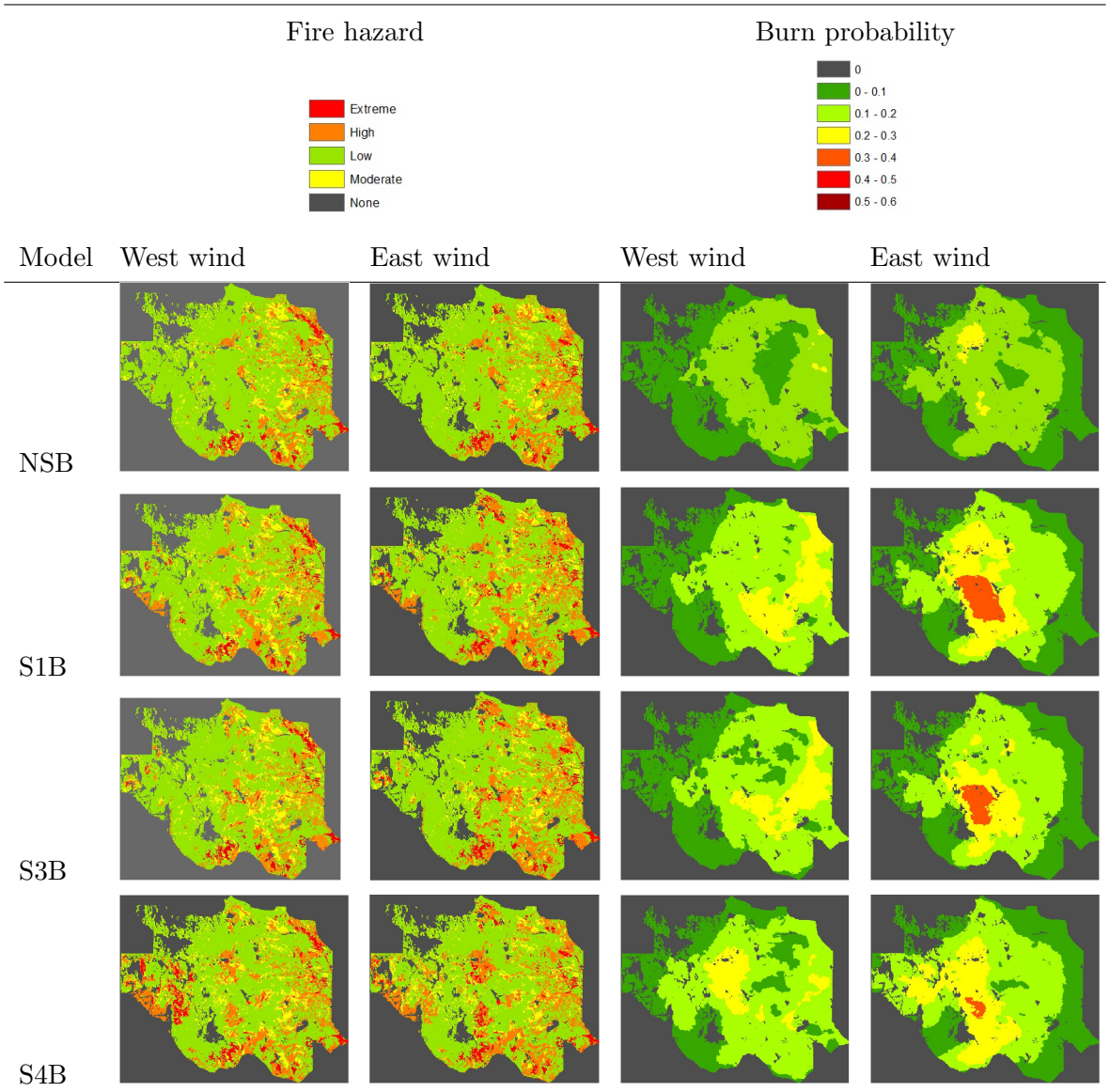
were over 24 hours. These models did not find optimal solutions and had their optimality gaps ranged from 1% to 30%.

3.4 Discussion

The key factor that affected fire hazard reduction was fuel removal. High fuel accumulation creates the potential for high intensity burning and could, consequently, result in crown fire (Bennett et al., 2010). Reduced fuel accumulation would prevent high intensity fires with high values of flame length. In this study, models S1–S4, which prioritized treatment allocation in distinct patterns, did not take fire behavior parameters into account; thus, they did not effectively decrease fuel accumulation. However, models S1B–S4B still did not perform as well as non-spatial model NSB, which produced the best results overall. The non-spatial models assigned treatments wherever treatments effectively mitigated fire behavior. The spatial models did not allow assignment of treatments based solely on fire behavior reduction, because the treatment pattern was prioritized in these models. Because of this peculiarity, the spatial models were less effective at fire hazard reduction: areas that were critical for treatment might have been left out to prioritize the treatment pattern. This implies that the spatial allocation of treatments is not as important as the reduction of fuels, both horizontal and vertical, which in this study was expressed as a reduction of fuel model rating.

Fuel model rating was a better proxy for fire hazard reduction than the standard fuel model. Models that incorporated fuel model ratings were able to identify and allocate treatments in those

Table 3.7: Burn probability distribution for selected models



(200)

Table 3.8: Solution times and optimality gaps for spatial and non-spatial models

Model	Solution time, sec	Optimality gap, %
NSA	0.55	0
NSB	0.23	0
NSC	0.09	0
S4 (50)	0.08	0
S4 (100)	0.11	0
S4 (200)	7.44	0
S1	52080.48*	0.79
S2	52218.69*	31.04
S3	52307.69*	4.5

* - The run was terminated at this point.

areas where fire hazard could be reduced the most, and avoided areas where treatments might exacerbate fire behavior.

The solutions of the models showed that area treated was an important factor in fire hazard reduction, which agrees with existing research. Collins et al. (2010) reported that consistent improvement of crown fire potential and reduction of modeled fire sizes could be achieved if at least 20% of the landscape area was treated. In this study, the spatial constraints in models S4, S4B, S1, and S1B caused a decrease in the total area treated in the corresponding solutions. In other models, MAT constraints were binding because higher values of treated area correlated with greater fuel reduction, which resulted in an improvement in fire hazard landscape-wide. The area in the extreme fire hazard class changed the least with area treated because most models selected the most critical areas for fire hazard reduction for treatment. The models chose units where the reduction was the greatest overall, which might leave untreated units with some area in the extreme fire hazard class.

None of the models that showed good results in fire hazard reduction allocated treatments

on the east side of the Drink area. Fire hazard remained high on the east side regardless of treatments, which can be explained by 1) a high flame length that cannot be reduced by the treatments specified for the area, and 2) a high crown fire potential that would be challenging to reduce in the mixed conifer plant association groups (Simpson, 2007). A similar result was reported by Halofsky et al. (forthcoming), where treatments reduced the area in the highest fire hazard classes, but not significantly, compared to the No Treatment scenario.

Because the treatments were restricted in terms of what trees can be removed, the fuel model and, consequently, fire hazard did not change significantly as a result of these treatments. Specifically, portions of the eastern Drink area have experienced wildfires in the past—Bridge Creek in 1979 and Swampy Lake in 1914—and the forest stands there are open. An open canopy allows more sunlight on the ground and, consequently, has created favorable conditions for a thick shrub/grass layer. The treatments defined for these areas would remove young trees, reduce canopy cover and support conditions for herbaceous vegetation and, consequently, increase the amount of small fuels produced by grasses and shrubs (Ager et al., 2011). Thus, the fuel model would change for the worse. The inclusion of additional treatments, such as mowing following thinning and slash removal, might help reduce small fuels, thus catastrophic flame length potential, and thus fire hazard.

Reversal of the wind direction affected the distribution of fire hazard classes in the Drink area. Fire progression driven by west winds has more catastrophic consequences, because it can reach watershed, NSO habitat, and populated areas. Winds from the east do not impose as great a threat as west winds, because these areas are located to the east of the study area. Moreover, the fire driven by east winds meets a natural barrier (mountain range) and is contained there. Therefore, for forest fire management in this area, it would be more useful to plan for fire that is driven by the west wind.

For burn probability, it was crucial that the areas of high and extreme fire hazard were broken up. The dispersion of these areas would prevent the rapid spread of fire on the landscape and reduce the chance of burning in units far from the ignition point. This result agrees with the findings in Finney (2007). Solutions where treatments isolated areas of high and extreme fire hazard classes showed the best reduction in burn probabilities (Table 3.7). Although models S1B and S3B were effective at reducing fire hazard, models NSB and S4B(200) broke up areas of high and extreme fire hazard and, therefore, achieved a greater reduction in burn probability.

In addition to effective fire hazard and burn probability reduction, model NSB solved quickly to an optimal solution. Although the spatial models reduced fire hazard almost as well as model NSB, they were significantly harder to solve because of the large number of decision variables which had to be added to capture the spatial components.

Vegetation and fire behavior projections were key elements, because these data were used to gauge the effectiveness of treatments on the landscape. FVS projections, including fuel model data, provided the most likely scenario for the development of vegetation and fuel accumulation on the study site. If more accurate input data on vegetation and fire behavior became available, the models could be rerun for comparison of the results. Similarly, FlamMap output provided crucial data for model comparison and evaluation, and the results might differ from those presented if another fire simulator were used. The conclusions regarding the effects of spatial patterns on fire hazard reduction, however, seem unlikely to be affected.

3.5 Conclusion

This study produced an important result regarding fire hazard reduction. The optimization results showed that the spatial allocation of treated units did not contribute to a change in fire hazard distribution; rather, the key criterion was fuel load reduction. Model NSB, which produced the best fire hazard reduction and solved quickly to optimality, can be used in planning fire management activities and assessing their potential effects on ecosystem services, such as water quality control or wildlife habitat protection.

Chapter 4

TRADEOFFS BETWEEN FOREST FIRE HAZARD REDUCTION AND THE PROVISION OF ECOSYSTEM SERVICES: A CASE STUDY IN THE DESCHUTES NATIONAL FOREST, USA

4.1 Introduction

Forest management, regardless of ownership, strives to protect forest ecosystems to ensure the sustainable provision of ecosystem services. Services are prioritized differently by different landowners, but mitigation of negative consequences of undesired natural processes, such as wildfire and insect outbreak, is critical for preventing both financial loss and degradation of ecosystem services. Public forest management is guided by environmental regulations (e.g., Healthy Forests Restoration Act, 2003; Multiple-Use and Sustained-yield Act, 1960; National Environmental Policy Act, 1969; National Fire Plan, 2002) that define the responsibilities of federal land management in enhancing forest health and protecting ecosystem services. Specifically, the National Fire Plan (2002) was adopted to direct the efforts of the Department of the Interior and USFS in firefighting, reduction of hazardous fuels, and post-fire rehabilitation and restoration. Private forest landowners are not required, but are encouraged, to plan treatments for fire hazard reduction to protect their forest resources. State programs are developed to reimburse and assist small forest owners in planning and implementing fire management strategies (Washington State Department of Natural Resources, 2013b).

While it has been established that fuel treatments are both important and effective in fire hazard reduction (Noss et al., 2006), these treatments may adversely affect ecosystem services such as water quality and/or the habitat of endangered species. From a fire management perspective, firefighting efforts can effectively prevent fires from destroying large areas of forest where fire hazard has been mitigated (Kalabokidis and Omi, 1998). The prevention of high-severity fires in watersheds is critical, because wildfires can increase soils' hydrophobicity, causing surface and mass erosion, and leading to a significant sedimentation, debris flow and potential channel reorganization (Meyer et al., 2001; Rieman and Clayton, 1997; Wondzella and King, 2003). The sediment increase after such

disturbances might have long-lasting effects on stream channel and water quality, and affect fish habitat and potable water supply. While fuel treatments can reduce fire hazard and prevent high severity fires, the treatments themselves can also negatively affect forest ecosystems. Prescribed burning affects ground cover, causing an increase in sediment and changes in carbon and nitrogen concentration in the soils (Neary et al., 2003). Many authors agree that fuel treatments cause an increase in sediment compared to natural levels (Elliot, 2005; Cram et al., 2007; Reid, 2010; Rhodes, 2007; USDA Forest Service, 2005; Wright et al., 1976). However, the effects of fuel treatments are not as adverse and long-lasting as the effects of wildfire (USDA Forest Service, 2005). Similarly, strategically allocated fuel treatments mitigate fire hazard and fire severity to reduce the risk of habitat attrition (Courtney et al., 2004). However, there is uncertainty regarding the short term effects of fuel treatments on northern spotted owl (*Strix occidentalis* caurina; NSO) habitat (Bond et al., 2002; Lee and Irwin, 2005). Because fuel treatments reduce canopy closure, stem density and ground cover, they might disrupt NSO habitat and/or that of its prey species. While low and mixed severity fire might enhance habitat in the long term, high severity wildfire would destroy it for a long period of time (Courtney et al., 2004). Fuel treatments showed to be effective in reducing fire severity and fire hazard (Fernandes and Botelho, 2003; Pollet and Omi, 2002) and, thus, can benefit habitat protection.

Forest management is a complex and multi-objective problem because of multiple goals, different ways to achieve these goals, and potential tradeoffs among these goals that should be accounted for in building sustainable management strategies. Multi-objective optimization has become an important tool in natural resource research (e.g., Bare and Mendoza, 1988; Kennedy et al., 2008; Maness and Farrell, 2004; Tóth et al., 2006; Tóth and McDill, 2009). Multi-objective models allow the analysis of the effects of management decisions on each of the objectives simultaneously, and the assessment of the tradeoffs among them. Optimization models have been used to analyze whether fuel treatments compromise NSO habitat: Gaines et al. (2010) reported habitat protection to be a limiting factor for optimal fuel treatments, whereas Lee and Irwin (2005) concluded that modest fuel treatments are unlikely to affect NSO habitat, though they conceded the need for empirical validation. Calkin et al. (2005) showed that treatment of the majority of the late-seral units would not substantially reduce fire threat in the landscape. Ager et al. (2007) showed that wildfire risk reduction, measured both as burn probability and expected loss, can be achieved by strategi-

cally allocating treatments outside the Davis Late-Successional Reserve, OR, valued for its wildlife habitat. Kennedy et al. (2008) and Lehmkuhl et al. (2007) proposed a new method for solving multi-objective optimization problems in environmental management and tested it by evaluating tradeoffs between fire spread reduction and the protection of the endangered species habitat. The review above showed that the results regarding tradeoffs between fire hazard reduction and NSO habitat protection are sparse and sometimes inconclusive. In addition, none of the published studies assessed tradeoffs between fire hazard reduction and the provision of several ecosystem services in combination for application in planning and decision making in forest management.

In this study, I propose using a mathematical modeling approach to evaluate tradeoffs between management objectives and the provision of ecosystem services. The proposed method allows 1) clarifying whether or not these objectives compromise each other, 2) quantifying tradeoffs between the provision of the ecosystem services and management objectives identified by the forest managers, and 3) designing fuel management plans that account for ecosystem services. I test this method using data for the Drink Planning Area in Deschutes National Forest and also assess short and long term environmental risks of fuel treatments for ecosystem services in this area. The results of this study could assist planning and decision making in forest management in general, and provide the basis for a comprehensive approach to simultaneously protect multiple ecosystem services.

4.2 Methods

4.2.1 Study area

The data for this study are from the Drink Planning Area, a 7056 ha area located on the eastern slopes of the Cascade Range in the Deschutes National Forest (Fig. 4.1). The area provides a variety of important ecosystem services that might compromise each other and, thus, is an excellent study site for exploring the research questions of this chapter.

