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Instituto Superior de Agronomia
Universidade de Lisboa

INTEGRATION OF FIRE MANAGEMENT IN FOREST PLANNING MANAGEMENT IN PORTUGAL

TESE APRESENTADA PARA OBTENÇÃO DO GRAU DE DOUTOR EM ENGENHARIA FLORESTAL E DOS RECURSOS NATURAIS

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LISBOA, 2013

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Alguém algum dia me disse:

"Sonhe com aquilo que você quer ser,
porque você tem apenas uma vida
e nela só se tem uma oportunidade
de fazer aquilo que quer.

Tenha felicidade bastante para fazê-la doce.
Dificuldades para fazê-la forte.
Tristeza para fazê-la humana.
E esperança suficiente para fazê-la feliz.

As pessoas mais felizes não tem as melhores coisas.
Elas sabem fazer o melhor das oportunidades
que aparecem em seus caminhos.

A felicidade aparece para aqueles que choram.
Para aqueles que se magoam
Para aqueles que buscam e tentam sempre.
E para aqueles que reconhecem
a importância das pessoas que passaram por suas vidas."

O meu bem haja, por estas palavras...

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To all those that despite not helping did not stand in my way: thank you.

Last, but most important, I would like to thank my family, they always were by my side, supporting me and trusting that I could do this and much more...

Again, all my sincere thanks and this dissertation is dedicated to all of them.

Lisbon, December 30 th 2013

Susete Maria Gonçalves Marques

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ABSTRACT

The importance of ecological and socio - economic forest fires shows the relevance of research techniques and approaches for integrating the planning processes of forest management and fire management . This PhD aimed to investigate these approaches and contributing to the effectiveness of strategies for preventing fires both at the stand level or landscape level. In this context, probabilistic models of fire occurrence and estimated mortality to the main Portuguese forest species (maritime pine and eucalyptus) in pure composition of regular and irregular structures were developed. These were later integrated into a management model that optimizes harvests scheduling for each stand in order to create more fire resistant landscapes. It's being developed a technologic platform to test these models and their combination with an innovative approach to incorporate fire risk and protective objectives in forest management planning.

RESUMO

A importância dos impactos ecológicos e sócio - económicos dos incêndios florestais mostra a relevância da investigação de técnicas e abordagens para a integração de processos de planeamento da gestão florestal e da gestão do fogo. Este doutoramento teve como objectivo a investigação dessas abordagens e a contribuição para a eficácia das estratégias de prevenção dos fogos quer ao nível do povoamento, quer ao nível da paisagem. Neste âmbito foram desenvolvidos modelos probabilísticos de ocorrência de incêndios e estimativa da mortalidade e danos para as principais espécies florestais portuguesas (pinheiro bravo e eucalipto) em composição pura em estruturas regulares e irregulares. Estes foram posteriormente integrados num modelo de gestão que optimiza qual o calendário de corte de cada povoamento por forma à criação de paisagens mais resistentes ao fogo. Encontra-se em desenvolvimento uma plataforma tecnologia, a fim de testar estes modelos e sua combinação numa abordagem inovadora para a incorporação de objetivos de risco e proteção no planeamento da gestão florestal.

PREAMBLE

This thesis is composed by several scientific articles. Some of the articles have already been published; others are being edited; while others are ready for submission. The manuscript encloses detailed descriptions of models developed to predict risk and mortality, as well their integration into harvesting models for the landscape level, containing explanations of procedures, decisions and assumptions made throughout this study.

The motivation and the work conducted for the thesis are explained in the general introduction (Chapter I). The articles are integrated as chapters of this document and have a Roman numeral assigned (II – V). The thesis is the compilation of the articles and the manuscript included as chapters of this document:

Table 0-1 - Articles that composes the thesis

Article 1 - Assessing wildfire risk probability in *Pinus pinaster* Ait.stands in Portugal
Article 2 - Developing post-fire *Eucalyptus globulus* stand damage and tree mortality models for enhanced forest planning in Portugal
Article 3 - A three-step approach to post-fire mortality modeling in Maritime pine (*Pinus pinaster* Ait.) stands for enhanced forest planning in Portugal
Article 4 - A Stochastic, Cellular Forest Harvesting Model Integrating Wildfire Risk and Dispersion (work in progress)

Susete Maria Gonçalves Marques had the main responsibility for the entire work in articles for which is the first author. Concerning the article that makes up Chapter IV, as a co-author, the candidate was responsible for all the data collection and treatment, as well did all the statistical work to obtain the models presented in this article. Besides that she helped writing and formatting the document. Chapter

VI, contains the final remarks, summarizing the conclusions of the work and also describing some tasks that were essential to move forward but that were not included in the articles.

The research reported here is part of the PhD supported by grant SFRH/BD/62847/2009 of the Portuguese Foundation for Science and Technology (FCT). The work done for this thesis was developed under the scope of the following international projects:

Table 0-2 – List of International and national projects

Project ref. **FP7-ENV-2011 nº 2825887** entitled **"INTEGRAL** – Future-oriented integrated management of European forest landscapes", in the 7th Framework Programme from UE (FP7-ENV-2011). Coordination: Swedish University of Agricultural Sciences, University of Freiburg (ALU-FR), Germany, University of Padua (UNIPAD), Italy, Wageningen University (WU), Netherlands, University of Forestry Sofia (LTU), Bulgaria, Fachhochschule Salzburg (FHS), Austria, University of Oxford (UOXF.AF), United Kingdom, Instituto Superior de Agronomia (ISA), Portugal, University of Molise (UNIMOL), Italy, Joint Research (JRC), Italy, Technical University Zvolen (TUZVO), Slovakia, Lithuanian University of Agriculture (LZUU), Technische Universität München (TUM), Germany, Fraunhofer Gesellschaft (FhG-MOEZ), Germany, AgroParisTech (ENGREF), France, University College Dublin (NUID UCD), Ireland, Portuguese Catholic University (UCAPOR), Portugal, Confederation of European Forest Owners (CEPF), Luxemburg, Stichting FERN (FERN), EU, European State Forest Association (EUSTAFOR), EU.
Project ref. **FP7-PEOPLE-2010-IRSES / 269257** entitled "**ForEAdapt**- Knowledge exchange between Europe and America on forest growth models and optimisation for adaptive forestry", in the framework of Marie Curie International Staff Exchange Scheme. Coordination: Instituto Superior de Agronomia. Other institutions involved: Swedish University of Agricultural Science, Sweden; Technical University of Madrid, Spain; Pennsylvania State University and Virginia Polytechnic

Institute and State University, USA; University of Eastern Finland, Finland; University of Chile and Catholic University of Chile, Chile; University of S. Paulo, Brasil.

Project ref**. FP7-ENV-2008-1 nº 266544** entitled "**MOTIVE** - Models for Adaptive Forest Managementin the 7th Framework Programme da UE (FP7-ENV-2008-1). Coordination: University of Freiburg (Germany). Other countries involved: Finland, Swissland, Denmark, France, Germany, Netherlands, Poland, Czech Republic, Austria, Portugal, UK, Spain, Romania e Bulgaria

PTDC/AGR-CFL/64146/2006 - Integração da gestão florestal e da gestão do fogo. Modelos e sistemas de decisão (Decision support tools for integrating fire and forest management planning), funded by Fundação para a Ciência e Tecnologia.

Table 0-3 – Other publications where Susete Marques is first author.

Marques S., Borges J. G., Garcia-Gonzalo J., Moreira F., Carreiras J. M. B., Oliveira M. M., Cantarinha A., Botequim B. & Pereira J. M. C., 2011. Characterization of wildfires in Portugal. European Journal of Forest Research, Volume 130, Issue 5 , Page 775-784 DOI: 10.1007/s10342-010-0470-4

Marques S., Garcia-Gonzalo J., Borges J. G., Botequim B., Oliveira M. M., Tomé J., Tomé M., 2011. Developing post-fire *Eucalyptus globulus* stand damage and tree mortality models for enhanced forest planning in Portugal. Silva Fennica 45(1): 69-84.

Marques S., Garcia-Gonzalo J., Botequim B., Ricardo A., Borges J. G., Tomé M., Oliveira M. M., 2012. Assessing wildfire risk probability in *Pinus pinaster* Ait. stands in Portugal. Forest Systems 21(1): 111-120. DOI: http://dx.doi.org/10.5424/fs/2112211-11374 Available online at www.inia.es/forestsystems

Table 0-4 – Other publications where Susete Marques is co-author.

Borges J. G., Garcia-Gonzalo J., **Marques S.**, Valdebenito V., McDill M. E. and Falcão A. O., Chapter 6 - Strategic management scheduling (submitted to Springer)
Borges J. G., Garcia-Gonzalo J., Guerra-Hernandez J., **Marques S.,** Palma J., 2013. Chapter IX – A decision support system for forest management planning under climate change. In Fitzgerald J and Lindner M, (eds).Adapting to climate change in European forests –results of the MOTIVE project. Pensoft Publishers, Sofia, 108 pp.
Natário I., Oliveira M. M., **Marques S.,** 2013. Using INLA to estimate a highly dimensional spatial model for forest fires in Portugal. In a future volume of the series Studies in Theoretical and Applied Statistics, Pacheco, A.; Oliveira, R.; Santos, R. (eds.), Springer, Berlin. Submitted 2012, Accepted 2012
Borges J. G., Garcia-Gonzalo J., Bushenkov V. A., McDill M. E., **Marques S.,** Oliveira M. M., 2013**.** Addressing multi-criteria forest management with Pareto Frontier methods: an application in Portugal. Published online in Forest Science
Botequim B., Garcia-Gonzalo J., **Marques S.,** Ricardo A., Borges J. G., Tomé M., Oliveira M. M., 2013. Developing wildfire risk probability models for *Eucalyptus globulus* stands in Portugal. iForest, Vol. 6 pp. 217-227. DOI: 10.3832/ifor0821-006
Garcia-Gonzalo J., Zubizarreta-Gerendiain A., Ricardo A., **Marques S.,** Botequim B., Borges J. G., Oliveira M. M., Tomé M. and Pereira, J. M. C., 2012. Modelling wildfire risk in pure and mixed forest stands in Portugal. Allgemeine Forst und Jagdzeitung (AFJZ) – German Journal of Forest Research 183 (11/12), 238-248.
Garcia-Gonzalo J., **Marques S.,** Borges J. G., Botequim B., Oliveira M. M., Tomé J., Tomé M., 2011. A three-step approach to post-fire mortality modeling in Maritime pine (Pinus pinaster Ait.) stands for

enhanced forest planning in Portugal, Forestry 84 (2): 197-206. doi: 10.1093/forestry/cpr006. Available online at http://forestry.oxfordjournals.org/content/early/2011/03/22/forestry.cpr006.abstract

Table 0-5– Oral presentations and posters

Communications

Marques S., McDill M., Garcia-Gonzalo J., Borges J. Oliveira M., 2013. Forest Harvesting Model Integrating Wildfire Risk and Dispersion at the Landscape Level, SSAFR 2013, Aug 19-21, Quebec City, Canada

Marques S., Garcia-Gonzalo J., Borges J. G., Oliveira M. M., 2013. Caracterização, riscos e danos dos fogos florestais em Portugal, Física e os Aerossóis, 23 de Maio, Évora, Portugal

Marques S., McDill M., Borges J., 2013. A Stochastic, Cellular Integrating wildfire risk and dispersion. FORSYS 2013 – Decision support systems for sustainable forest management, Umea, Sweden, April 24-26.

McDill M., **Marques S.,** Borges J., 2012. Preventing orphan stands in spatially explicit forest management planning. INFORMS Annual Meeting 2012 Phoenix, Oct 14 – Oct 17

Marques S., McDill M., Borges J., 2012. A Stochastic, Cellular Multi-objective Forest Harvesting Model with the Risk of Fire, INFORMS Annual Meeting 2012 Phoenix, Oct 14 – Oct 17

Oliveira M. M., **Marques S.,** Botequim B, Garcia J., Borges J. G., Zubizarreta A., 2012. Characterization and modelization of forest fires in Portugal. Instituto Nacional de Pesquisas Espaciais-INPE. Ministério da Ciência, Tecnologia e Inovação. São José dos Campos. Brasil. August.

Cantarinha A., Mexia J. T., Oliveira M.M., Borges J. G., Moreira E., **Marques S.,** 2012. Multivariate collective models. In: Book of Abstracts of the Joint Meeting of y-BIS and jSPE, 23-26 Julho, Lisboa, Portugal, pp.148.

Moreira E., Cantarinha A., Mexia J. T., Oliveira M. M., Borges J. G., **Marques S**., 2012. Structured families of multivariate collective models. An application to forest fires in Portugal. In: Book of Abstracts of the Joint Meeting of y-BIS and jSPE, 23-26 Julho, Lisboa, Portugal, pp.146.

Garcia-Gonzalo J., Borges J. G., Palma J., **Marques S.,** Botequim B., Soares P., Tomé M., Pereira J. S., 2011. Assessing impacts of climate change in forested landscape planning with advanced decision support tools. A case study in Portugal. In proceedings of INFORMS Annual Meeting 2011 in Charlotte, North Carolina, United States of America, 12th to 16th November.

Marques S., Moreira F., Carreiras J., Oliveira M.M., Borges J., 2011. SIG, uma ferramenta para a caracterização dos fogos em Portugal, II Encontro de Sistemas de Informação Geográfica - Aplicações SIG em Recursos Agro-Florestais e Ambientais – 19 e 20 de Maio de 2011, Castelo Branco

Marques S., 2011. Assessing wildfire risk and damage probability on the Portuguese Forest. in Workshop Risk and multicriteria analysis. An application to natural resources management, 15th March, Universidade de Évora, Évora

Oliveira M. M., **Marques S.,** Botequim B., Borges J. G. and Garcia-Gonzalo J.. 2011. Characterization and modelization of forest fires in Portugal. 14th Symposium for System Analysis in Forest Resources, 8-11 March, 2011, Reñaca, Maitencillo, Chile

Natário I., Oliveira M.M., Carvalho L., **Marques S.,** Borges J., 2011. A Space-Time Model for Wild Fires in Portugal. Workshop on Bayesian Inference for Latent Gaussian Models with Applications. 2-5 January, University of Zurich. Zurich.

Marques S., 2010. Desenvolvimento de modelos de danos provocados por incêndios em povoamentos florestais. In: Workshop NOVTEC2010 - Novas Tecnologias em Gestão Florestal Sustentável IV - A gestão do risco de incêndio e a gestão da cadeia de valor. 25 e 26 de Outubro, ISA, UTL, Lisboa

Posters

Marques S., 2012. Integration of fire risk into forest stands management in Portugal. FORSYS Training School – Decision support systems for sustainable forest management, 17th to 21st September, Vienna, Austria.

Botequim B., Borges J. G., Silva A., **Marques S.,** 2010. Linkage between forest fire behavior and risk assessments in Portuguese Landscapes, Workshop on Decision Support Systems in Sustainable Forest Management – Experiences and Perspectives - DSFM, 19th 21th April, Lisbon, Portugal

Botequim B., Borges J. G., **Marques S.,** Mérida A., Silva A. 2010. Modeling fire behavior to assist forest management in Portuguese Landscapes, In: Book of Abstracts of the EFIMED Progress meeting and scientific seminar on "Knowledge base management of Mediterranean forest under climate driven risks: the ways ahead", 13 – 16 April, Antalya, Turkey

Marques S., Borges J. G., Botequim B., Cantarinha A., Carreiras J. M. B., Garcia-Gonzalo J., Moreira F., Oliveira M. M., Tomé M.,2009. Modeling fire incidence in Portugal In: Mediterranean Forests in the context of integrated management and land resources: soil, water and fodder, EFIMED Annual Progress Meeting and Scientific Seminar, 29th April-1st May, Marrakech, Morocco

CHAPTER I

GENERAL INTRODUCTION

Mainland Portugal extends over approximately 89,000 km2 located between 37ºN and 42ºN latitude and between 6ºW and 10ºW longitude. Forestry is a key element in the Portuguese landscape pattern and forests and woodlands extend over one-third of the country. Further, shrub lands and extend over about 32% of the country's area. Albeit ecological diversity as a result of climatic influences that range from Mediterranean to Atlantic or continental, over 89% of the forest area is occupied by five species: eucalypt (*Eucalyptus globulus*), cork oak (*Quercus suber*), Maritime pine (*Pinus pinaster*), holm oak (*Quercus rotundifolia*) and umbrella pine (*Pinus pinea*), occupying 26%, 23%, 23%, 11% and 6%, respectively (ICNF,2013).

Forest fires severity has increased substantially in the Mediterranean and in Portugal in the last decades (Alexandrian et al. 2000, Velez 2006, Pereira et al. 2006). In Portugal, burned area reached a total of about $3.89*10^6$ ha in the period from 1975 to 2007, i.e., equivalent to nearly 40% of the country area (Marques et al, 2011). Several factors may explain the ignition and spread of forest fires, such as fuel characteristics (Rothermel 1972, 1983; Albini 1976), climate, ignition sources, and topography (Agee 1993, Barton 1994, Viegas and Viegas 1994, Mermoz et al. 2005, Pereira et al. 2005). Fuel characteristics are a function of vegetation structure and composition in addition to anthropogenic factors. Topography, climate, and socioeconomic factors determine the mix available at any given site (Rothermel 1983, Cardille et al. 2001, Lloret et al. 2002, Badia-Perpinya and Pallares-Barbera 2006, Sebastián-López et al. 2008). Topography further affects fire behavior, via its direct influence on flame geometry and, indirectly, through its effect on weather (Rothermel 1983, Kushla and Ripple 1997). Climate, cover type, and topographical data are frequently used to develop fire risk indices (Pereira et al. 2005, Carreiras and Pereira 2006). Recent characterizations of forest fires in Portugal underlined the impact of climate variables, e.g. wind velocity, wind direction, the number of days with extreme fire hazard weather, on the number and size of fires (Viegas and Viegas 1994, Pereira et al. 2005, Gomes and Radovanovic 2008). Pereira et al. (2006) further claimed that more than 2/3 of the interannual variation of the area burned can be explained by changes in weather conditions. Other studies have analyzed the impact of species composition and of fuel reduction activities on fire intensity and spread (Fernandes 2001, Fernandes et al. 2005, Fernandes and Rigolot 2007). Wildfires have a substantial economic, social, and environmental impact and became a public calamity and an ecological disaster affecting a considerable area in Portugal (Gomes 2006). Forest managers and policy-makers thus face the challenge of developing effective fire prevention policies.