One of the management objectives in the Drink area is to reduce fire hazard to allow managers to plan effective firefighting strategies to contain wildfires. The Drink area has not been harvested and fuels have not been treated there; the forest stands in the area have reached or are reaching the site's maximum productivity. Dense stands in the study area create an environment prone to severe fires (Hessburg et al., 2005), which are challenging to contain. Because the climate in the study

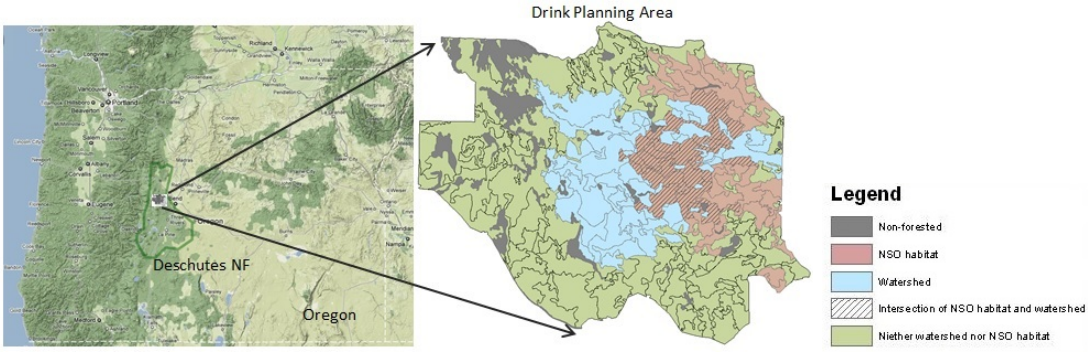


Figure 4.1: Bend municipal watershed and potential NSO habitat in the Drink Planning Area.

area is hot and dry in the summer and cold in the winter, the decomposition of dead vegetation is slow and fuel accumulation is high. All of that contributes to high fire hazard.

The Bend municipal watershed comprises one third of the Drink area. Water quality control is a key ecosystem service, because 50% of the water supplied to the city of Bend is drawn from the watershed in the area. However, the soils in the watershed are sensitive to any disturbances, and any removal of ground cover would cause an increase in sediment yield and delivery to streams and, thus, water quality degradation. Both wildfire and fuel treatments will affect water quality, though, the magnitude and longevity of the impact will differ.

Finally, 18% (1259 ha) of the Drink area is potential NSO habitat, and the USFS must prevent the loss of such habitat according to the requirements of the Endangered Species Act (1973). Fuel treatments in the area would reduce fire hazard, but also change vegetation parameters, and might disqualify the area from being viable NSO habitat.

In this study, I assess the effects of fuel treatments designed to reduce fire hazard on water quality and NSO habitat. The two ecosystem service objectives in this project were defined based on the most important functions of the study area.

4.2.2 Data

The study area was divided into treatment units, each of which was associated with one or more tree lists. Using a division into units based on tree lists alone would not be beneficial, because

often contiguous groups of the same tree list were too small to treat each one as a single unit. Tree list data was provided by the Forest Service and was compiled with nearest neighbor imputation (GNN; Ohmann and Gregory, 2002). Estimates for the initial values of the surface fuels of different size classes was obtained from fuel model data (Yanez, 2013).

Treatments specified for the Drink area aim to reduce crown bulk density and remove surface fuels. However, because the Drink area is roadless, the fuel treatments must be performed manually, and only trees smaller than 18 cm (for a few stands 30.5 cm) in diameter can be cut. (The treatment specification is given in Appendix A.) For each unit, there was only one treatment defined, based on its vegetation characteristics and plant association group. Thus, the only decision regarding a given unit was whether to treat it in a chosen period, or not.

I used the Forest Vegetation Simulator (FVS; Dixon, 2013) SORNEC variant (C.E. Keyser, 2012) and its Fire and Fuel Extension (FFE; S.A. Rebain, 2010) to obtain projections of vegetation growth and fire characteristics given different treatment scenarios. FVS variants account for local peculiarities of vegetation and fuel types in different forests throughout the US; thus, this simulator is widely used for the projection of forest characteristics. This software simulates vegetation and fuel parameters in cycle-by-cycle lists, where each cycle represents a period of time for which increments of tree characteristics are predicted. FVS uses inventory tree lists as input data, and the accuracy of the projections depends on the quality of the input.

FVS has its limitations, and I followed the advice of area specialists by making two modifications to obtain more realistic output for the Drink area. The decay rates in the simulations were adjusted using data provided by the Forest Management Service Center, USDA-FS (Rebain, 2012). Canopy closure was adjusted to approximate recent LiDAR data for the area (Yanez, 2013). Finally, crown base height projections were decreased by a factor of two (Yanez, 2013).

I considered a 40-year planning horizon, divided into two 20-year planning periods with treatments occuring in the middle of each period. This period length was chosen according to practical and theoretical assumptions regarding the longevity of the fuel treatments and vegetation growth in the study area (Agee and Skinner, 2005; Fernandes and Botelho, 2003). Because of the slow vegetation growth, projections for shorter planning periods would not differ significantly in terms of vegetation and fuel characteristics.

Vegetation and fire parameters accounted for regeneration in the area following treatment. Simu-

lations were run four times to obtain projections under four scenarios: "No treatment", "Treatment in the first period", "Treatment in the second period" and "Treatment in both periods", using the corresponding treatment specifications for each unit. The output was used to build the models and to evaluate both their performance and whether the treatments changed the vegetation parameters so as to disqualify areas from being NSO habitat.

The criteria used to define NSO habitat in the Drink area included 1) elevation less than 1830 meters, 2) canopy cover at least 60%, and 3) presence of large trees at least 76 cm in diameter. Units that met all of these criteria qualified as potential NSO habitat, provided that they formed clusters, contiguous areas of at least 200 ha (Webb, 2013). If a unit met the criteria regarding the forest structure but was not a part of a cluster it was still accounted for as habitat, but discounted at 50%. I identified the groups of units that can form clusters accounting for those units that become viable NSO habitat under certain treatment scenarios at the end of the planning horizon even if they did not qualify at the beginning.

4.2.3 Procedure for sediment rate estimation

The following procedure describes how sediment coefficients were estimated for each treatment unit in the watershed.

1. I calculated average sediment rates using WEPP Online (Flanagan and Nearing, 1995) using soils texture input, because detailed data about soil series in the format required by the WEPP model was not available. Sediment rates were obtained for five different scenarios: no treatment, thinning, prescribed fire, low severity fire and high severity fire.

2. WEPP Online produced average sediment rates. I used the Modified Soil Loss Equation (MSLE; Warrington, 1980) to extrapolate per-pixel sediment rates from the average sediment rates because the model accounts for forest specific factors.

   MSLE includes rainfall (R), soil erodibility (K), slope length (L), slope (S), and vegetation management (VM) factors. For extrapolation, R was assumed to be constant because the study area is relatively small, K was the same for each soil texture, and VM was the same for each type of management (no treatment, prescribed burn, etc.). Therefore, the model can

be simplified:

$$S = C \cdot LS, \tag{4.1}$$

where $C = R \cdot K \cdot VM$ was estimated from the rates calculated in the WEPP model using the average LS factor. The constant C was particular to each combination of soil texture K and management VM.

3. The portion of sediment delivered to the stream is characterized by the sediment delivery ratio (SDR). I used Spatially Explicit Delivery MODel (SEDMOD; Fraser, 1999), an ArcGIS module, to calculate spatially distributed sediment delivery ratios for each pixel. Sediment yield was then calculated as a product of estimated erosion and SDR:

$$A = S \cdot SDR \tag{4.2}$$

where A is the sediment rate, and S the sediment estimated using MSLE.

4. The per-pixel sediment data were combined to obtain sediment rates for each unit in the watershed.

5. The sediment rate in (4.2) is the rate in the first year after a disturbance. Because the ecosystem recovers after a disturbance, sedimentation will decrease over time, returning to the pre-disturbance level in 3–5 years (Reneau et al., 2007). Although the decrease is presumed to follow an exponential curve, there are no studies that reported a formal result and/or a model. To calculate the post-disturbance decrease of sediment rates, I used the following procedure to approximate the sediment dynamics reported previously (e.g, (Reneau et al., 2007; Ryan and Elliot, 2005).

   (a) Estimate the sediment rates to represent the sediment delivery in the first year after a disturbance;

   (b) Sediment rates in the first year are discounted at 70% to obtain the second year rate;

   (c) Sediment rates in the second and every subsequent year are calculated by discounting the sediment rate in the previous year by 50% until the value is the same as pre-disturbance sediment rate.

4.2.4 Model

To analyze the tradeoffs between fire management and protection of ecosystem services in the study area, I used a multi-objective optimization model. The objectives considered were fire hazard reduction, water quality control in terms of sediment, and NSO habitat protection.

Fuel model projections from the FVS simulation were used to obtain coefficients to express fire hazard reduction as in model NSB (Chapter 3). The coefficients represent the reduction in fuel model rating, where the rating was assigned to each fuel model based on the flame length, spread rate and fuel loading associated with each of the fuel models. Fuel models with high values of these parameters had a poorer (i.e. higher) rating. I weighted the fuel model rating for each unit by its area and, for each treatment scenario, calculated a coefficient representing its reduction compared to the No Treatment scenario. Fire hazard reduction, in this model, is achieved by maximizing the reduction of the sum of the weighted ratings.

Water quality is expressed as the peak sediment caused by treatments in the two planning periods; and higher sediment value corresponds to lower water quality. I minimized peak sediment, as opposed to total sediment in the entire planning horizon. Peak sediment values are more meaningful for this study because these values distinguish those years when the sediment increases above the natural level, and provide information regarding the water quality decrease caused by treatments. Total sediment values might be high, but would not allow the differentiation and comparison of water quality year to year following treatments.

The protection of NSO habitat is expressed as total area that qualifies as viable habitat post-treatment at the end of the planning horizon. The units that meet the NSO habitat criteria must form clusters; otherwise such units are accounted for as fragmented habitat with a lower ecological value.

The multi-objective model formulation is the following:

Objectives:

$$\max \sum_{i \in I, r \in R \setminus 0} C_{ir} x_{ir} \tag{4.3}$$

$$\max \sum_{i \in I_w, j \in J_r} (a_i p_i + e a_i (\sum_{j \in J} x_{ij} - p_i)) \tag{4.4}$$

$$\min(\max\{\sum_{i\in I, r\in\{1,12\}} s_{ir}x_{ir}, \sum_{i\in I, r\in\{2,12\}} s_{ir}x_{ir}\}) \tag{4.5}$$

subject to:

$$\sum_{i\in I\setminus U} b_i x_{ir} \leq B \quad r \in \{1,2,12\} \tag{4.6}$$

$$\sum_{d\in D_c}\sum_{j\in R_d} x_{rj} - |S_c|q_c \geq 0 \quad \text{for all } c \in C \tag{4.7}$$

$$\sum_{c\in C_d} q_c - p_d \geq 0 \quad \text{for all } d \in I_w \tag{4.8}$$

$$\sum_{r\in R} x_{ir} = 1 \quad \text{for all } i \in I \tag{4.9}$$

$$\sum_{i\in I} a_i(x_{ir} + x_{i12}) = H_r \quad \text{for all } r \in \{1,2\} \tag{4.10}$$

$$H_r \leq A \quad \text{for all } r \in \{1,2\} \tag{4.11}$$

$$lH_1 - H_2 \leq 0 \tag{4.12}$$

$$-uH_1 + H_2 \leq 0 \tag{4.13}$$

$$x_{ir}, p_i, q_c \in \{0,1\} \quad \text{for all } i \in I, r \in R, c \in C, d \in D \tag{4.14}$$

Variables in the multi-objective model are the following:

x_{ir} = one if unit i is assigned a prescription r and zero, otherwise.

p_d = one if unit d belongs to a cluster that satisfies the NSO habitat criteria;

q_c = one if cluster c meets the NSO habitat criteria at the end of the planning horizon;

H_r = an accounting variable for the total area treated in period r.

The model parameters are the following:

I = the set of treatment units (indexed by i);

R = {1, 2, 12, 0}, indicating treatment in period 1, 2, both or none, respectively; indexed by r;

I_w = the set of units that potentially satisfy the NSO habitat criteria ($I_w \subset I$).

U = the set of treatment units that are non-forested;

D_c = the set of units that form cluster c ($D_c \subset I$), indexed by d;

R_d = the set of prescriptions where unit d satisfies the habitat criteria, indexed by j;

C = the set of all clusters, indexed by c;

C_d = the set of clusters that contain management unit d ($C_d \subset C$);

B = the budget limit allocated for fuel treatments in each planning period, in US dollars;

b_i = the cost of treatment in unit i in US dollars, discounted at 4% (Kourantidou and Christodoulou, 2012; Row et al., 1981; Rambaud and Torrecillas, 2005);

s_{ir} = the sediment coefficient from unit i if prescription r was applied;

a_i = the area of unit i;

A = the maximum total area that can be treated in each planning period (MAT);

e = the discounting coefficient to account for habitat fragmentation; $e = 50\%$;

l, u = the lower and upper bounds on area fluctuations treated in periods 1 and 2.

Function (4.3) expresses the fire hazard minimization. Function (4.4) maximizes the area that satisfies the NSO habitat criteria at the end of the planning horizon. Function (4.5) minimizes the peak sediment that fuel treatments cause.

Inequality (4.6) does not allow the total cost of fuel treatments exceed budget B. Constraint (4.7) determines whether the entire cluster c satisfies the NSO habitat criteria and constraint (4.8) defines whether unit d belongs to such a cluster at the end of the planning horizon (adopted from Rebain and McDill (2003)). Constraint (4.9) ensures that only one prescription is assigned to a unit in the planning horizon. Constraints (4.10–4.13) work together to balance the area treated in the two planning periods. Accounting constraint (4.10) calculates the area treated in each period. Inequality (4.11) constrains the total area treated in each period. The last two inequalities (4.12)–(4.13) ensure that the area treated in the second period does not decrease (increase) by more than a factor of l (u). Finally, the last constraint set (4.14) defines variables x_{ir}, p_i, and q_c as binary.

I compared tradeoffs among the three objectives under two different restrictions on the total area treated in each period—17% (1214.06 ha) and 34% (2428.12 ha) of the landscape in each panning period—to produce models M_l and M_h, respectively. These limits were defined based on the conclusions of Collins et al. (2010) and Finney (2007) regarding the effects of treatment extent on fire behavior parameters.

The solutions of the models are spatially and temporally explicit management plans that allocate treatments on the landscape. The solutions are Pareto optimal, meaning that an improvement in the value of one objective cannot be achieved without compromising the other(s). Each management plan is associated with specific values of ecosystem services: an estimate of the sediment that will be delivered to the stream and the area of protected NSO habitat as a result of treatments. These

values were used to build tradeoff curves. For a set of management plans, I obtained fire hazard and burn probability maps using FlamMap (Finney, 2006).

I estimated the environmental risks of fuel treatments using the approach described by O'Laughlin (2005). Fires in treated areas are typically low severity (Pollet and Omi, 2002; Safford et al., 2009; Skinner et al., 2004); therefore, for the estimation of sediment production in the treated areas I applied rates for low severity fire. Sediment rates after a high severity fire were applied to the non-treated units accounting for the burn probability: assuming that the wildfire starts in the Drink area, it is likely to be a high severity fire given the vegetation characteristics and fire regimes in the study area (Simpson, 2007).

I accounted for fire severity in estimation of habitat loss: treated units were accounted as intact habitat in case of wildfire (Gaines et al., 2010), whereas non-treated units were considered to be lost. I calculated habitat loss in terms of acres burned, applying burn probabilities spatially. Such calculations provided a very conservative estimation, because wildfires can break the spatial continuity of the habitat, resulting in collections of unburned units that do not form clusters and therefore have lower habitat quality. Because of the unpredictable character of wildfire and wind direction, however, it would be difficult to predict the actual spatial configuration of burned areas and the disruption of the spatial continuity of the areas that qualify as NSO habitat post-fire.

4.3 Results and Discussion

4.3.1 Analysis of the tradeoffs between fire hazard reduction and provision of ecosystem services

The results (Fig. 4.2–4.4) suggested that there were tradeoffs among the three objectives. Fire hazard reduction compromised the provision of both ecosystem services to different extents. Fuel treatments reduce fire hazard, but placed within watershed or NSO habitat, they increase sediment delivery or decrease habitat area, respectively. In addition, fire hazard reduction mediated tradeoffs in the provision of the two ecosystem services.

Tradeoffs between water quality control and fire hazard reduction were the most visible in model M_h: higher fire hazard reduction was associated with higher sediment (Fig. 4.2; C Fig. C.4, C.5). The tradeoff curves for both models showed that better fire hazard reduction would cause a decrease in water quality. Model M_h allowed more treatments compared to model M_l, thus, both

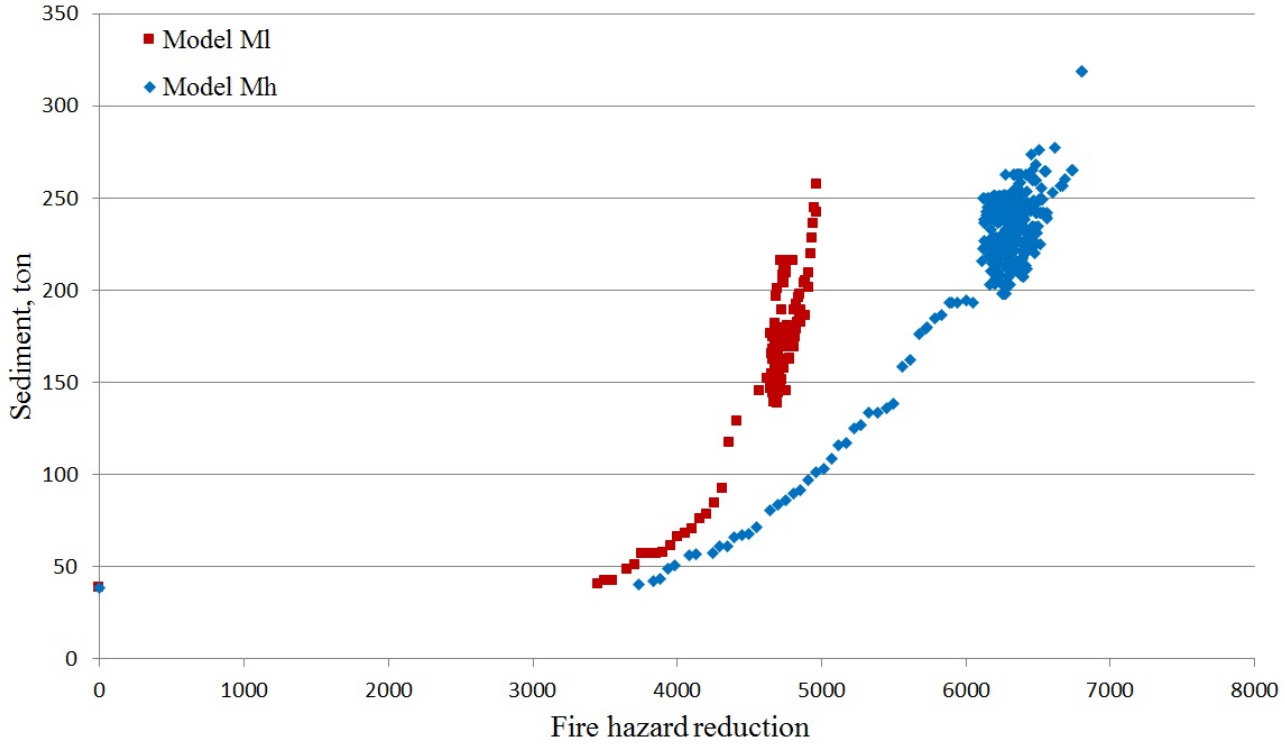
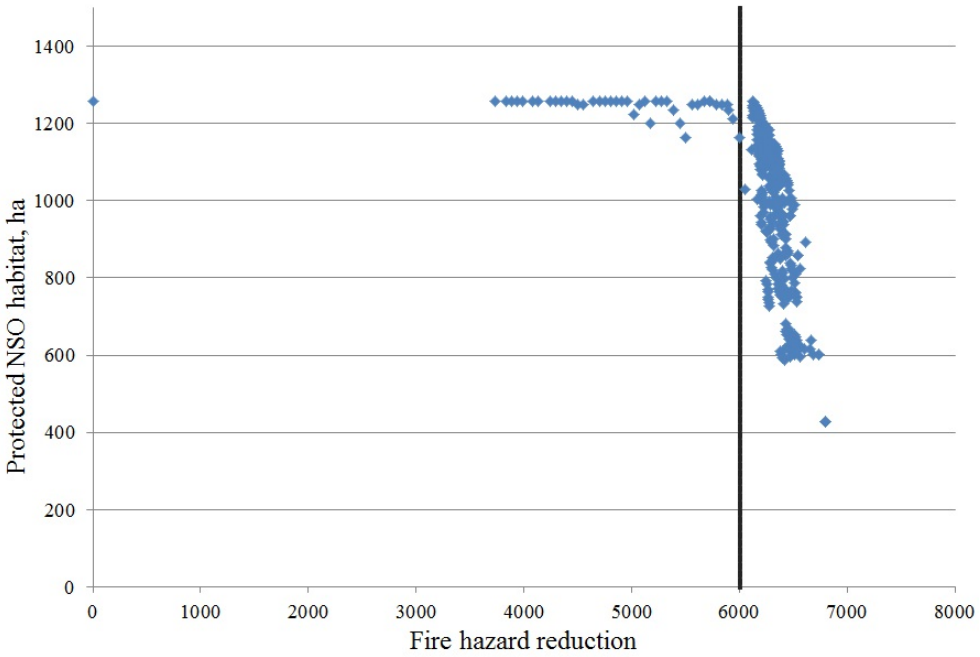


Figure 4.2: Tradeoffs between fire hazard reduction and water quality control in terms of sediment. High values of fire hazard reduction represented an improvement in fire behavior, whereas high values of sediment represented a decrease in water quality.

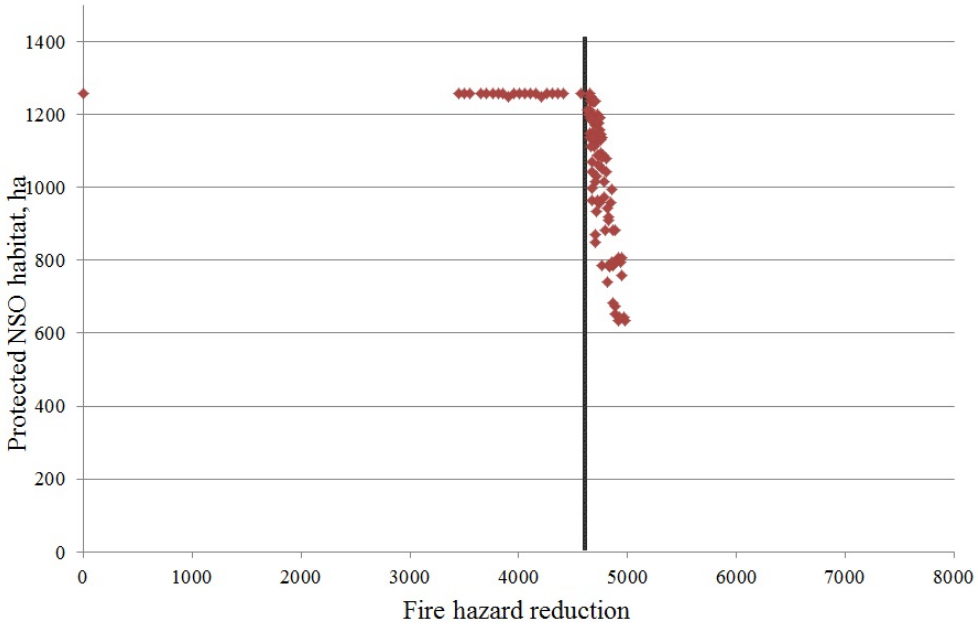
fire hazard reduction and sediment were higher. In model M_l, better fire hazard reduction was achieved only at a higher cost in terms of water quality: the slope of the curve was steeper than that of model M_h as the values of fire hazard reduction increased. The tight constraints on area treated prevented treatment outside the watershed because treatments in the watershed were more effective at reducing fire hazard and contributed the most to the value of the objective function. In model M_l both fire hazard reduction and post-treatment sediment were lower, however, the decrease in the values was not proportional. This explains the differences in the shapes of the curves (the curve for model M_l is steeper) and the character of the tradeoffs between these objectives.

Fire hazard reduction compromised NSO habitat protection when treatments are placed within the existing habitat (Fig. 4.1; Fig. 4.3; C Fig. C.4, C.5). Plotted values of these objectives from both models had similar "elbow"-shaped curves, where the group of solutions to the left of the dividing line corresponded to management plans with no tradeoffs between the objectives. The two objectives compromise each other in solutions to the right of the divider because treatments were allocated on the east side of the Drink area, where the habitat is concentrated. The slope of the curve to the right of the divider was steep, indicating that even a marginal increase in fire hazard reduction would cause a significant decrease in protected NSO habitat.

There was a negative correlation between water quality in terms of sediment and habitat protection for each level of fire hazard reduction. These tradeoffs were more visible in the solutions of model M_h (Fig. 4.4; C Fig. C.2, C.3). Similar to the tradeoff curves for fire hazard reduction and NSO habitat protection, the vertical part of the "elbow" corresponded to the solutions where fuel treatments were allocated outside of the NSO habitat; therefore, there were no tradeoffs between the provision of the two ecosystem services. The horizontal part of the surface showed that, for greater values of NSO habitat protection, fewer treatments were allocated in the potential habitat area but more in the watershed, and vice versa. For water quality control treatments were "pushed" out of the watershed onto the potential habitat, disqualifying units from remaining viable NSO habitat. In the solutions of model M_l, the tradeoffs between water quality and NSO habitat protection were less noticeable because with the low MAT value in model M_l, there were more possibilities for treatments outside both the watershed and the NSO habitat area. The area that was simultaneously potential habitat and watershed was relatively small (Fig. 4.1); nevertheless, solutions where treatments were allocated in the intersection of the watershed and NSO habitat



(a) Model M_h



(b) Model M_l

Figure 4.3: Tradeoffs between fire hazard reduction and NSO habitat protection.

represented scenarios where the provision of both ecosystem services decreased.

4.3.2 Effects of MAT values and total cost of treatments on the objectives

The solutions of both models showed that the correlation between fire hazard reduction and cost of treatments was positive, with higher values of fire hazard reduction corresponding to higher cost (Fig 4.5). Although the total cost of treatments was approximately half as much in model M_l, fire hazard reduction was only 25% lower when compared to the respective objective in model M_h.

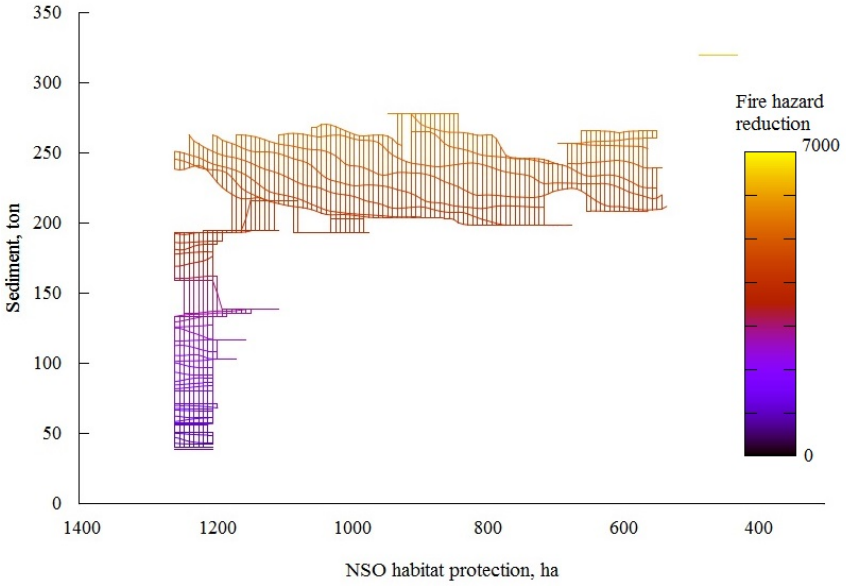
The protection of ecosystem services was better in model M_l, while better fire hazard mitigation was achieved in the solutions of model M_h (Fig. 4.6). The MAT constraints did not affect the values of NSO habitat protection: the range of the habitat protected was the same in both models. However, a decrease in MAT was correlated with a decrease in sediment and fire hazard reduction.

Total area treated in each period affected fire hazard reduction and provision of the ecosystem services differently. Treatments scheduled on larger areas would mitigate fire behavior and, thus, improve fire hazard at the landscape scale but decrease the provision of the ecosystem services. However, fire hazard could be significantly reduced even when the area treated in each period was low. Because the treatments were allocated strategically, in those areas where fire behavior would be mitigated the most, fire hazard was effectively reduced at a lower cost, with less area treated, and less adverse effects on the ecosystem services.

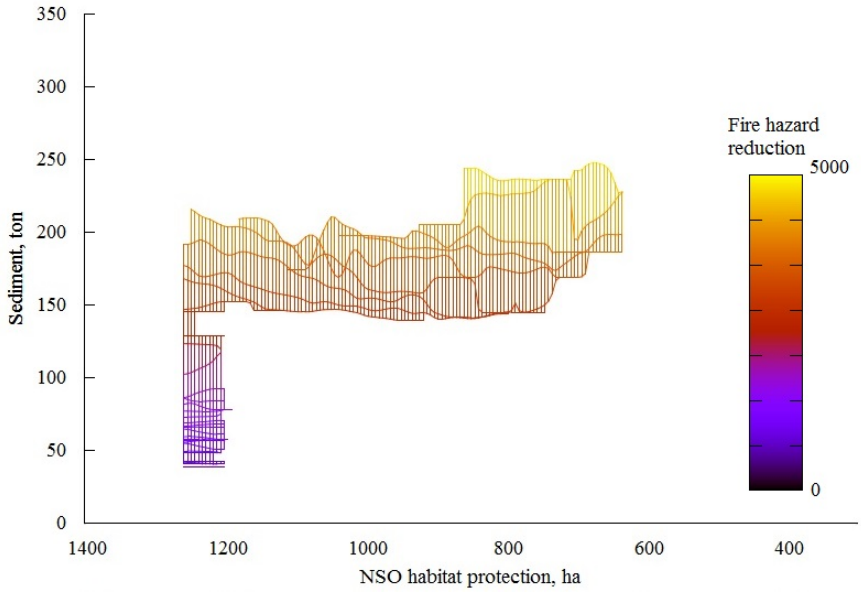
4.3.3 Comparison of environmental risks

Both expected post-fire sediment and expected habitat loss were significantly lower compared to the corresponding values for the No Treatment scenario (Table 4.1, Fig. 4.7). Expected post-fire sediment under the No Treatment scenario was twice as high as the largest expected post-fire sediment in the solutions. Similarly, expected habitat loss was at least 25% greater under the No Treatment scenario compared to the scenarios where treatments were scheduled to reduce fire hazard. Expected post-fire sediment and habitat loss in the solutions of model M_l were generally greater than in model M_h.