Fire risk models (e.g. Cumming 2001, Gonzalez et al. 2006) are key to explain fire occurrence probability both at the landscape level and according to stand's characteristics. The term risk has been defined in several ways in the natural hazard literature. It has been associated with the probability of occurrence of a natural hazard (González et al., 2006; Jactel et al., 2009). In the context of forest planning, risk has been defined as the expected loss due to a particular hazard over a given area and reference period (Gadow, 2000). We will refer to risk as the probability of wildfire propagation if there is an ignition point.

Many authors have studied the impact on wildfire risk of variables that are uncontrollable by forest managers such as weather variables (Andrews 1986, Velez 1990, Piñol et al 1998, Preisler et al. 2004, Preisler and Westeling 2005, Finney 2005, Chuvieco et al. 2009), physiographical variables (Stephens 1998, Schoenberg et al., 2003, Preisler et al. 2004, Finney 2005, Gonzalez et al. 2006, Carreiras and Pereira 2006, Marques et al,2011) and demographic and development variables (Carreiras and Pereira 2006, Quintanilha and Ho 2006, Chuvieco et al. 2009, Marques et al,2011). In Portugal, former studies are related with the characterization of wildfire ignition, using topographic and soil characteristics variables (Vasconcelos et al. 2001, Catry et al. 2009) and wildfire risk (Pereira & Santos 2003, Nunes et al. 2005, Carreiras &Pereira 2006, Catry et al. 2008, Marques et al. 2011a). However these models are not forest planning oriented.

Previous research suggests that de wildfire in Portugal is selective concerning different forestry cover types (Moreira et al. 2001, Nunes et al. 2005, Moreira et al. 2009, Silva et al. 2009). Further, several studies were developed to assess the wildfire risk with each forest cover type (Nunes et al. 2005, Godinho-Ferreira et al. 2005, Moreira et al. 2009, Silva et al. 2009). Recently, simple readily available biometric variables, shrub biomass load and socio-economic information have been used to model fire risk probability (Marques et al. 2012). Nevertheless, no modeling strategies to assess the impact of changes in controllable biometric variables on fire occurrence in maritime pine forests are available in

Portugal. This lack of information is a major obstacle to effective *Pinus pinaster* forest management planning in fire-prone regions. Yet no such models have been developed to explain fire occurrence probability in forest cover types in Portugal.

Fire damage models (e.g. Beverly and Martell 2003) are key to evaluate forest prescriptions and yet, again, no such models have been developed for forest cover types in this country. With this working plan when want to fill this lack, and development of fire risk and fire hazard models that may provide needed information for effective integration of the risk of fire in forest management in Portugal both at the stand and the landscape levels. Post-fire mortality has been studied using a variety of methods (e. g. Fowler and Sieg 2004; Sieg et al. 2006). Different methods have been used to model catastrophic disturbances, either by using fire behavior simulators (e.g. Finney 1998, 2006) or by using fire-damage descriptors that are based on measurements of tree tissue damage and use two main categories of readily observable indicators to assess tree mortality: crown damage and bole damage (Ryan 1982; Sieg et al. 2006; Keyser et al. 2006). However, these approaches require data such as tree tissue damage, fire intensity or specific meteorological conditions at the time of the fire event that are hard to predict over long planning periods (e.g. 20-60 years) (Rothermel 1991; Finney 1999; He and Mladenoff 1999; González et al. 2007; Garcia-Gonzalo et al. 2011a). The unavailability of this information constrains the applicability of both approaches in long-term forest management planning as they cannot effectively be used for predicting the long-term consequences of management alternatives (González et al. 2007). Many studies demonstrate that variables controllable by the manager (e.g. mean diameter, stand density) are related with fire damage (Linder et al. 1998; Pollet and Omi 2002; Hély et al. 2003; McHugh and Kolb 2003). Stand structure is related to fire intensity (Fernandes 2009), fire severity (Fernandes et al. 2010) and with damage/mortality (Agee and Skinner 2005; González et al. 2007; Garcia-Gonzalo et al. 2011a; Marques et al. 2011b). Furthermore, stand-level prescriptions provide the biological framework for managing the stands under fire risk conditions (Weaver 1943; Agee and Skinner 2005; Peterson et al. 2005; González et al. 2005a, 2007; Garcia-Gonzalo et al. 2011a; Marques et al. 2011b; Pasalodos-Tato et al. 2009, 2010).

Management may thus modify effectively stand conditions to control expected levels of post fire mortality (Pollet and Omi 2002; González et al. 2007; Fernandes et al. 2010; Garcia-Gonzalo et al 2011b). For example, studies reveal that stands with lower densities and higher tree diameters may decrease post fire mortality (González et al. 2007; Garcia-Gonzalo et al. 2011a; Marques et al. 2011b). Hence, post-fire mortality models oriented to forest planning (i.e. using predictor variables controllable by the manager) are a valuable forest management planning tool. They support effectively the design of silvicultural strategies (e.g. management alternatives) that may decrease the fire mortality (González-Olabarría et al. 2008; Pasalodos-Tato et al. 2009, 2010; Garcia-Gonzalo et al. 2011b; Ferreira et al. 2011, 2012). Moreover, they help to reduce the uncertainty by predicting the outcomes of different management alternatives (Gadow 2000).

Literature shows examples of the development and/or use of post-fire mortality models in forest planning (Peterson and Ryan 1986; Reinhardt et al. 1997; Reinhardt and Crookston 2003; González et al. 2007; Hyytianinen and Haight 2009). Originally, mortality models were mostly developed to serve as guidelines for timber salvage following fire, to be used for prescribed fires or to make post-fire management decisions (Ryan and Reinhardt 1988; Botelho et al. 1996; Reinhardt et al. 1997; Guinto et al. 1999; Rigolot 2004; Sieg et al. 2006). However, more recently models have been developed to address long term planning periods (i.e including explanatory variables easily obtainable from forest inventories without using tree tissue damage or detailed climatic data) to make pre-fire management decisions (González et al. 2007; Marques et al 2011b; Garcia-Gonzalo et al. 2011a).

Most of the post-fire mortality models developed in Portugal have addressed fire effects on pure maritime pine stands (e.g. Fernandes et al. 2004; Fernandes and Rigolot 2007; Fernández et al. 2008), pure eucalypt stands (Curtin 1966; Guinto et al. 1999) and cork oak covers (Catry et al. 2010). Only one has been developed for uneven-aged stands (Catry et al. 2010). Marques et al (2011b) and Garcia-Gonzalo et al. (2011a) have recently presented post-fire mortality models developed for pre-fire forest planning for pure eucalypt and maritime pine stands, respectively. However, no such models have been developed for stands with different structures and species compositions (e.g. pure, mixed, even and even-aged) in Portugal. These models would allow predicting the effect of changes in stand structure and species compositions on the expected mortality and therefore may be applied in forest management optimization systems.

The occurrence of stem death in a sample plot over a given period of time is a binomial outcome that may be modeled by logistic regression (Hosmer and Lemeshow 2000). These methods have been previously used to predict the probability of a single tree to survive or die due to fire (e.g. McHugh and Kolb 2003; Rigolot 2004; Kobziar et al. 2006; Sieg et al. 2006; González et al. 2007). When modeling tree mortality it is often the case that researchers find many plots where no post-fire mortality occurred (Monserud and Sterba 1999). Thus, if the data set for tree mortality modeling includes all plots, as in the case of traditional methods, the final models will always generate some mortality in all plots due to the binomial nature of the mortality event (Fridman and Stahl 2001; Álvarez González et al. 2004). Conversely, if only the plots where mortality has occurred are used, the model may overestimate the mortality rate (Eid and Oyen 2003). For this reason, recent studies have suggested the use of two or three step modeling methods (Woollons 1998, Fridman and Stahl 2001, Álvarez González et al. 2004). Some authors have previously used the three stage modeling technique in Portugal for pure species compositions (Marques et al. 2011b; Garcia-Gonzalo et al. 2011a).

This context suggests the need for the development of effective fire prevention policies. It further places a challenge to forest researchers and managers as they call for methods and tools that may help integrate forest and fire management planning activities currently carried out mostly independently of each other. Forest planning is characterized for the long-term nature of their outcomes. With such a long planning horizons many sources of uncertainty and risk are faced, i.e. regarding growth, market conditions, occurrence of catastrophic events, and behavior of the managers or preferences of the decision maker.

The economics of forest management under risk of disturbances have been analyzed in several studies, both at the stand level (Martell 1980; Gonzalez et al 2005a) and at the forest level (Lohmander 1987; González et al 2005b).

The development of stand management models with fire risk has involved an adaptation of the Faustmann framework to address stochastic events (e.g. Martell 1980, Reed 1984, Caulfield 1988, Englin et al. 2000, González et al. 2005a). The optimization of stand management has encompassed the definition of rotation age, thinning regime and regeneration treatment. Non-linear techniques have proved to be effective in addressing this problem. Yet there is little experience in developing models where treatments, namely fuel treatments, affect the level of fire risk. This Ph.D. working plan will enable expanding research on the use of quantitative techniques to address the risk of fire in stand management. Namely it will focus on models where fuel treatment scheduling impacts the risk of fire. In Portugal, there is some experience in developing dynamic programming stand-level management models for Maritime pine (e.g. Borges and Falcão 1999). Yet these models to not incorporate the risk of fire. This project will integrate existing stand and fuel growth and yield models (e.g. Falcão 1999, Tomé et al. 1998, Pereira et al. 2006) within a stand management optimization framework. It will further provide management models for integrating the risk of fire in stands of forest cover types in Portugal.

At a broader spatial scale, a forest serves a multitude of functions across a range of land uses that may include a populated wildland urban interface. Other fuel treatments such as fuel breaks should be considered. Further, in addition to fuel treatments, there are other activities such as the spatial layout of timber harvesting and road construction that affect fire size and intensity and should also be addressed. The landscape mosaic that results from forest management plans has been used to simulate and monitor fire behavior (e.g. Viegas et al. 1997, USDA 1999, Finney 2001) within a fire management framework. The development of decision trees (e.g. Cohan et al. 1983), mathematical programming models (e.g. Boychuck and Martell 1996) and heuristic methods (e.g. González et al. 2005b) has helped address fire risk considerations in landscape-wide forest management. Yet new approaches are needed to integrate effectively fire and forest management activities.

This Ph.D. will enable expanding research on the use of both mathematical programming models and heuristic methods to formulate and solve the integrated problem of determining fuel treatment, harvest scheduling, and road construction to optimize various objectives while sustaining effective fire prevention levels. Specifically, it will build upon state-of-the-art spatial analysis heuristics (e.g. Borges et al. 2002) and mathematical programming formulations (e.g. Martins et al. 2005) and it will involve the integration of simulation models for fire occurrence and spread with spatially explicit models for scheduling fuel treatments, road construction and maintenance and timber harvests.

Forest Management Decision Support System have been proved to be suitable platforms for the integration of information, models and methods required to solve complex forest management problems (e.g. Reynolds et al. 2005). Yet forest planning packages exist that do not always integrate state-of-the-art models and technology and are a source of confusion to forest managers wishing to apply the new technology (Rose, 1999). This project will enable the enhancement of current Portuguese technological applications (Borges et al, 2003) by integrating new information and new ecosystem management models, thus contributing for advanced and user-friendly decision support. Specifically, it will provide a decision support tool with new capabilities for addressing fire risk both at stand and landscape levels, for integrating forest and fire management and thus for effective fire prevention planning.

CHAPTER II

ASSESSING WILDFIRE RISK PROBABILITY IN PINUS PINASTER AIT. STANDS IN PORTUGAL

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Assessing wildfire occurrence probability in *Pinus pinaster* Ait. stands in Portugal

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Abstract

Maritime pine (*Pinus pinaster* Ait.) is an important conifer from the western Mediterranean Basin extending over 22% of the forest area in Portugal. In the last three decades nearly 4% of Maritime pine area has been burned by wildfires. Yet no wildfire occurrence probability models are available and forest and fire management planning activities are thus carried out mostly independently of each other. This paper presents research to address this gap. Specifically, it presents a model to assess wildfire occurrence probability in regular and pure Maritime pine stands in Portugal. Emphasis was in developing a model based on easily available inventory data so that it might be useful to forest managers. For that purpose, data from the last two Portuguese National Forest Inventories (NFI) and data from wildfire perimeters in the years from 1998 to 2004 and from 2006 to 2007 were used. A binary logistic regression model was build using biometric data from the NFI. Biometric data included indicators that might be changed by operations prescribed in forest planning. Results showed that the probability of wildfire occurrence in a stand increases in stand located at steeper slopes and with high shrubs load while it decreases with precipitation and with stand basal area. These results are instrumental for assessing the impact of forest management options on wildfire probability thus helping forest managers to reduce the risk of wildfires.

Key words: forest management; risk; fire occurrence model; *Pinus pinaster* Ait.

Resumen

Evaluación de la probabilidad de ocurrencia de fuegos en rodales de *Pinus pinaster* en Portugal

El articulo presenta un modelo para evaluar la probabilidad de ocurrencia de incendios en masas regulares y puras de *Pinus pinaster* en Portugal. Se desarrolla un modelo basado en datos de inventario fácilmente disponibles de tal forma que pueda ser una herramienta útil para los gestores forestales. Los datos proceden de los dos Inventarios Nacionales de Portugal (NFI) y de los datos de los parámetros de incendios forestales durante los años 1998-2004 y de 2006 a 2007. Se ha utilizado un modelo de regresión logística binarias utilizando datos biométricos del NFI. Los datos biométricos incluyen indicadores que puedan ser cambios en las operaciones prescritas en los planes forestales. Los resultados muestran que la probabilidad de ocurrencia de incendios en un rodal aumenta en rodales localizados en grandes pendientes y con una carga alta de matorrales, mientras que decrece con la precipitación y con el área basimétrica. Estos resultados son instrumentos para evaluar el impacto de las opciones de gestión forestal en la probabilidad de incendios ayudando por tanto a los gestores a reducir el riesgo de incendio.

Palabras clave: gestion forestal, riesgo, modelo de ocurrencia de incendios, *Pinus pinaster* Ait.

Introduction

In Portugal, nearly 40% of the country's territory was burned in the last three decades (Marques *et al.*, 2011). These wildfires had a substantial impact in the forested landscape configuration and composition. For example, the relative importance of the maritime pine area decreased from 30% to 22% of the total forest area in the period from 1995 to 2006 (DGRF, 2006). In the last ten years wildfires burned about 26,000 hectares,

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which is around 3.7% of total maritime pine stands (NIR, 2009). Maritime pine is still the most important timber producing species in Portugal and is mainly managed as even-aged plantations with a clear-cut harvest. Forest owners and forest managers need information that may help them develop management plans to minimize wildfire risk.

The term *risk* has been defined in several ways in the natural hazard literature. According to the definitions proposed by Hardy (2005), fire risk is defined as the chance that a fire might start in a context characterized by both natural and human causes (e.g. ignitions) and fire hazard as the potential fire behavior for a fuel type, regardless of the fuel type's weather-influenced fuel moisture content. González *et al.* (2006) proposed the term endogenous risk and Jactel *et al.* (2009) proposed the term hazard likelihood. We will refer to the term risk as the probability of a stand to be affected by a wildfire (i.e. probability of occurrence).

Many authors have studied the impact on wildfire risk of variables that are uncontrollable by forest managers such as weather variables (Chuvieco *et al.*, 2010; Durão *et al.*, 2010; Finney, 2005; Pereira *et al.*, 2005; Preisler *et al.*, 2004), physiographical variables (Carreiras and Pereira, 2006; Finney, 2005; González *et al.*, 2006; Marques *et al.*, 2011; Moreira *et al.*, 2009; Preisler *et al.*, 2004) and wildfire ignition in Portugal (Catry *et al.*, 2009; Vasconcelos *et al.*, 2001). However these models are not forest planning oriented. Yet the effectiveness of forest management depends on the availability of information about the impact on wildfire occurrence of biometric variables that are controllable by forest managers.

The forest cover type, the presence of multi-layered or young stands and the fuel load have a substantial impact on the probability of wildfire occurrence (Castro *et al.*, 2003; Ceccato *et al.*, 2002; Cumming, 2001; Reed, 1994; Velez, 1990). Modification of any of these fuel strata by silvicultural operations will thus have implications on wildfire occurrence (Jactel *et al.*, 2009; Peterson *et al.*, 2005). Thus in order to address wildfire risk, forest managers need information about the impact of "controllable" variables such as stand density, species composition, fuel availability at surface level (i.e. shrubs) and vertical structure of the stand on the probability of fire occurrence (Cumming, 2001; Finney, 2005). In this framework, González *et al.* (2006) developed a fire probability model for forest stands in Catalonia with biometric variables that may be readily available in order to include them in forest planning optimization to minimize risk (González-Olabarria *et al.*, 2008). In Portugal such models are not yet available and would help reverse current trends of maritime pine forestry.

The successful management of maritime pine in fire-prone regions is thus a challenging task that calls for the integration of wildfire risk in forest management planning. This research addresses this integration need by developing a management-oriented model (i.e. using easily measurable biometric variables) that may be able to predict the effects of management options (e.g. silvicultural treatments) on the probability of wildfire occurrence in pure and even-aged maritime pine stands.