Fuel treatments decreased fire hazard and burn probability in the Drink area and, consequently, lowered the expected loss of ecosystem services. The treatments, however, caused a short term

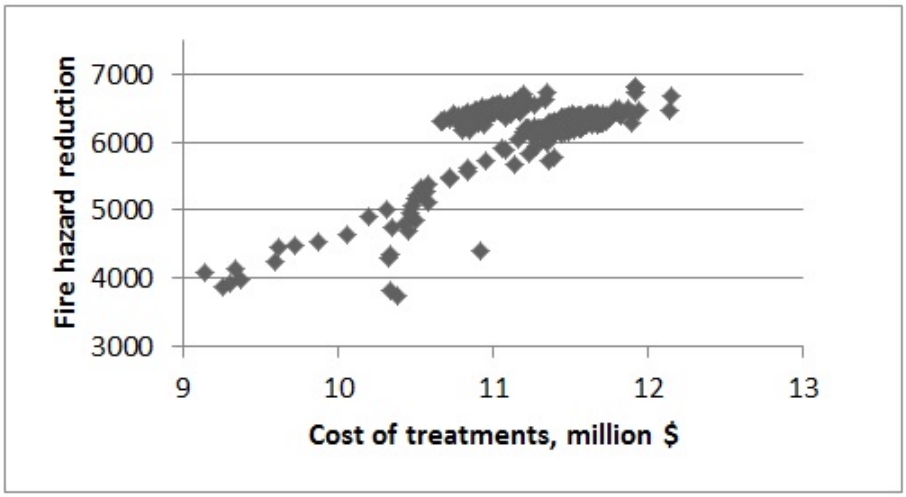


(a) Model M_h

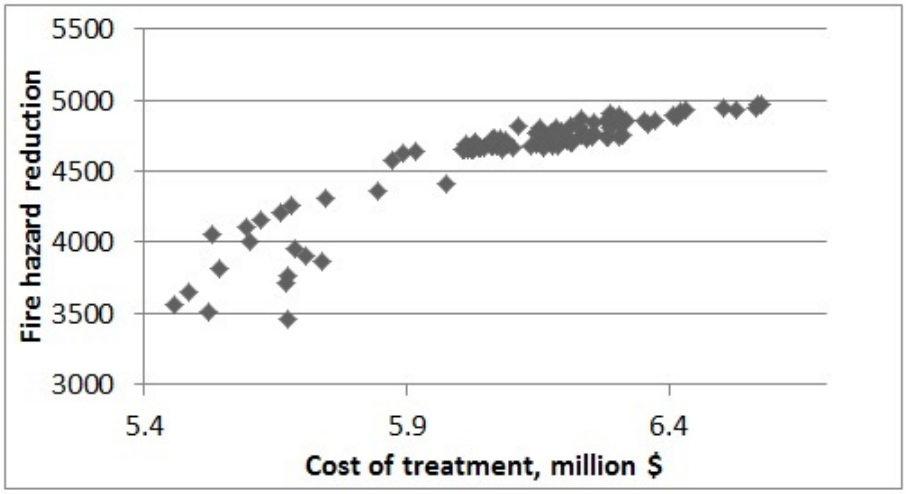


(b) Model M_l

Figure 4.4: Tradeoffs between water quality control in terms of sediment and NSO habitat protection. The contour lines are the levels of fire hazard reduction, which is lowest in the lower right corners of the plots.

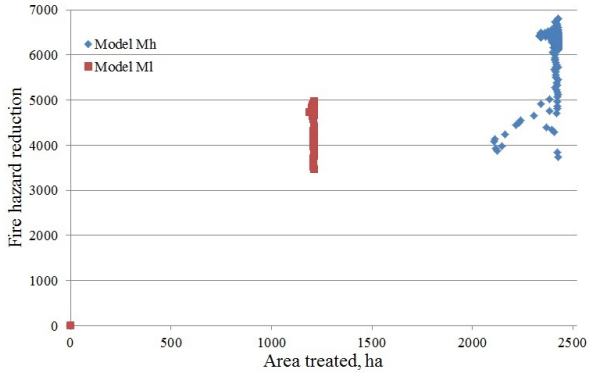


(a) Model M_h

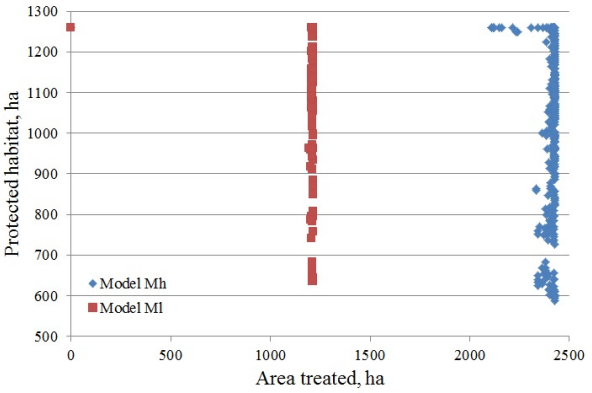


(b) Model M_l

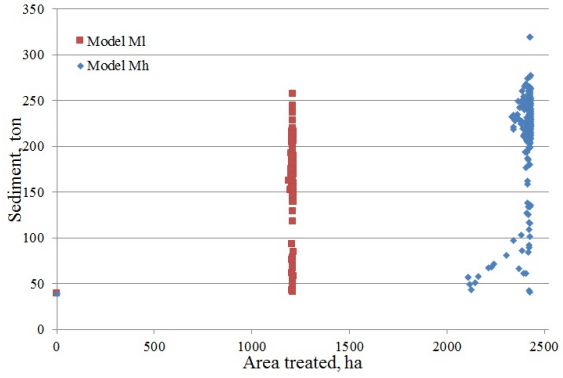
Figure 4.5: Correlation between the total cost of treatments and fire hazard reduction.



(a) Area treated vs. Fire hazard.



(b) Area treated vs. Habitat protection.



(c) Area treated vs. Water quality.

Figure 4.6: Correlation between MAT and fire hazard reduction and provision of the ecosystem services.

sediment increase and loss of NSO habitat. In other words, treatments cause an environmental impact in the short term, but help reduce the risk of a much greater impact in the long term (O'Laughlin, 2005). Because model M_h more effectively reduced hazard landscape-wide, the post-fire loss of the ecosystem services was lower, even when approximately the same area of potential habitat or watershed was affected by treatments in both models.

In both models, the expected post-fire sediment was lower for solutions where fire hazard reduction and treated watershed area were high (Table 4.1). For the same level of fire hazard reduction, the expected post-fire sediment was higher in those solutions where less watershed area was treated. Similarly, for the same area of watershed treated, a higher post-fire sediment level corresponded to a lower level of fire hazard reduction. However, there is an interaction between these factors, because a better fire hazard reduction can be achieved if more area is treated, which in turn causes an increase in post-treatment sediment. A low fire hazard reduction corresponded to a low post-treatment sediment but a high expected post-fire sediment in both models.

Expected habitat loss was correlated with fire hazard reduction in the Drink area, but the total treated area of the NSO habitat did not affect its post-fire loss significantly. Less habitat area was affected by fire in the scenarios with maximum overall fire hazard reduction. In those solutions where treatments were allocated in the NSO habitat area, post-treatment habitat loss was greater. However, the results do not suggest that the treatments in NSO habitat would prevent its loss in case of wildfire. The treatments defined for the Drink area did not effectively reduce fire hazard on the east side of the area, where most of the potential habitat was located. The ineffectiveness of the treatments prevents treatment allocation in the NSO habitat area and, thus, prevents its loss due to treatments as well. The inclusion of additional treatments, such as mowing following thinning and slash removal, might help reduce small fuels, thus catastrophic flame length potential, and thus fire hazard. The results of the study, however, did not show a significant correlation between area of habitat treated and decrease in potential habitat loss due to fire. Treatments allocated outside the habitat area do no affect it but reduce fire hazard and burn probability and, thus, prevent significant loss of habitat in case of fire. Therefore, treatments in the habitat area might not be necessary independent of the fire return period.

The results of this study agreed with previously published reports regarding the tradeoffs between fire hazard reduction and its effects on habitat protection (Kennedy et al., 2008; Lee and

Table 4.1: Post-treatment and post-fire sediment and habitat loss.

Solution	Fire hazard reduction	Short term sediment increase, ton		Area of habitat not affected by treatments, ha	Habitat loss, ha	
		Post-treatment	Expected post-fire		Post-treatment (short term)	Expected post-fire
No-Treat	0.00	0.00	853.59	1,258.96	0.00	1054.62
			MAT = 2428.12 ha/per			
1	6,274.47	262.76	195.74	1,182.43	76.53	170.05
2	6,180.87	224.83	224.03	1,183.74	75.22	182.82
3	6,124.64	236.84	237.64	1,240.83	18.12	199.1
4	5,938.09	193.30	268.80	1,211.23	47.73	205.59
5	5,897.81	193.26	265.67	1,235.75	23.2	211.61
6	5,068.42	108.74	363.69	1,248.87	10.08	250.96
			MAT = 1214.06 ha/per			
7	4,762.76	210.31	283.13	1,135.73	123.22	217.63
8	4,719.58	215.78	296.77	1,201.76	57.2	231.18
9	4,701.20	200.96	320.18	1,235.75	23.21	246.47
10	4,678.22	158.80	370.37	1,182.43	76.53	250.12
11	4,657.67	152.24	381.57	1,198.61	60.35	269.12
12	4,655.76	165.43	355.84	1,248.87	10.08	264.56
13	4,630.03	152.34	375.26	1,211.73	47.23	268.57
14	4,570.10	145.26	384.10	1,258.96	0.00	284.23

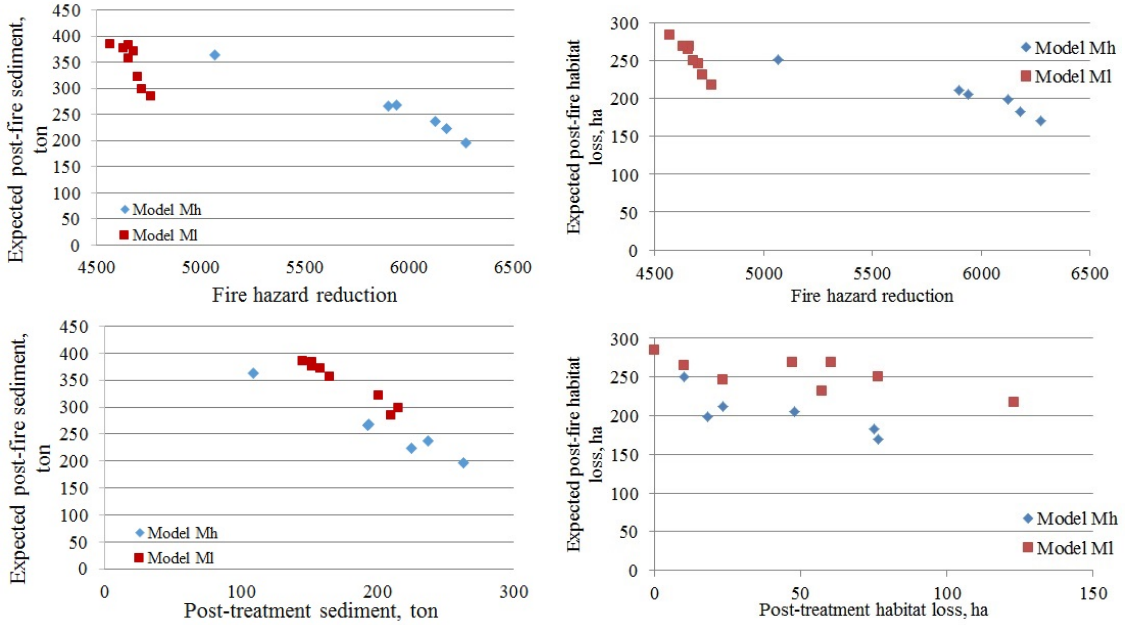


Figure 4.7: Correlation between fire hazard reduction and post-treatment and post-fire sediment and habitat loss. Correlation between fire hazard reduction and post-fire loss of ecosystem services was strong (p-value <0.05). While correlation between post-treatment and expected post-fire sediment delivery was strong (p-value <0.05), the correlation between post-treatment and post-fire habitat loss was weak (p-value ≥ 0.05).

Irwin, 2005). Previous studies showed that treatment of potential habitat might not be necessary to prevent large-scale habitat loss in case of fire (Ager et al., 2007). However, as Lee and Irwin (2005) suggested, moderate treatments might be compatible with both conservation and fire management goals.

4.4 Future research and other applications

In this chapter I analyzed tradeoffs between the provision of the ecosystem services of water quality and NSO habitat protection, but the method I applied is not limited to these two services. For example, the Deschutes National Forest is a popular recreational destination, and fuel treatments might affect the aesthetic values of the forest. Additionally, carbon sequestration is another impor-

tant function for mitigating climate change and currently attracts significant attention (Sundquist et al., 2008). Tradeoffs between the provision of these ecosystem services and/or treatments can be assessed similarly once the effects of treatments on carbon sequestration and aesthetic values are established. The method for assessing environmental risks and tradeoffs between management goals can be applied for decision support in privately owned forests as well: the main changes would be adjusting objective functions to include revenue maximization and/or additional constraints, e.g., timber flow and ending inventory.

The method proposed in this chapter can help improve the understanding of the potential effects of fire management on ecosystem services, and can be used for tactical and strategic planning. Recently, a case was considered in the court of the city of Bozeman, MT, where the reduction of fuels in the forested watershed was proposed as a way to avoid catastrophic post-fire sediment increase in the streams that provide 80% of the source water for the city. Although these treatments would cause moderate water quality degradation in the short term, without the treatments the city's water treatment facilities would not be able to continue supplying water in case of fire (Northwest Environmental Forum, 2012). The city of Bend might face a similar dilemma, and the results of this study could be used to estimate the risks and benefits of different management strategies, inform decision making and the design of long term forest management plans, advocate the potential benefits of forest management and communicate the prospective outcomes of the chosen strategies to stakeholders. Similarly, the method and the results of this study can be used for the realization of the ecosystem services framework being adopted by the Forest Service (Smith et al., 2011), one of whose components involves the evaluation of ecosystem services and the effects of forest management on their provision. Also, this method can be part of an integrated framework for combining ecosystem services for more effective implementation of conservation programs (Deal et al., 2012).

4.5 Conclusion

The approach proposed in this chapter can serve as the basis for a comprehensive framework for analysis, planning and decision support in sustainable forest management. The study in this chapter provides an example of how an optimization-based approach can be used to design management strategies that integrate ecosystem services of water quality and habitat and quantify

tradeoffs among them. The results can inform planning efforts in forest management, because the models operate on objective data with explicit objectives and assumptions. The expertise of forest managers will further refine the suggested management plans, creating well-informed and effective management strategies.

Chapter 5

CONCLUSION AND RESEARCH CONTRIBUTION

The chapters in my dissertation can be considered as separate scientific papers united by the common topic of the integration of ecosystem services in forest management. It is increasingly important to consider this factor while planning actions on forested land. Being a good land steward and providing various ecosystem services might be financially challenging. Thus, approaches are needed both to build management plans taking the provision of ecosystem services into account, and to evaluate the feasibility of these plans in terms of costs and revenue generation. The studies in my dissertation integrated multiple factors that can affect the achievement of the management goals and the provision of ecosystem services, and demonstrated the utility of these approaches independent of forest ownership and area specifics. Each chapter showed a different aspect of the inclusion of ecosystem services—potential conflicts among them and their provision under different environmental regulations.

My research provided a method to analyze whether the provision of different ecosystem services can potentially compromise each other. Active forest management, and in particular, fuel treatments, are necessary for reducing fire hazard and fire severity; however, the treatments can affect the ecosystem and, consequently, provision of ecosystem services in the short term. Furthermore, if there are multiple ecosystem services of interest in addition to management objectives, the achievement of all these goals simultaneously might be impossible. The approach that was built and tested in Chapter 4 allowed a rigorous quantitative analysis of the tradeoffs between different ecosystem services, and showed the potential for its application in different settings.

The approach that was developed for assessment of the effects of environmental regulations (Chapter 2) can be used in similar projects to analyze whether the regulations achieve their intended goals. The contribution of this study is methodological: I built and tested the approach to assess whether the existing environmental regulations achieve their intended goals and whether these regulations inadvertently affect land management when applied in combination with economic and

management constraints.