Materials and methods

Materials

The assessment of wildfire risk probability in pine stands was based on historical fire information from 1998 to 2004 and 2006 to 2007. The fire data consisted of all perimeters of wildfires larger than 5 hectares, obtained by semi-automated classification of high-resolution remote sensing data (i.e., Landsat Multi-Spectral Scanner (MSS), Landsat Thematic Mapper (TM), and Landsat Enhanced TM+), that occurred in the two periods. In total 9,960 and 2,313 fire perimeters were identified in the first and in the second period, respectively. These wildfires burned over 150,000 ha.

The official wildfire database from the Portuguese Forest Service (AFN) that stores the starting coordinates (ignition) of wildfires was further used. For each year, a buffer around each ignition point was created with the minimum size needed to cover all burned plots in that year (Fig.1).

Biometric and environmental data considered for further analysis was acquired in 233 and 500 pure and even-aged maritime pine plots out of the 2,336 and 12,000 plots measured in the National Forest Inventories (NFI) carried out in the periods from 1995 to 1998 and 2005 to 2006, respectively. These plots were identified by overlapping NFI maps and wildfire perimeters. The status (burned/unburned) of each plot and the fire event were also recorded.

Our research extended the approach of González *et al.* (2006) in order to obtain an annual probability of wildfire. For that purpose an estimate of the bio-

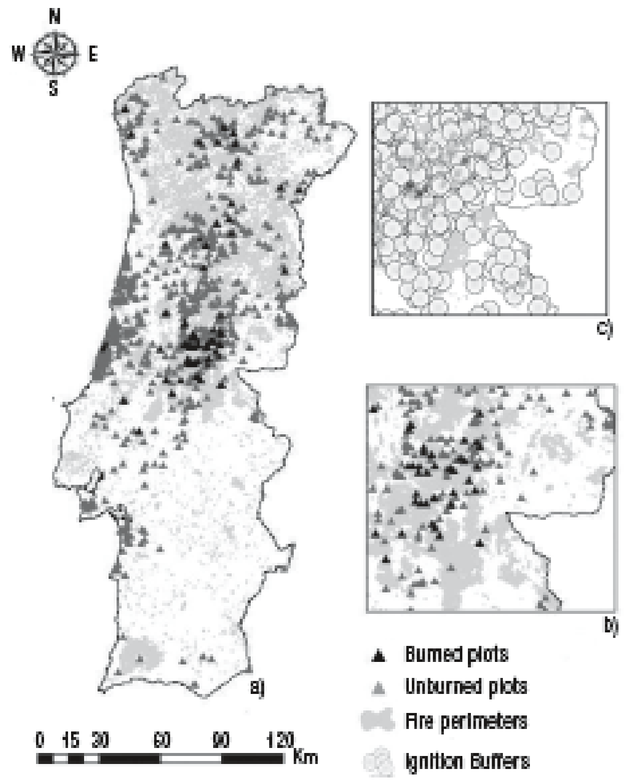

Figure 1. The map displays the fire perimeters occurred in Portugal during two periods: 1997-2004 and 2005-2007 and plots of pure / even-aged Maritime pine plots (a), a part of the data acquisition national forest inventory (NFI) plots used in the study (b), elimination of unburned plots due to the large distance to ignition points (c).

metric variables of each plot in each year in the period ranging from the inventory date to either the fire event date or the date of the next inventory was needed (Table 1). Thus, the stand-level growth and yield model, DUNAS (Falcão, 1997) was used to project Maritime pine growth and to estimate biometric variables in each plot. For modeling purposes a categorical variable was created for each observation and year with the value of 0 (fire did not occur) or 1 (fire did occur), (Table 1). If the stand burned a dichotomous variable (1) was assigned and the projection was stopped. If the stand did not burn, a value 0 was assigned and a projection done for the next year. As a consequence of the growth projections over time, one plot from the NFI resulted in several observations. The year 2005 was not included because we considered that projecting the forest growth over more than 6 years might lead to errors due to forest cover changes (e.g. harvests).

The maps with the buffers were overlaid with the maps with the Maritime pine observations (i.e. NFI plots estimated over time) (Fig. 1). Only observations within the ignition buffers were taken into account for modeling purposes. This methodology allowed us to eliminate observations that were not affected by a wildfire because there was no ignition point around. In total, 1945 observations estimated from the 733 NFI plots were used to fit the model, 66 of which were burned plots (Table 2).

Wildfire occurrence depends on further environmental variables (Catry *et al.*, 2008, 2009; Marques *et al.*, 2011; Wittenberg and Malkinson, 2009). The altitude of each plot was obtained from the country's Digital Terrain Model (DTM). The weather information was based on the same data from Tomé *et al.*

Table 1. Characterization of inventory plots in each year of the study period. The DUNAS growth and yield model (Falcão, 1997) was used to project all state variables in each 1998 NFI plot. If the stand burned a dichotomous variable (1) was assigned and the projection was stopped. If the stand did not burn, a value 0 was assigned and a projection done for the next year. Projections stopped in year 2004 as another inventory was available for year 2005

Inventory plot ID	Inventory 1998		Projection 1999		....		Projection 2004		Number of Observations per plot
	State Variables	Status	State Variables	Status	State Variables	Status	State Variables	Status	
1	X	0	X	1	...	...	–	–	2
2	X	0	X	0	...	...	X	1	7
3	X	0	X	0	...	...	X	1	7
4	X	0	X	0	...	...	X	0	7
5	X	1	X	–	...	...	–	–	1
6	X	0	X	0	...	...	X	0	7
7	X	0	X	0	...	...	X	0	7
...	...	...	...	...	...	...	...	...	...
233	X	0	X	1	...	...	X	–	2

...: indicates missing rows and columns. –: Indicates no projection done for that year.

As a first step, an analysis of the relationship of each individual independent variable with response variables was performed for a preliminary assessment of the relative importance of each variable on wildfire occurrence probability in maritime pine stands. The final multivariate model is obtained using stepwise regression on the training set combined with an understanding of the process of wildfire risk probability. Thus, the final model building considered ecological consistency, management relevance and its statistical significance (i.e. 0.05 significance level).

The different models were compared using the Akaike Information Criterion (AIC) (Burnham and Anderson, 2003; Silva *et al.*, 2009), and the one with lowest AIC considered the more parsimonious. Model performance was assessed through the likehood-ratio statistic (full model χ^2) and by calculating the area under the Receiver Operating Characteristics (ROC) curve (Hosmer and Lemeshow, 2000; Shapiro, 1999). For the multivariate model, Wald statistic test was also computed, for each selected variable. Thus the wildfire risk occurrence model in maritime pine stands was developed using a procedure that estimates the parameters of the logistic equation with maximum likelihood methods using PROC Logistic procedure of SAS 9.2 (SAS Institute, Cary, NC). Collinearity was assessed by adding new variables to the model and observing the effect on the slope coefficients and the estimated standard errors (Hosmer and Lemeshow, 2000).

The logistic model predicts a probability of an occurrence ranging continuously between 0 and 1. For certain applications a dichotomous variable is needed (e.g. burned or not burned) and a cut-point must be defined and compared to each estimated probability (Hosmer and Lemeshow, 2000). Different selection criteria have been proposed by some authors as Ryan and Reinhardt (1988), Hosmer and Lemeshow (2000), Monserud and Sterba (1999) and Neter and Maynes (1970). If the use of the model is to calculate the probability of wildfire occurrence a cut-point is not needed. However, we calculated a cut-point as an indicative value for other studies. This value was calculated using the Hosmer and Lemeshow method that consists in finding the value where the sensitivity curve and the specificity curve intersect. Classification classification rates (CCR) associated with different criteria to define cut-points also help select the best cut-point value.

Results

Fire probability model

The logistic model selected to predict wildfire occurrence is:

$$Pburn = \frac{1}{1 + e^{-(-2.0226 + 0.0204 \times Slp + 0.0597 \times SBiom - 0.0155 \times Prec - 0.5856 \frac{G}{dg})}} \text{ [Eq. 3]}$$

Where Slp is slope (degrees), SBiom is the total biomass of shrubs (Mg ha–1), Prec is the number of days with precipitation higher than 1 mm, G is the stand basal area (m^2 ha^{-1}) and dg is the quadratic mean diameter of the stand (cm). The predictor *G*/dg is non-linearly related to the number of trees per hectare (m^2 ha^{-1} cm^{-1}), it provides information about density and tree sizes.

All coefficients in Eq. 3 were significant, at least at the 0.05% level as judged by the Wald $\chi 2$ statistic (Hosmer and Lemeshow, 2000). The model predicted the right outcome (fire occurrence) in the case of 66.3% of the observations. The adequacy of the model was further assessed by the analysis of the ROC curve from the logistic model (area under the ROC curve of 0.677). Hosmer and Lemeshow goodness-of-fit test statistics were calculated and examining the partition in this test we can see that few models had low expected frequencies, thus suggesting that the p-values are accurate enough to support the hypothesis that the model fits. The assessment showed no collinearity among variables included in the model.

According to the equation 3 the model indicates that higher increase of slope and shrubs biomass increases the probability of a Maritime Pine stand to be burned. On the contrary the increase of precipitation and G/dg in a stand will decrease this probability (Fig. 2).

The odds ratio was further used to help interpret results, which is a more intuitive and easily understood way to capture the relationship between the independent and dependent variables. (Hosmer and Lemeshow, 2000; Kleinbaum, 1994). The odds ratio can be interpreted as the change in the odds for any increase of one unit in the parameter analyzed. However, the change in odds for some amount other than one unit is often of greater interest. Exponentiation of the parameter estimate(s) for the independent variable(s) in the model by the number c yields the odds ratio, where c is the increase in the corresponding independent variable.

From the analyses conducted it can be interpreted that an increase of 5 degrees in slope, would increase the probability of a stand to be burned in 1.107 times.

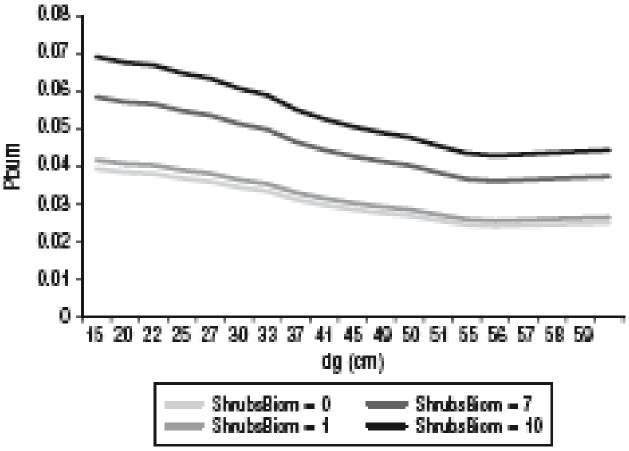

Figure 2. Effect of shrubs biomass and G/dg on the proportion of fire occurrence probability in stand with 93 days of precipitation per year.

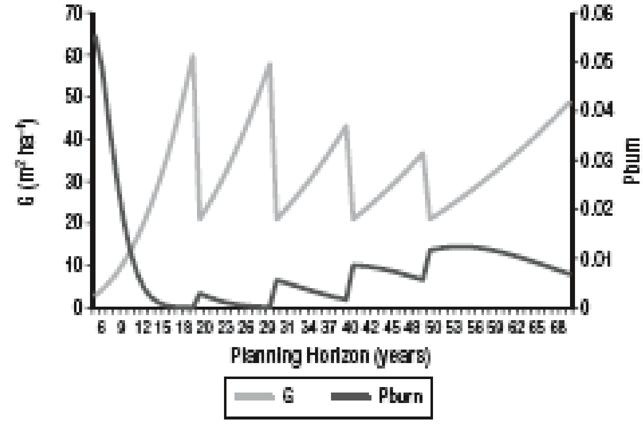

Figure 3. Effects of thinning and shrub cleanings on the probability of wildfire occurrence depending on a typical maritime pine stand located in central Portugal and assuming constant probability of ignition.

The chance of a stand to burn also increases in 1.062 times if the total biomass of shrubs increases one Mg per ha–1. On other hand, a increase if 1 one day of precipitation higher than 0,01 mm in a maritime pine stand, would decrease this probability in 0.985 times, but a increase in 5 days would decrease the risk to be burned in 0.9263 times. The odds for different combination of variables were checked for the predictor G/dg, being the effect of variation both variables (i.e. G and dg) analyzed. An increase in 8 cm of dg on a stand with 20 $m^2 ha^{-1}$ of G decreases 0.2313 times the fire hazard probability whereas an increase in 20 cm of dg for the same stand decreases fire probability in 0.5568 times. The effect of increasing G in 10 $m^2 ha^{-1}$ on a stand with a dg of 30 cm, would decrease the fire probability 0.8227 times, but an increase of 25 $m^2 ha^{-1}$ would decrease this probability 0.6139 times.

Because for some application there might be the need to transform the annual probability in a dichotomous variable (i.e. burns or does not burn), a cut-point was calculated (0.035). Using this value led to a CCR of 62.3% and the percentage of stands classified as having mortality was 31%. According to the chosen cut-point, the frequency table was calculated and from the analysis, this model predicts well 62.2% of the burned plots, and 37.8% of the unburned plots.

Application example

To evaluate the effects of potential management actions, the model was used to compute the probability of wildfire occurrence in one stand located in central Portugal with an elevation of 235 m, slope of 35°, with 85 days of precipitation per year and assuming constant probability of ignition. A typical maritime pine stand in Portugal development starts with 2000 trees per ha and typically four thinning are performed. Each thinning is accompanied by a shrub cleaning. Thinnings with removal of shrubs decrease fire probability. The one-year fire probability ranged from 0% to 5% in a pure even-aged maritime pine stand (Fig. 3).

Discussion and conclusions

Some studies have addressed the characterization of wildfires in Portugal (Carreiras *et al.*, 2006; Marques *et al.*, 2011; Nunes *et al.*, 2005; Pereira and Santos 2003) focusing on variables that are either uncontrollable by forest managers (e.g. climate, topography) or that may mostly support strategic decision-making (e.g. to support strategic zoning and regeneration decisions). Yet forest management requires further information. Namely, wildfire risk models are needed that may help foresters design prescriptions to decrease the probability of wildfire occurrence.

Our study addressed Portuguese conditions and the need to include biometric variables that are readily available to forest managers to develop wildfire occurrence models. Logistic regression was used to develop the wildfire occurrence model for pure and even-aged Maritime pine stands in Portugal. Contrarily to González *et al.* (2006) it was not assumed that biometric variables did not change in the period extending from the inventory date to the wildfire occurrence date. This study further extended former studies by pioneering the introduction of shrub biomass in a wildfire occurrence model and by using ignitions points.

Previous studies used logistic methods to predict wind and snow damage probability as a function of stand variables (Jalkanen and Mattila, 2000; Lohmander and Helles, 1987) and also to predict fire ignition probabilities (Catry *et al.*, 2009; Vanconcelos *et al.*, 2001), showing to be an appropriate technique to model events which occurrence is a binomial outcome (Silva *et al.*, 2009; Monserud and Sterba, 1999).

A data set encompassing even aged maritime pine plots located in 61 wildfire perimeters was used to develop and test 4,109 models so that all relevant combinations of explanatory variables might be addressed. The model selection process preferred models with good ecological behavior over models with purely good statistical fit. The model selected showed good ecological behavior and good goodness-of-fit. All its explanatory variables were statistically significant and have a relationship with variables normally used to explain potential fire behavior. Validation of the models was done through studies of the performance of the functions. No specific validation data sets were set-aside and later used for that purpose. This was for two main reasons. First, the relatively small number of observations in the dataset. Second, the best possible parameter estimates were of greater interest. There are advantages and disadvantages of splitting the data set for model validation purposes as discussed by Kozak and Kozak (2003). They concluded that cross validation by data splitting and double cross validation may provide little information in the process of evaluating regression models.

Our results show that annual probability of wildfire occurrence increases with the shrub biomass load. Maritime pine is a normally planted for timber in pure stands. The lack of management of these areas, related to socio-economic constraints, may be the origin of these results. Some studies show that the fire occurrence probability and severity will increase as the shrub layer become more conspicuous, substantially dryer and more flammable due to higher temperatures (Castro *et al.*, 2003; Fernandes *et al.*, 2010; Schmidt *et al.*, 2002).

Wildfire occurrence is also impacted by quadratic mean diameter and number of trees. The probability of wildfire occurrence decreases with basal area. The indicator G/dg is negatively related to wildfire occurrence probability indicating that higher densities reduce fire probability. This is in line with other studies where, for example, tree size parameters (i.e. quadratic mean diameter) and density parameters (basal area) have also been used as an indicator of stand-level competition and have been shown to influence fire risk probability in forest stands in Catalonia (González *et al.*, 2006). Dense tree canopies in conifer stands reduce the exposure of surface fuels to wind and solar radiation and minimize understory vegetation development, hence decreasing surface fire intensity and fire probability (Fernandes *et al.*, 2010). The application of our risk model using a typical silviculture for maritime pine stands shows a slight increase in fire risk after thinning. This is in line with common knowledge as thinnings may result in an increase of dead surface fuels (slash) that increase the risk of forest fires (Carey and Shumann, 2003). Thinnings may also help decrease the moisture in the forest due to the increased surface wind speed and light availability as well as the increased growth of herbs and shrubs (Fernandes and Rigolot, 2007; Fernandes *et al.*, 2010; Jactel *et al.*, 2009).

According to the proposed model, wildfire risk increases with slope. This result is in concordance with findings from several studies (Carreiras and Pereira, 2006; González *et al.*, 2006; Pereira *et al.*, 2006; Rothermel and Philpot, 1983; Silva *et al.*, 2009) that indicate that slope facilitates the initiation of passive crown fires (torching) as increases likelihood of flame length attaining the tree crown. Pereira and Santos, (2003), developed a wildfire risk map for Portugal showing that areas with steeper slopes are more prone to burn. Often these fires are not controlled adequately. Climatic variables, and stand location variables were tested in the modeling process, but none of them were finally included since they did not improve the model. This was unexpected result, since previous research showed and influence of, for instance climatic conditions (González and Pukkala, 2007; Preisler *et al.*, 2004).