Fire management is equally, if not more, important than the provision of ecosystem services because of wildfire's potential destructive effects on forest ecosystems and their services. The study in Chapter 3 contributed to understanding the factors that affect fire hazard reduction at the landscape scale. Other important contributions of this study are a rigorous definition of fire hazard and models used to assess changes in fire hazard depending on different fuel treatment allocations. Further work can automate the process described in Chapter 3 to find the most optimal pattern of treatment allocation, where the optimal solution (management plan) is fed directly to the fire simulator (FlamMap). Such a process would allow the inclusion of additional ecosystem services as objectives to the multi-objective optimization models and would allow a more precise assessment of the effects of fuel treatments on forest ecosystems and their services.

In addition to the scientific contribution above, the approaches built for these studies have practical applications. The models built for the case studies can be adjusted for the local constraints and conditions of other sites. Similarly, other ecosystem services can be added to the models depending on management objectives specific to the area of interest. The approaches presented can inform managers about potential tradeoffs among the management objectives, so that decision making and planning relies on objective data. Private forest owners can benefit from using these approaches when assessing opportunities for placing ecosystem services on the market and estimate possible revenues from selling the services either alone or in bundles. The methodologies applied in the studies allow forest managers building strategies and plans that take into account multiple factors and, most importantly, integrate ecosystem services.

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Appendix A

TREATMENT SPECIFICATION FOR THE DRINK PLANNING AREA

Table A.1: Treatments specified for the Drink Planning Area

Stand type	Characteristics	Treatment	Cost
Lodgepole pine dominated	Lodgepole pine stocking is greater than all other species and other species are less than 60 sq ft Basal Area		
AND	Crown Bulk density is > 0.037 kg/cubic meter and trees <7" dbh more than 20 trees per acre	Thin trees <7" dbh to achieve <0.037 kg/cubic meter crown bulk density. Pile and burn slash.	$962.50/acre thinning and piling; $340/acre burning piles
AND	Thinning has occurred and Fuel model = 10, 11, 12 or 13	Pile and burn fuels <7" diameter to achieve 4.4 tons per acre	$722/ acre cutting and piling assume burning at same time as green treatment burning

Table A.1 continued

Stand type	Characteristics	Treatment	Cost
Mixed conifer wet or Mountain Hemlock plant associations not dominated by lodgepole pine	Crown Bulk density is >0.037 kg/cubic meter and trees <7" dbh more than 20 trees per acre	Thin trees <7" dbh to achieve <0.037 kg/cubic meter crown bulk density. Pile and burn slash.	$962.50/acre thinning and piling; $340/acre burning piles
AND	Fuel model = 10, 11, 12 or 13	Pile and burn fuels <7" diameter to achieve 4.4 tons per acre.	$722/ acre cutting and piling assume burning at same time as green treatment burning
AND/OR	Fuel model = 10 (before or after other treatments) and mountain hemlock or white fir >18" dbh greater than 80 sq ft basal area	Prescribed fire burn	$380/ mile hand line construction; $285/ acre underburning
AND	In second entry if burned before and Fuel Models are 6,8,9 or 10	Prescribed fire burn	$380/mile hand line construction; $285/acre underburning
Mixed Conifer dry	Crown Bulk density is >0.037 kg/cubic meter and trees <7" dbh more than 20 trees per acre	Thin trees <7" dbh to achieve <0.037 kg/cubic meter crown bulk density. Pile and burn slash.	$962.50/acre thinning and piling; $340/acre burning piles

Table A.1 continued

Stand type	Characteristics	Treatment	Cost
AND	Fuel model = 10, 11, 12 or 13	Pile and burn fuels <7" diameter to achieve 4.4 tons per acre.	$722/acre cutting and piling assume burning at same time as green treatment burning
AND/OR	Fuel model = 10 or 11 (before or after other treatments)	Prescribed fire burn	$380/mile hand line construction; $285/acre underburning
AND	In second entry if burned before and Fuel Models are 6,8,9,10 or 11	Prescribed fire burn	$380/ mile hand line construction; $285/ acre underburning
All plant associations	SDI <35	Prescribed fire burn	$380 mile hand line construction; $285/ acre underburning

Appendix B

COMPARISON OF OPTIMIZATION MODELS FOR FUEL TREATMENT ALLOCATION

Table B.1: Fire hazard and burn probability distribution for non-spatial models

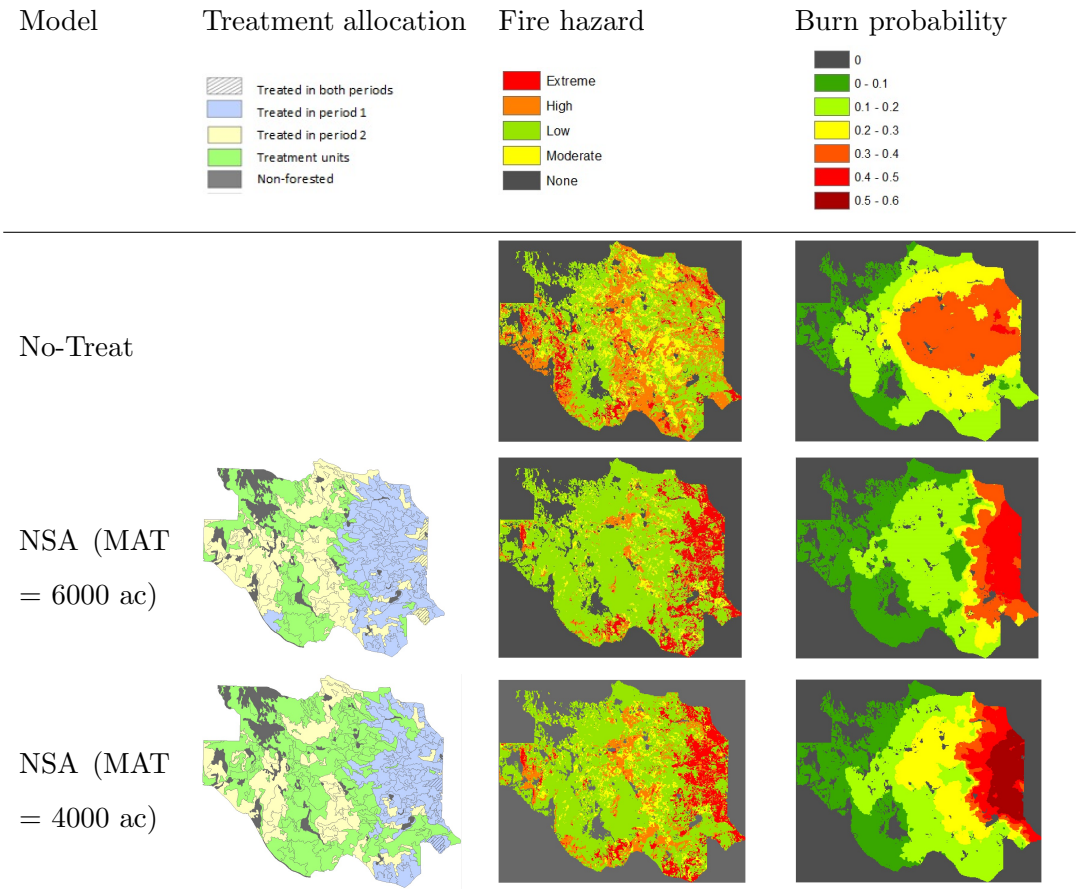


Table B.1 continued

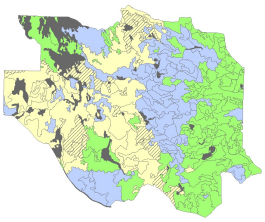
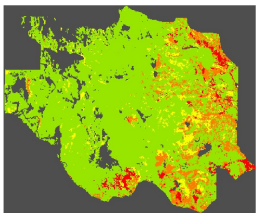
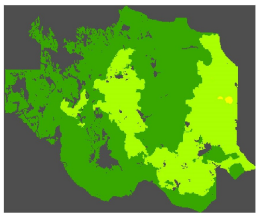
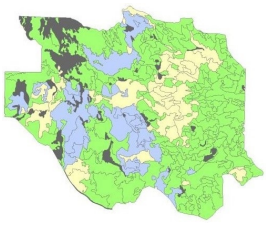
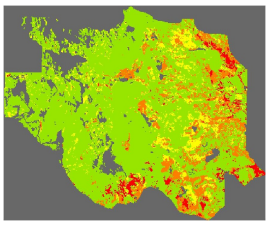
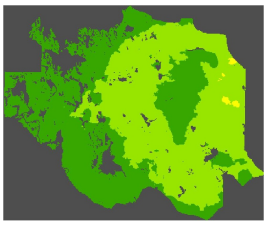
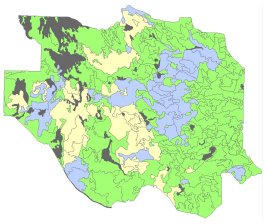
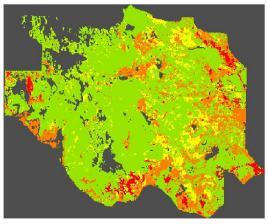
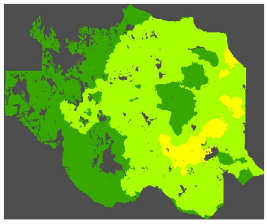
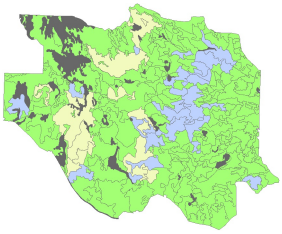
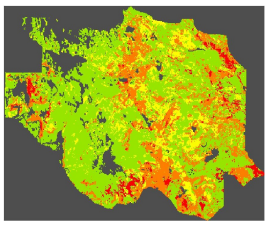
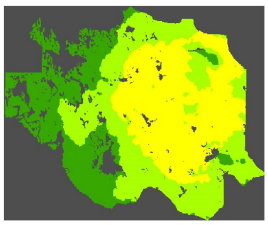
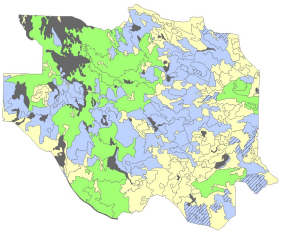
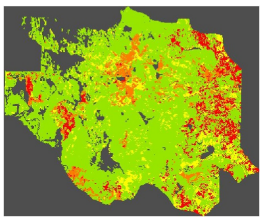
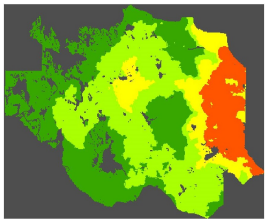
Model	Treatment allocation	Fire hazard	Burn probability
			
NSB (MAT = 6000 ac)			
NSB (MAT = 4000 ac)			
NSB (MAT = 3000 ac)			
NSB (MAT = 2000 ac)			
NSC (MAT = 6000 ac)			

Table B.1 continued

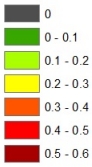
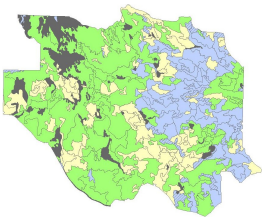
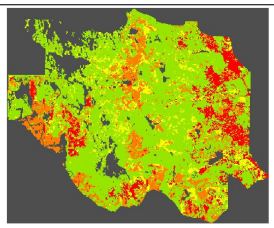
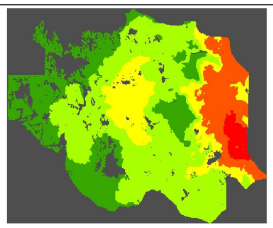
Model	Treatment allocation	Fire hazard	Burn probability
			
NSC (MAT = 4000 ac)			

Table B.2: Fire hazard and burn probability distribution for spatial models

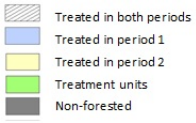
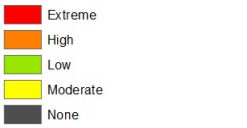
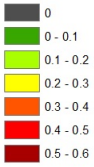
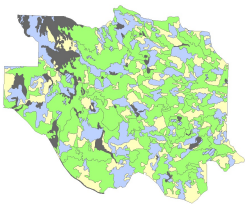
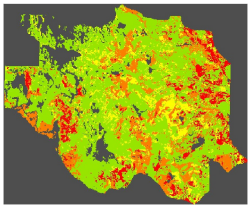
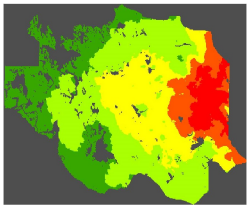
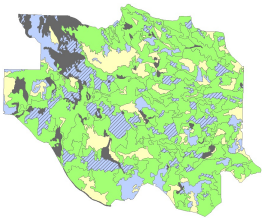
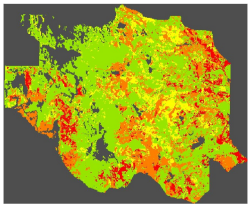
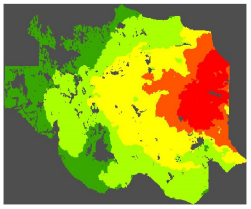
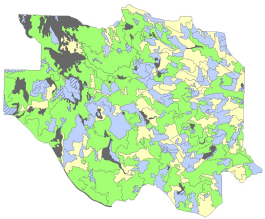
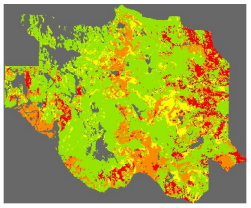
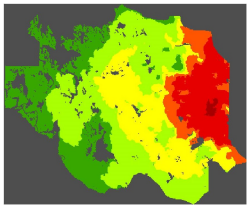
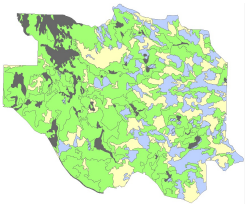
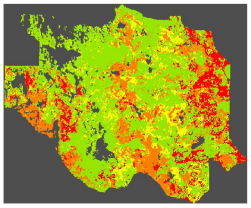
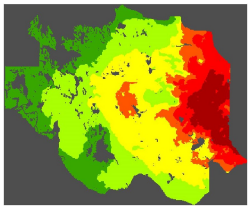
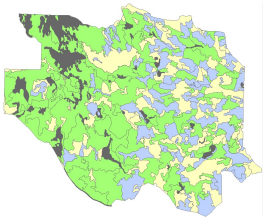
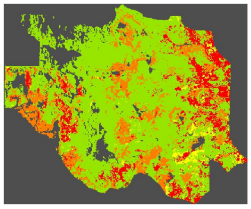
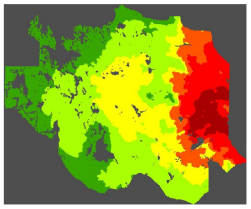
Model	Treatment allocation	Fire hazard	Burn probability
			
S1 (MAT = 4000 ac)			
S2 (MAT = 4000 ac)			
S3 (MAT = 4000 ac)			
S4 (MCTA = 50 ac; MAT = 4000 ac)			
S4 (MCTA = 100 ac; MAT = 4000 ac)			

Table B.2 continued

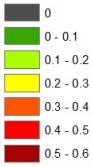
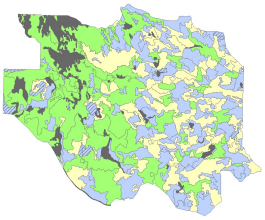
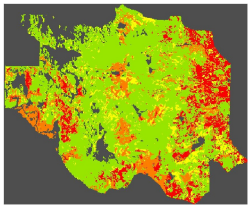
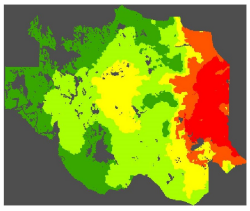
Model	Treatment allocation	Fire hazard	Burn probability
			
S4 (MCTA = 200 ac; MAT = 4000 ac)			

Table B.3: Fire hazard and burn probability distribution for combined models

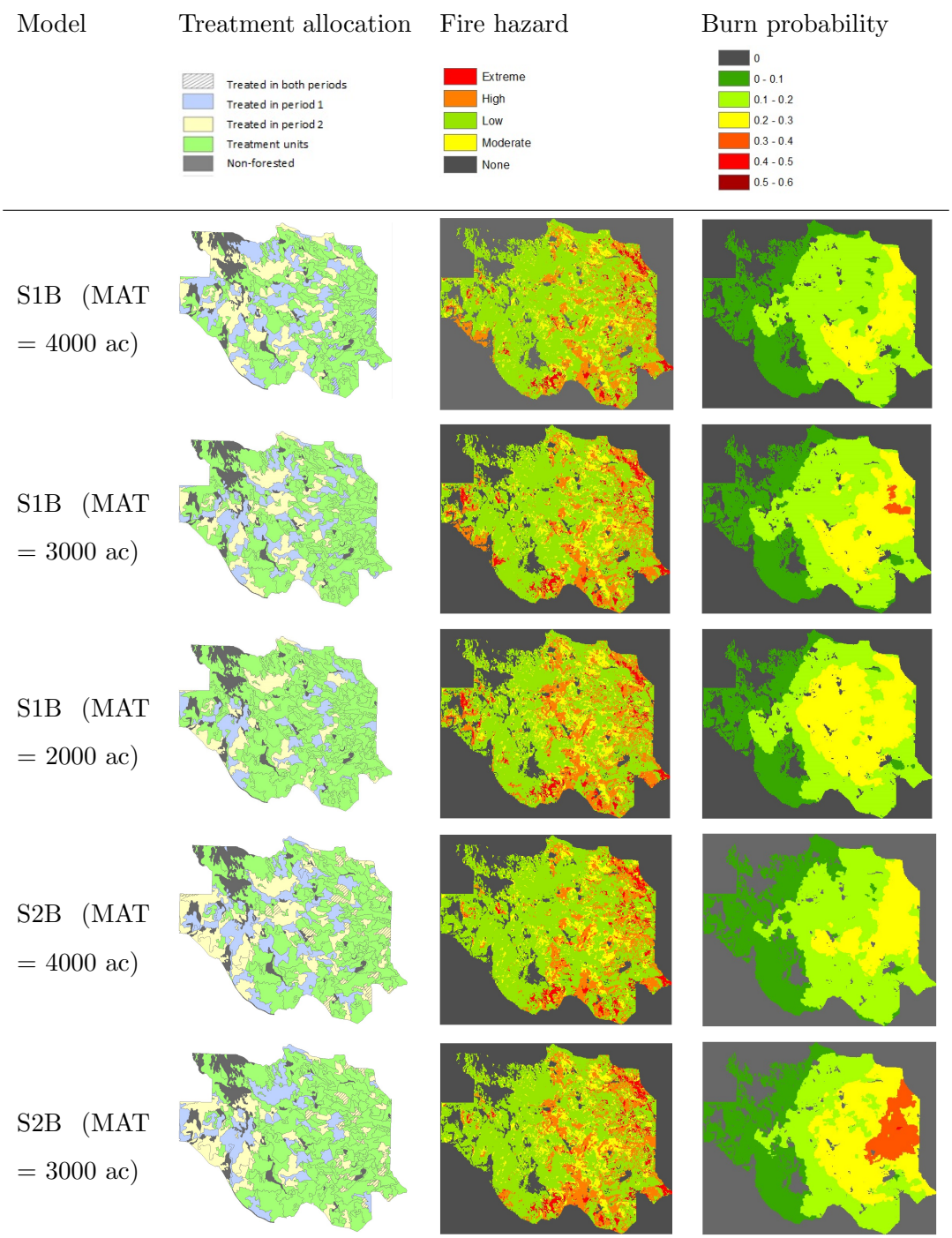


Table B.3 continued

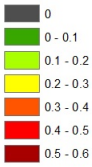
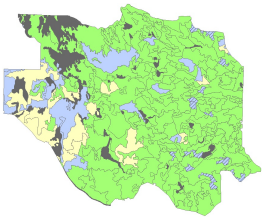
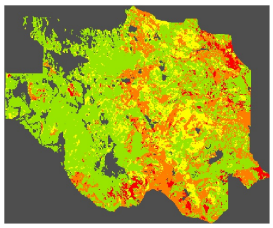
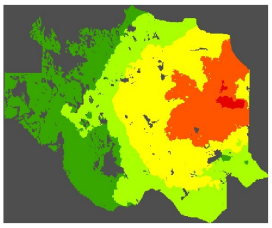
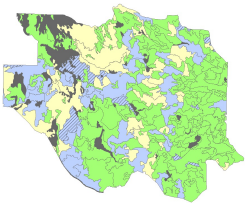
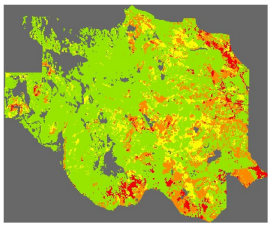
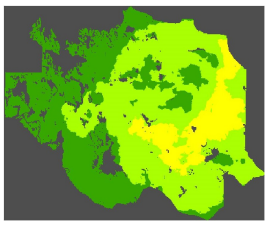
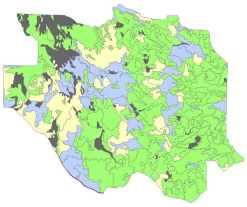
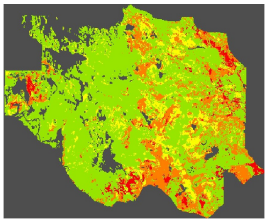
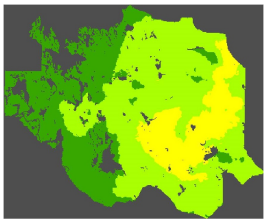
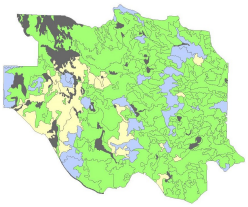
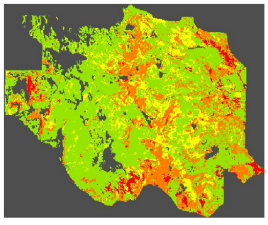
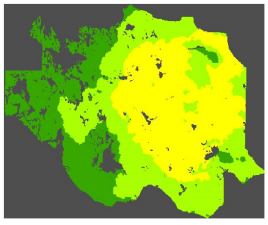
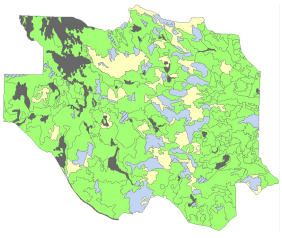
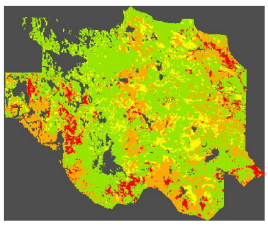
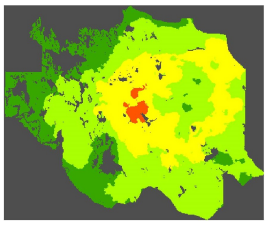
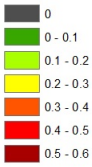
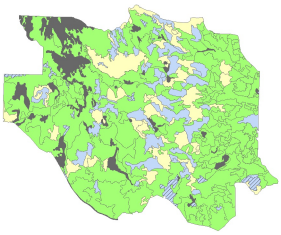
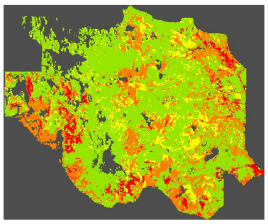
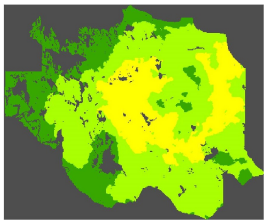
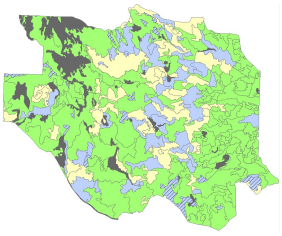
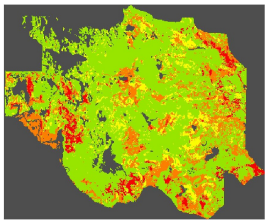
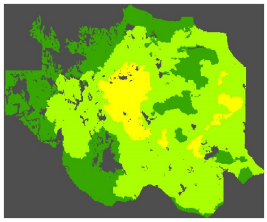
Model	Treatment allocation	Fire hazard	Burn probability
			
S2B (MAT = 2000 ac)			
S3B (MAT = 4000 ac)			
S3B (MAT = 3000 ac)			
S3B (MAT = 2000 ac)			
S4B (MCTA = 50; MAT = 4000 ac)			

Table B.3 continued

Model	Treatment allocation	Fire hazard	Burn probability
			
S4B (MCTA = 100; MAT = 4000 ac)			
S4B (MCTA = 200; MAT = 4000 ac)			

Appendix C

EXAMPLES OF MANAGEMENT PLANS FOR THE DRINK PLANNING AREA

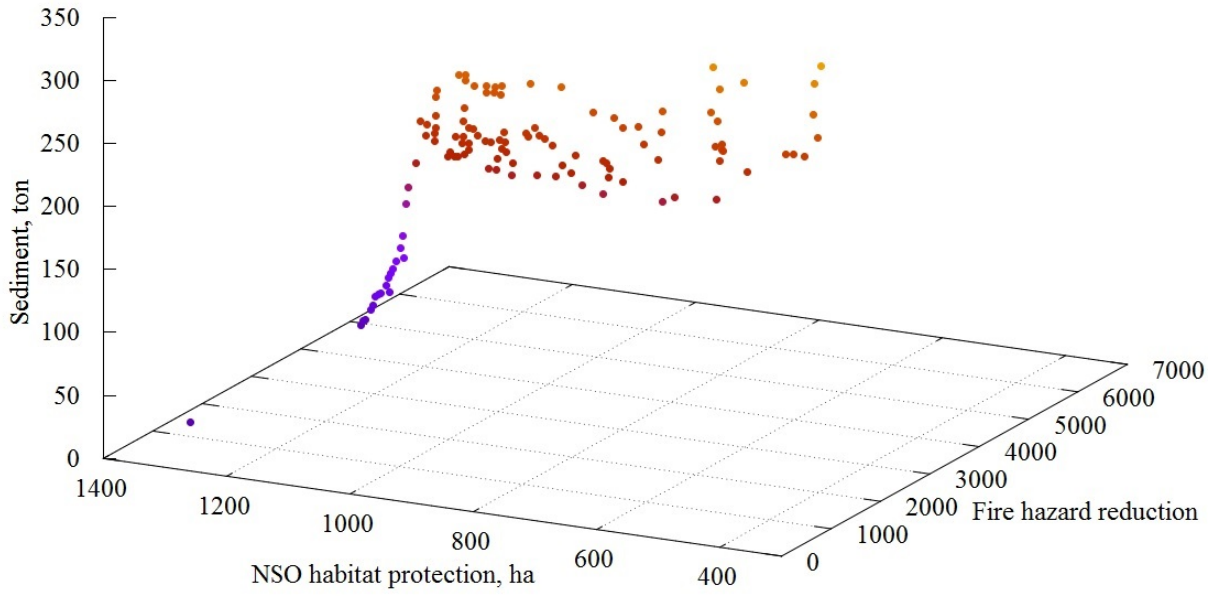
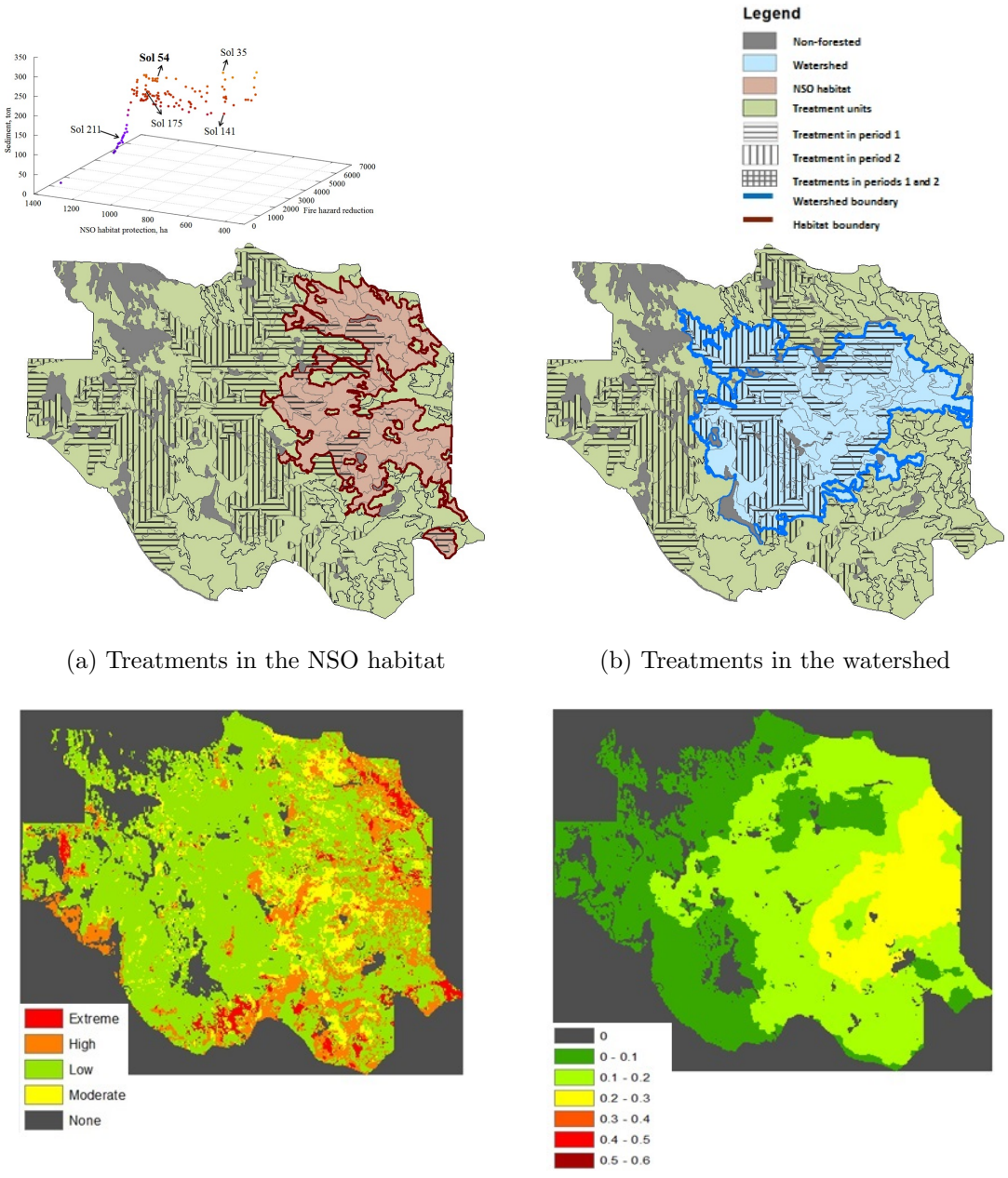


Figure C.1: A point cloud that represents the solution surface for the model with MAT = 1,214.06 ha. Each point corresponds to a management plan that is associated with the values of fire hazard reduction and provision of the ecosystem services of water quality and NSO habitat protection.