In the framework of forest management planning, this model may be used to predict the probability of a wildfire to occur if there is an ignition. Thus, it should be applied after using a wildfire ignition model such as the ones developed by Catry *et al.* (2008, 2009) or Vasconcelos, *et al.* (2001). Further these models may be integrated with a growth and yield model which predicts the stand development over time (Hanewinkel *et al.*, 2010). At each step of the growth simulation if an ignition occurs the simulator estimates the probability of wildfire occurrence. Then depending on the approach followed to integrate fire in forest management planning this probability may be transformed into a dichotomous variable (e.g. wildfire occurs or does not occur). This would be the case of using fire spread simulators (e.g. González-Olabarria and Pukkala, 2011) or stochastic simulation where the estimated probability would be compared to the cut-point (González-Olabarria *et al.*, 2008). However, if only information on the probability of wildfire occurrence is required, no cut-point would be used. This would fit for example to approaches presented by Pasalodos-Tato *et al.* (2010) and Garcia-Gonzalo *et al.* (2011).

If the approach followed needs to calculate whether a wildfire occurs or not over the planning horizon a cut-point must be defined and compared to each estimated probability (Hosmer and Lemeshow, 2000). In this study we would recommend to use a cut-value of 0.035. Although the false positives are higher than using smaller cut-values, this threshold allows correct prediction of the non-fire events in our dataset. This means that this model would overestimate the fire events but we consider that is most important to predict well these stands that most likely would not get burnt due to their structural conditions.

The knowledge that results from this study may be instrumental to understand the influence of certain variables on the probability of wildfire occurrence. It provided valuable information to integrate risk considerations in both operational and strategic management planning. This information may be used to decrease fire hazard by promoting less fire-prone stands. Reduced wildfire risk can be included as an objective in forest planning problems by means of targeting fuel loads and stocking levels. Developing plans that include risk reduction as an objective may help managers address fire prevention issues in forest planning.

It is important to acknowledge that whether a fire may or not occur in a stand does not depend solely on stand endogenous variables. It further depends on landscape composition and structure (Fernandes *et al.*, 2010; González *et al.*, 2006). A study from Reed (1994) shows that stands are often burned by wildfires that started in neighboring stands (the probability of a stand burning being increased by other stands burning). Further research may expand the current model to consider for example other climate (e.g. wind speed, maximum temperature in the fire season) or landscape structure variables (e.g. neighboring stands biometric variables). Yet the proposed model may help forest managers design prescriptions to manipulate stand endogenous variables that impact the probability of wildfire occurrence. In addition, fuel treatments (i.e. reduction of fuels in forests) may change wildfire behavior and enhance the effectiveness of fire suppression tactics (e.g. Mercer *et al.*, 2008).

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CHAPTER III

DEVELOPING POST-FIRE *EUCALYPTUS GLOBULUS* STAND DAMAGE AND TREE MORTALITY MODELS FOR ENHANCED FOREST PLANNING IN PORTUGAL

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Developing Post-Fire *Eucalyptus globulus* Stand Damage and Tree Mortality Models for Enhanced Forest Planning in Portugal

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Marques, S., Garcia-Gonzalo, J., Borges, J.G., Botequim, B., Oliveira, M.M., Tomé, J. & Tomé, M. 2011. Developing post-fire *Eucalyptus globulus* stand damage and tree mortality models for enhanced forest planning in Portugal. Silva Fennica 45(1): 69–83.

Forest and fire management planning activities are carried out mostly independently of each other. This paper discusses research aiming at the development of methods and tools that can be used for enhanced integration of forest and fire management planning activities. Specifically, fire damage models were developed for *Eucalyptus globulus* Labill stands in Portugal. Models are based on easily measurable forest characteristics so that forest managers may predict post-fire mortality based on forest structure. For this purpose, biometric data and fire-damage descriptors from 2005/2006 National Forest Inventory plots and other sample plots within 2006, 2007 and 2008 fire areas were used. A three-step modelling strategy based on logistic regression methods was used. In the first step, a model was developed to predict whether mortality occurs after a wildfire in a eucalypt stand. In the second step the degree of damage caused by wildfires in stands where mortality occurs is quantified (i.e. percentage of mortality). In the third step this mortality is distributed among trees. Data from over 85 plots and 1648 trees were used for modeling purposes. The damage models show that relative damage increases with stand basal area. Tree level mortality models indicate that trees with high diameters, in dominant positions and located in regular stands are less prone to die when a wildfire occurs.

Keywords forest fires, forest management, *Eucalyptus globulus* Labill, damage model, post-fire mortality
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1 Introduction

Forest fire severity has increased substantially in the Mediterranean and in Portugal in the last decades (Alexandrian et al. 2000, Velez 2006, Pereira et al. 2006). In Portugal, since 1975 an average of 114000 hectares per year have been burned by wildfires. The need to address fire risk in forest management planning is evident and yet forest and fire management planning activities are currently carried out mostly independently of each other. In order to include fire risk in forest management planning several issues need to be addressed. For example, it is important for foresters to know which trees are likely to survive after a wildfire. What variables are important predictors of tree mortality?

A variety of methods have been used to study post-fire mortality (e. g. Fowler and Sieg 2004). Most of them have been used to predict which trees will survive a fire after the event has occurred. Further, post-fire tree survival models have been mainly used to study the effects of prescribed burning on trees (Ryan and Reinhardt 1988) or to give guidelines to post-fire salvage logging operations (Rigolot 2004).

These methods may be classified into direct and indirect approaches. Indirect approaches for prediction of tree mortality are based on fire behavior parameters. They require the use of fire behavior simulators (e.g. Finney 1998, 2006). These simulators need information about weather conditions and fuel accumulation. Nevertheless this information is hard to predict over long planning periods (Rothermel 1991, Finney 1999, He and Mladenoff 1999, González et al. 2007). On the other hand, direct approaches to predict mortality are based on measurements of tree tissue damage. Direct approaches use two main categories of readily observable indicators to assess tree mortality (Ryan 1982). The first, crown damage, considers all damage to the tree canopy e. g. both without foliage ignition (crown scorch) and with foliage ignition (crown consumption). The second, bole damage assesses the impact of wildfires on the cambium. However, tissue damage is a variable that can hardly be predicted in management planning contexts.

If post-fire models are to be used in forest planning, they must provide information about the impact on mortality of variables whose future value can be estimated with reasonable accuracy. Further, these variables should be under the control of forest managers. Thus mortality models should include variables such as forest density, species composition or mean diameter. It has been shown that variables such as these are related with fire damage (Linder et al. 1998, Pollet and Omi 2002, McHugh and Kolb 2003). Managers may modify effectively expected levels of fire damage by targeting specific values for these variables (Pollet and Omi 2002, Agee and Skinner 2005, González et al. 2007). In this context, post-fire models may be used to develop alternatives that reduce expected losses due to fire.

Nevertheless, the development and/or use of a mortality model in forest planning has been limited to few studies (Reinhardt and Crookston 2003, González et al. 2007, Hyytiäinen and Haight 2009) and none of them related to Portuguese conditions. In this context, this study aims at developing post-fire mortality models for *Eucalyptus globulus* Labill that may be used for generating optimal management plans taking into account fire risk. In fact, albeit ecological diversity as a result of climatic influences that range from Mediterranean to Atlantic or continental, over 80% of the forest area is occupied by four species: Maritime pine (*Pinus pinaster*), eucalypt (*Eucalyptus globulus*), cork oak (*Quercus suber*) and holm oak (*Quercus rotundifolia*) (Marques et al. 2011). Eucalypts are exotic to Portugal, having been introduced to the country in 1830, mainly for ornamental purposes (Fontes et al. 2006). Currently, eucalypt is the most important pulpwood producing species in Portugal. Eucalypt plantations extend over 647000 ha – about 20.6% of the total forest area in Portugal with a total yield of about 5.75 million m^3 per year (DGRF 2006). Eucalypt pulpwood is the key raw material of the pulp and paper industry.

Eucalypt is a highly flammable species. The bark catches fire easily. Deciduous bark streamers and lichen epiphytes tend to carry fire into the canopy and to disseminate it. Other features of eucalypt that promote fire spread include heavy litter fall, flammable oils in the foliage, and open crowns bearing pendulous branches, which encourages maximum updraft (Esser 1993). Nevertheless, despite the presence of volatile oils that produce a hot fire, leaves of eucalypt are classed as intermediate in their resistance to combustion, and juvenile leaves are highly resistant to flaming (Dickinson and Kirkpatrick 1985). However, eucalypt is seldom killed by fire (Esser 1993). Many authors have studied effects of fire on eucalypt; however, few studies have developed mortality models for eucalypt stands (Curtin 1966, Guinto et al. 1999).