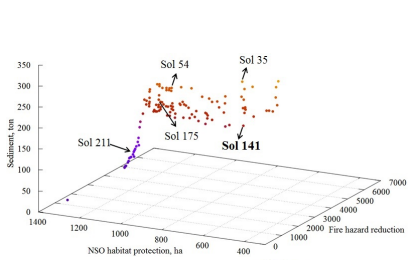
(a) Treatments in the NSO habitat

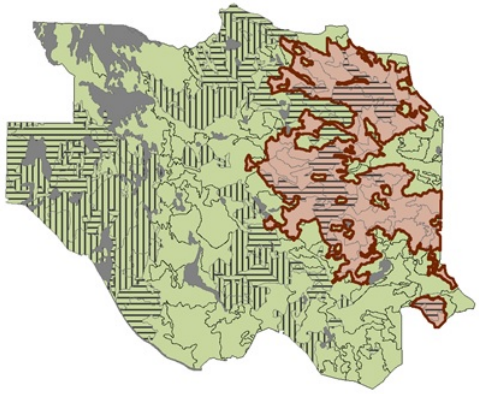
(b) Treatments in the watershed

(c) Fire hazard

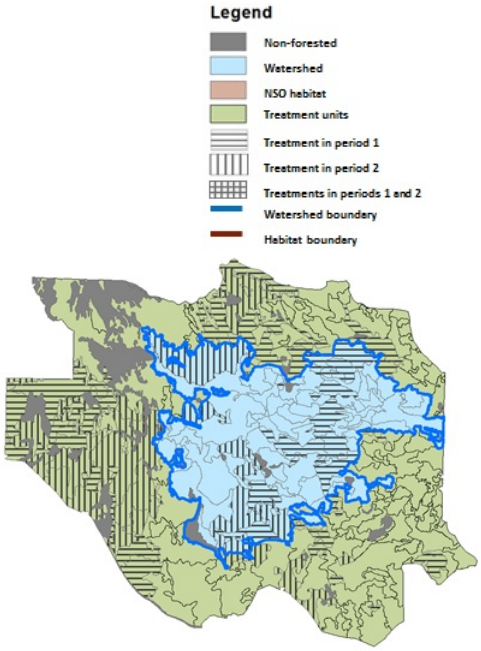
(d) Burn probability

Figure C.2: Treatment allocation for Solution 54 (MAT = 1,214.06 ha). The corresponding management plan provides high fire hazard reduction. NSO habitat is affected by the treatments (habitat protected = 1,135.73 ha) and water quality is low (sediment = 210.31 ton).

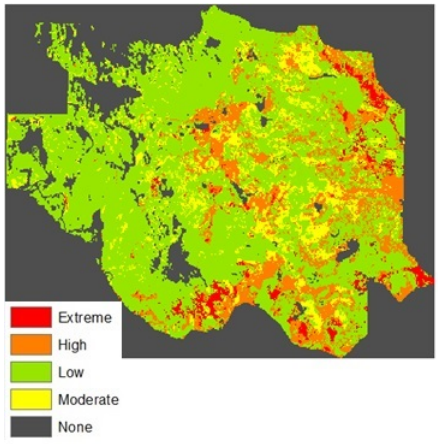




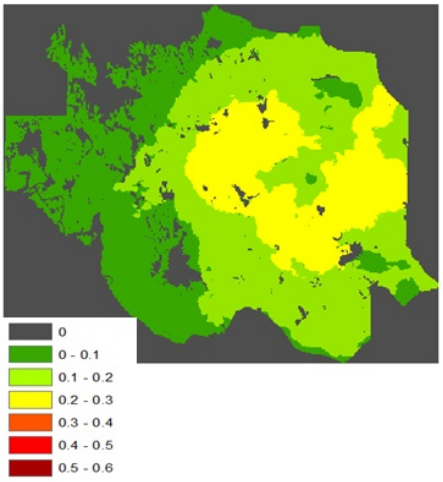
(a) Treatments in the NSO habitat



(b) Treatments in the watershed

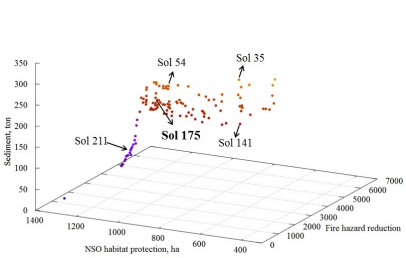


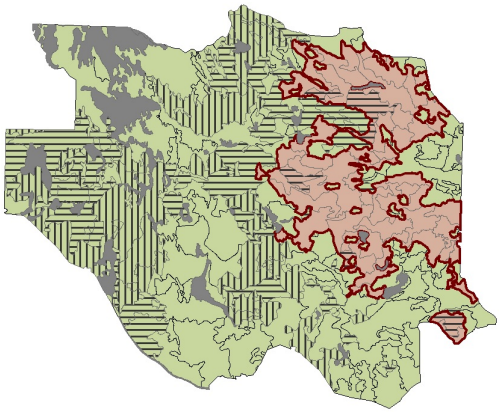
(c) Fire hazard



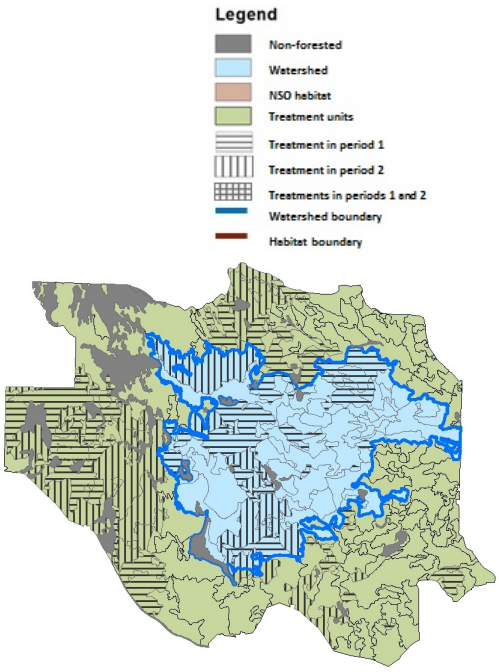
(d) Burn probability

Figure C.3: Treatment allocation for Solution 141 (MAT = 1,214.06 ha). Fire hazard reduction achieved in this management plan is relatively high. Water quality is moderately affected by the treatments (sediment = 145.07 ton), whereas treatments in the NSO habitat disqualifies its significant part: habitat protected is 786.7 ha.

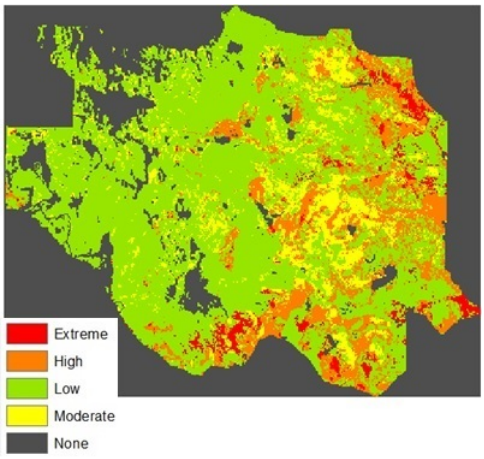




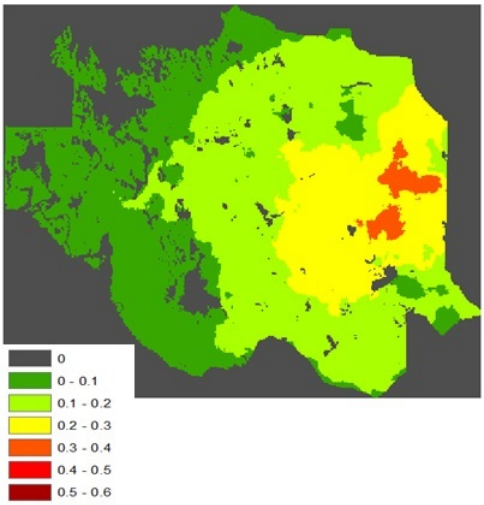
(a) Treatments in the NSO habitat



(b) Treatments in the watershed

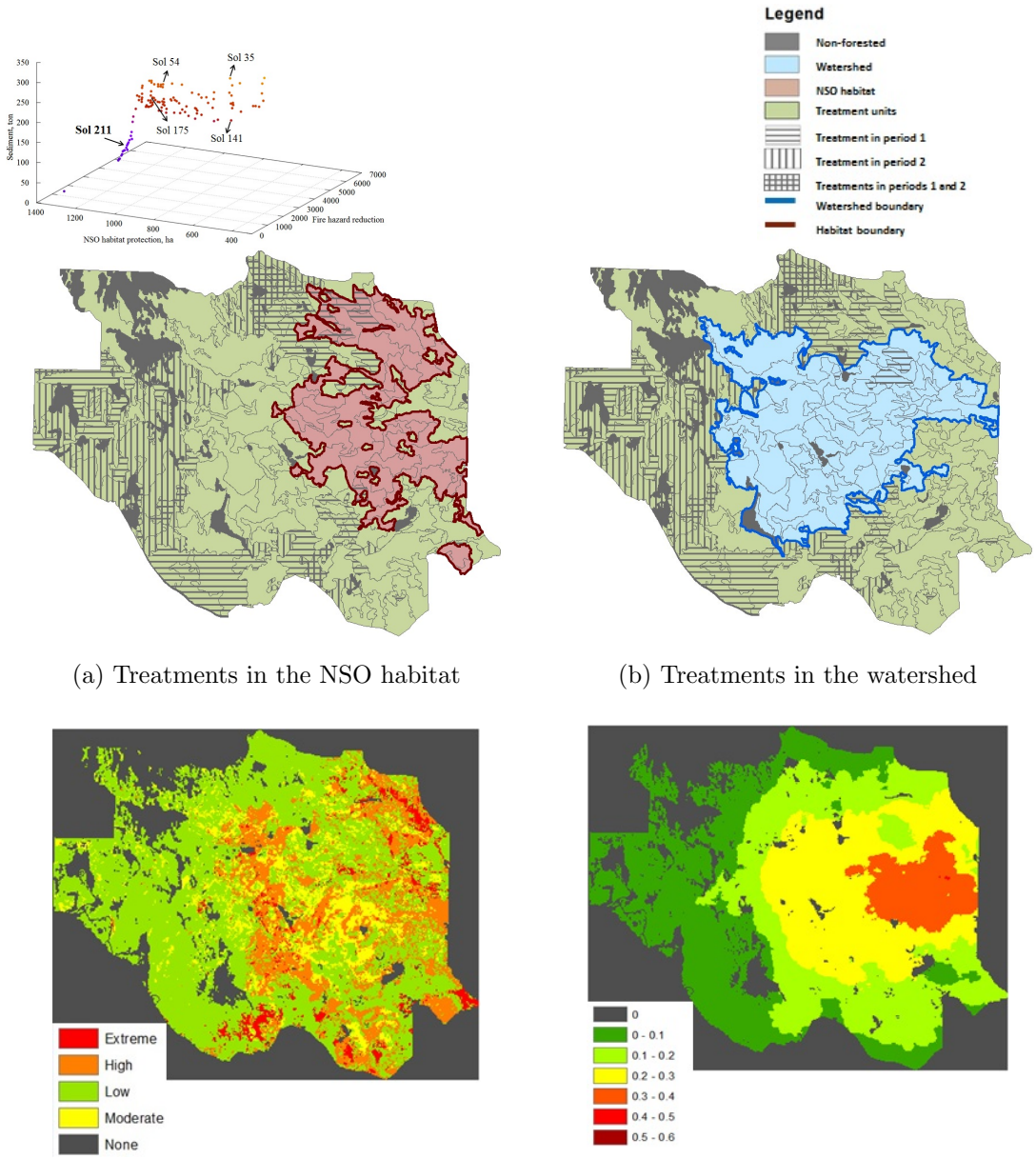


(c) Fire hazard



(d) Burn probability

Figure C.4: Treatment allocation for Solution 175 (MAT = 1,214.06 ha). The management plan achieves relatively good fire hazard reduction. Both water quality and NSO habitat are moderately affected: sediment load is 152.24 ton and NSO habitat protected is 1,198.6 ha.



(a) Treatments in the NSO habitat

(b) Treatments in the watershed

(c) Fire hazard

(d) Burn probability

Figure C.5: Treatment allocation for Solution 211 (MAT = 1,214.06 ha). Treatments according to this management plan do not affect NSO habitat and only marginally decrease water quality (sediment = 40.65 ton). However, the level of fire hazard reduction in this management plan is low.

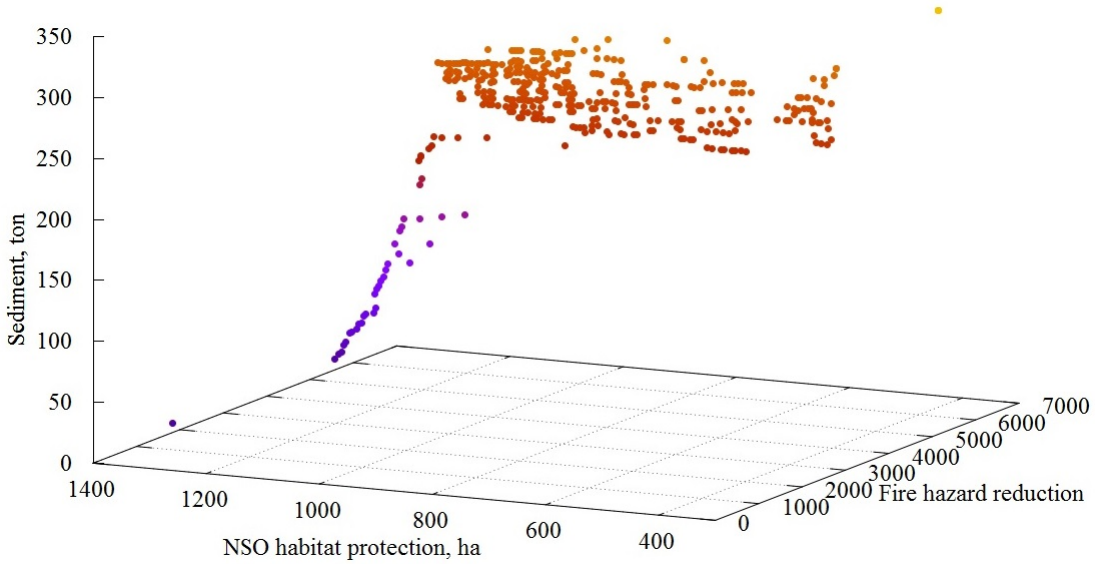
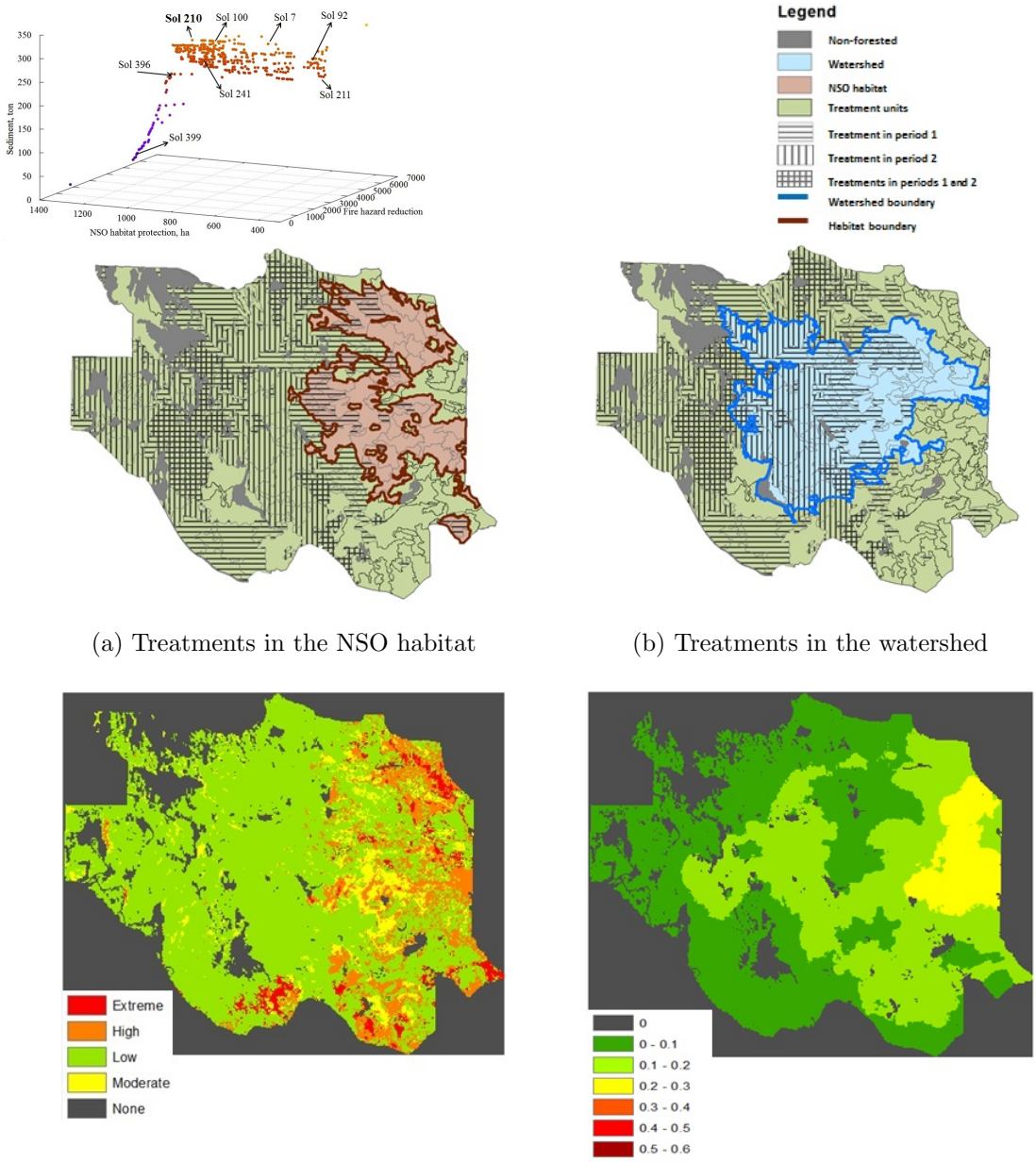


Figure C.6: A point cloud that represents solution surface for the model with MAT = 2,428.12 ha. Each point represents a management plan that provides a certain level of fire hazard reduction, water quality control, and NSO habitat protection.



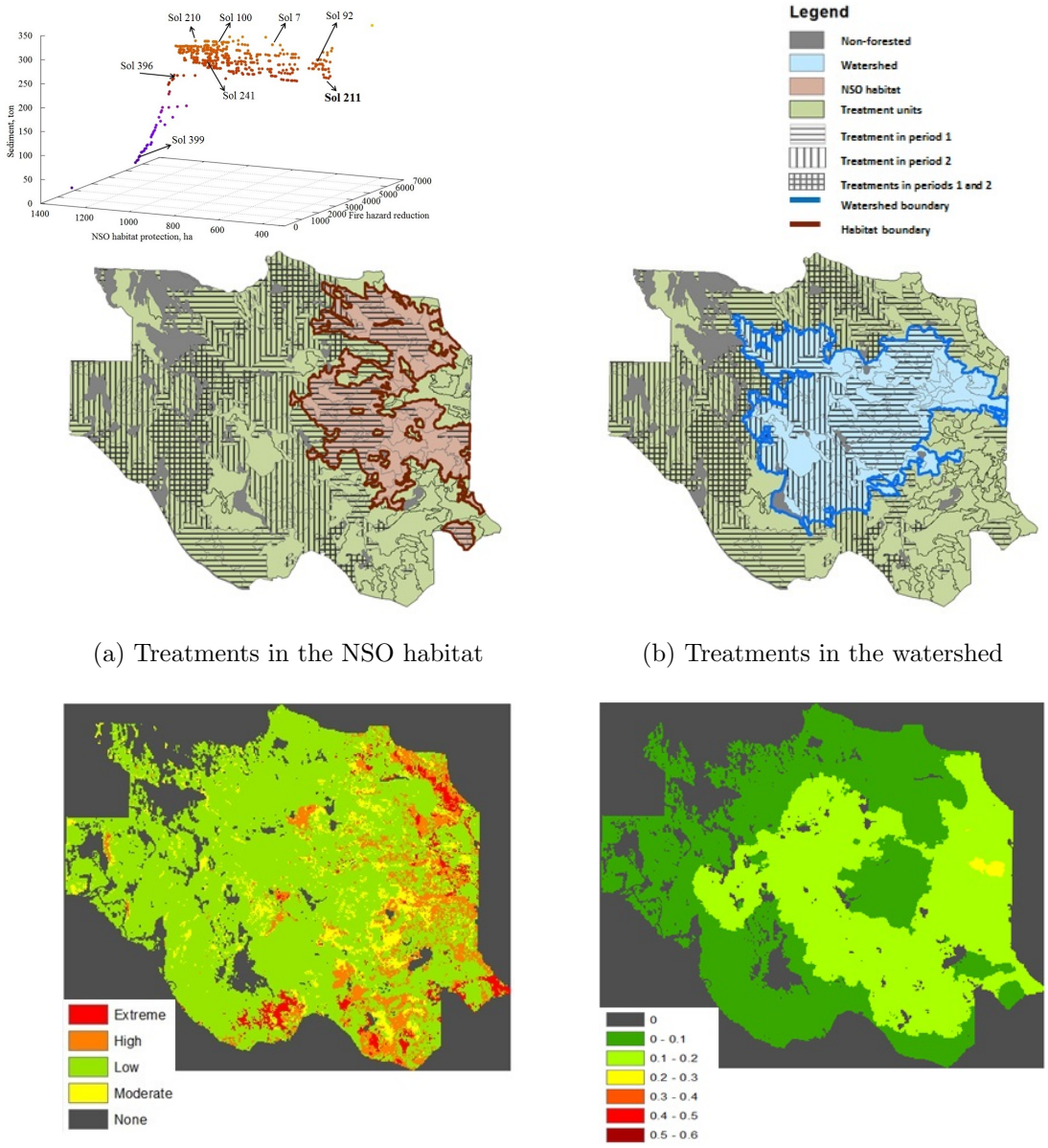
(a) Treatments in the NSO habitat

(b) Treatments in the watershed

(c) Fire hazard

(d) Burn probability

Figure C.7: Treatment allocation for Solution 210 (MAT = 2,428.12 ha). This management plan achieves a high level of fire hazard reduction; however, water quality decrease is significant (sediment = 262.76 ton) and NSO habitat is moderately affected (NSO habitat protected = 1,182.43 ha).



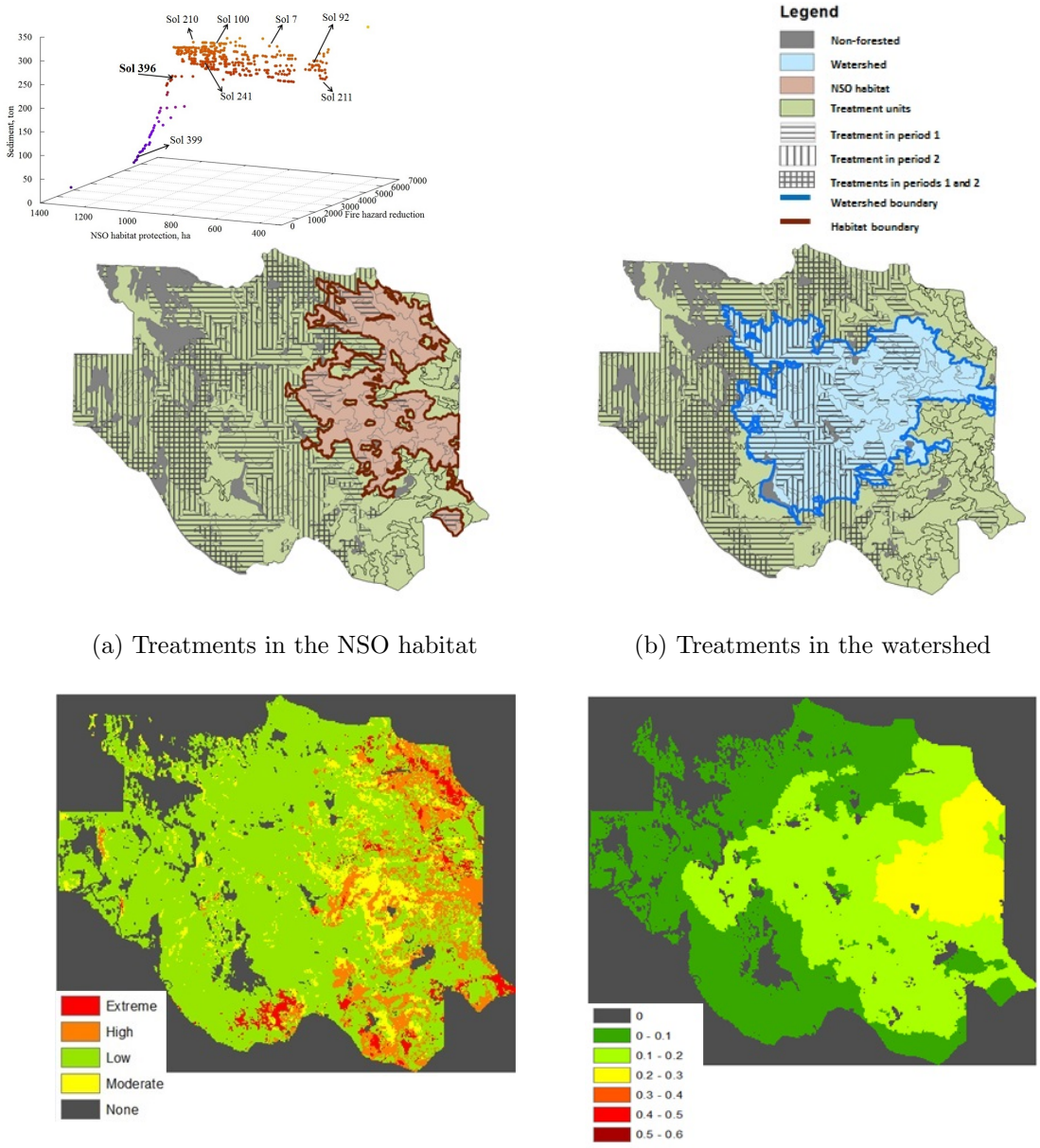
(a) Treatments in the NSO habitat

(b) Treatments in the watershed

(c) Fire hazard

(d) Burn probability

Figure C.8: Treatment allocation for Solution 211 (MAT = 2,428.12 ha). Treatments according to this management plan would reduce fire hazard significantly but decrease water quality (sediment = 211.83 ton) and destroy more than a half of the NSO habitat (NSO habitat protected = 587.94 ha).



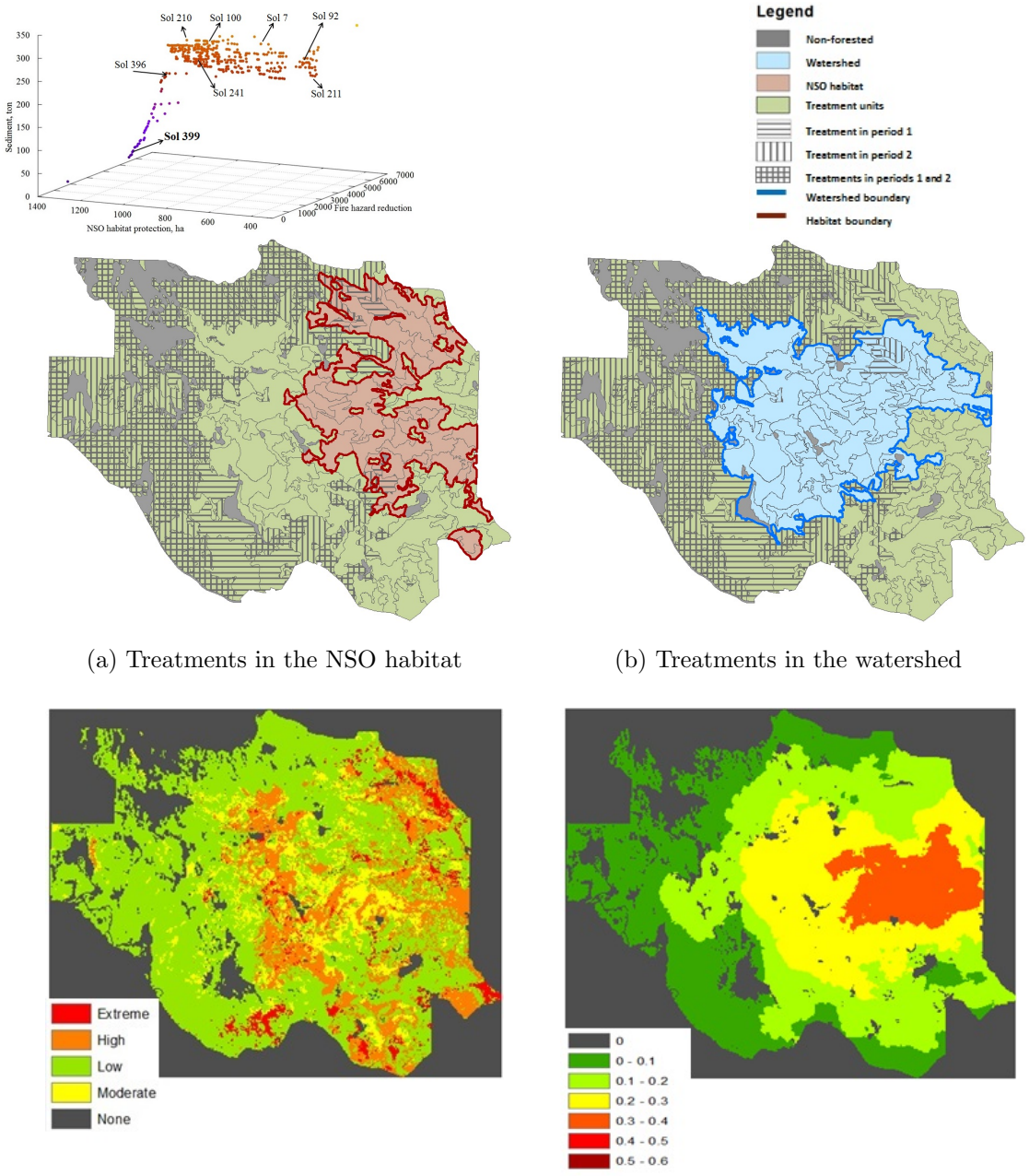
(a) Treatments in the NSO habitat

(b) Treatments in the watershed

(c) Fire hazard

(d) Burn probability

Figure C.9: Treatment allocation for Solution 396 (MAT = 2,428.12 ha). This management plan would provide relatively high level of fire hazard reduction while moderately affecting the ecosystem services: sediment load is 193.24 ton and NSO habitat protected is 1,235.75 ha.



(a) Treatments in the NSO habitat

(b) Treatments in the watershed

(c) Fire hazard

(d) Burn probability

Figure C.10: Treatment allocation for Solution 399 (MAT = 2,428.12 ha). The provision of ecosystem services is not affected in this management plan: water quality decrease is insignificant and sediment level is close to natural. The NSO habitat is protected in its entirety. However, the level of fire hazard reduction is low.

VITA

Svetlana Alekseyevna Kushch (Schroder) was born in Voronezh, Russia. She achieved a Specialist degree, the equivalent to a Masters degree, in Economics and Enterprise Management in Forestry and Forest Industry from the Voronezh State Academy of Forestry Engineering in 2005. She was awarded another Specialist degree in Mathematics from the Voronezh State University in 2007. Svetlana received her Ph.D. degree from the University of Washington in 2013. She worked as a Research Assistant at UW on different projects and built multi-objective optimization models for decision support in forestry management. Svetlana's research interests are multidisciplinary: she uses mathematical programming, specifically multi-objective optimization, for analysis and planning in natural resource management. Svetlana is a member of the Society of American Foresters.