The occurrence of stem death in a sample plot over a given period of time is a binomial outcome that may be modeled by logistic regression (Hosmer and Lemeshow 2000). These methods have been previously used to predict the probability of a single tree to survive or die due to different causes (Monserud and Sterba 1999, Guinto et al. 1999, McHugh and Kolb 2003, Rigolot 2004, Keyser et al. 2006, González et al. 2007). However, traditional modeling approaches generate mortality on all plots (Fridman and Ståhl 2001). Moreover, many studies predict the mortality rate without distributing mortality among trees in the stand.

When applying logistic models to predict mortality, both deterministic and stochastic approaches can be used (Monserud 1976, Monserud and Sterba 1999, Álvarez González et al. 2004). A deterministic method consists in the use of a threshold value within the interval 0–1; if the estimated probability of mortality exceeds the threshold value, the tree is assumed to die. A stochastic approach may encompass the drawing of a uniform random number in the interval 0–1; if the random number is lower than the estimated probability, the tree is assumed to die (González et al. 2007, Fridman and Ståhl 2001).

In this research, a three-step modeling strategy was used to develop the post-fire stand damage and tree mortality models (Woollons 1998, Fridman and Ståhl 2001, Álvarez González et al. 2004). Logistic regression methods were used in all three steps. In the first step, a model was developed to predict whether mortality occurs after a wildfire in a eucalypt stand. In the second step the degree of damage caused by wildfires in stands where mortality occurs is quantified (i.e. percentage of mortality). In the third step this mortality is distributed among trees. Data from over 85 plots and 1648 trees were used for modeling purposes. Models with good ecological behavior were preferred over models with purely good statistical fit.

2 Materials and Methods

2.1 Materials

The fire data used in this study consisted of wildfire areas of 2006 to 2008 in Portugal that were larger than 5 ha. Burned area mapping in 2006 to 2008 was obtained by automated classification of high-resolution remote sensing data (i.e., Landsat Multi-Spectral Scanner (MSS), Landsat Thematic Mapper (TM), and Landsat Enhanced TM+). In this period, about 125 thousand hectares burned in 3436 fire events. Data acquisition further encompassed the post-fire inventory of 85 plots in 2007 and 2008. 17 plots had been measured before the wildfire occurrence in the framework of the 2006 National Forest Inventory (NFI). These plots were identified by the overlay of NFI plots and fire areas using GIS tools (ArcGIS 9.2) (Fig. 1). In total, this analysis showed that 17 eucalypt plots out of the 12237 NFI plots were burned between 2006 and 2008. 68 additional burned plots in eucalyptus' stands were considered. These plots were measured in areas where the fire perimeter was known and trees had not been harvested. They were located all over the country and were inventoried (after the fire) at the same time as the burned NFI plots. The total 85 plots were located in 24 fires areas. In all these plots no trees had been harvested after the wildfire.

The post-fire inventory involved, in the case of all the 85 plots, both the measurement of biometric variables (e.g. height, diameter at breast height, burned stump height, burned canopy height, degree of stump destruction, fire damage) and the characterization of the plot (e.g. elevation, aspect, slope, presence of soil erosion, shrub species)(Table 1).

In the case of the 68 plots that had not been measured before the wildfire occurrence, reverse engineering was used to re-build the forest before the fire. In the case of plots with standing burned trees, pre-fire diameter dbh was assumed to be unaffected by fire. The equation developed by Soares and Tomé (2002), was used to estimate pre-fire height:

$$h = h_d\left(1+\left(0.10694+0.02916\frac{N}{100}-0.00176d_{max}\right)e^{0.0354h_d}\right)\left(1-e^{-1.81117\frac{dbh}{h_d}}\right) \quad (1)$$

where h_d is the dominant height (m), N is the stand density (number of trees per hectare), d_{max} is the maximum tree diameter in the stand (cm) and dbh is the tree diameter at breast height (cm).

When at inventory date, the burned trees were broken or tissue was damaged making impossible to measure dbh, the stump diameters were measured and tree characteristics were obtained using reverse engineering (McClure 1968, Bylin 1982, Diéguez Aranda et al. 2003). For this purpose, an equation was adjusted using a 3966 eucalypt trees' dataset to predict the dbh for eucalyptus with R^2 of 0.9517:

$$dbh = -0.5207 + 0.82841d_{stump} \quad (2)$$

where dbh is the tree diameter at breast height (cm) and d_{stump} is the stump diameter (cm). Once dbh was estimated, an equation developed for Eucalyptus (Tomé et al. 2007) was used to calculate the tree height:

$$h = \frac{dbh}{0.6733 + 0.0130dbh} \quad (3)$$

where dbh is the tree diameter at breast height (cm). Then, using dbh and tree height at the moment of inventory, the tree pre-fire height was estimated as before using Eq. 1.

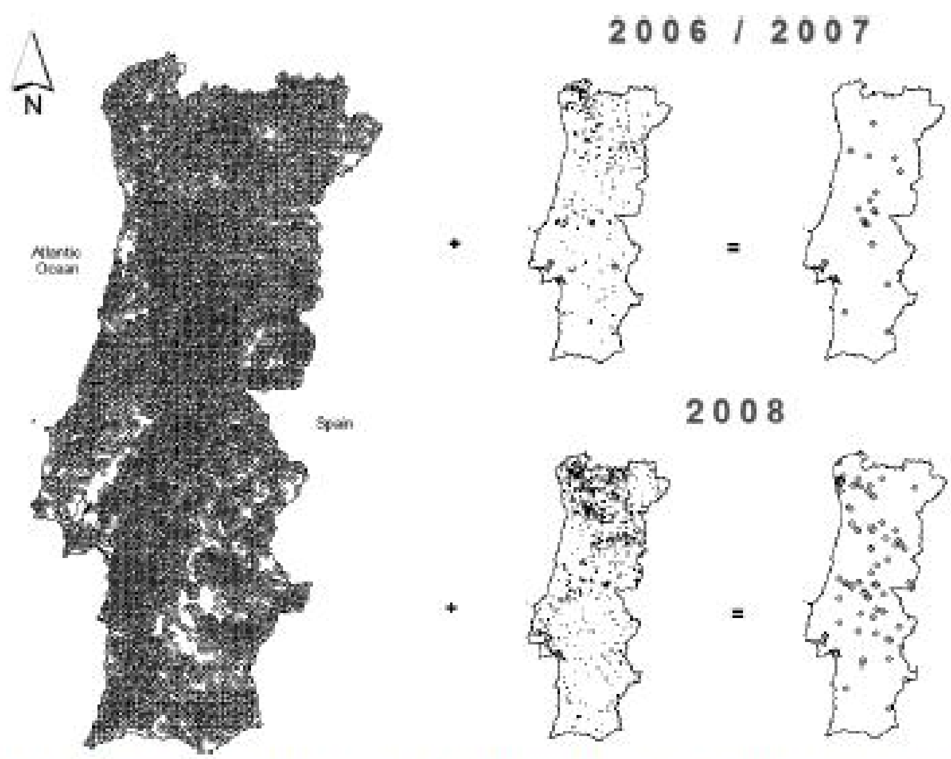

Fig. 1. Locating inventory plots for data acquisition. The map on the left shows the national forest inventory plots (≈12 200); the maps on top-right show the fire areas in 2006–2007 and the 43 burnt eucalypt plots; the maps on bottom-right, show the fire areas in 2008 and the 49 burnt eucalypt plots.

Table 1. Descriptive statistics for stand and tree level data. G is stand basal area (m^2 ha^{-1}); dg is the quadratic mean diameter (cm); N, number of trees; Pdead, proportion of dead trees in the stand; Sd standard deviation of tree diameters and Sh, standard deviation of tree heights of the stand; G/dg is non-linearly related to the number of trees per hectare. The predictor Sd/dg expresses the relative variability of tree diameters. dbh is de tree diameter at breast height (cm); g is the tree basal area (m^2 ha^{-1}); dbh/dg and g/G are competition indexes.

Variable	Stands without dead trees=44				Stands with dead trees=41			
	Max	Mean	Min	Sd	Max	Mean	Min	Sd
Altitude	491	186.05	0	140.48	644	212.96	0	126.41
Slope	32	13.12	0.60	8.67	27.80	12.63	0	6.54
Aspect	350	139.16	0	105.99	350	154.81	0	106.46
N	1811	574	20	444	1459	565	20	379
G	29.73	7.46	0.08	7.19	26.92	5.95	0.27	5.19
dg	36.51	12.78	7	5.38	26.00	12.18	4.79	4.54
Sd	10.76	3.06	0	2.30	12.30	3.72	0	2.22
Sh	5.14	2.05	0	1.34	7.06	2.90	0	1.94
G/dg	1.75	0.56	0.02	0.44	1.77	0.50	0.04	0.33
Sd/dg	0.45	0.24	0.01	0.12	0.81	0.34	0.01	0.20
Pdead	0	0	0	0	1	0.77	0.03	0.28
Variable	**Alive trees=877**				**Dead trees=771**			
	Max	Mean	Min	Sd	Max	Mean	Min	Sd
dbh	59.30	12.49	5.20	5.77	46.30	11.00	4.66	4.38
g	0.28	0.01	0.002	0.02	0.17	0.01	0.002	0.01
H	32.17	15.70	6.50	4.37	30.81	14.07	6.88	4.08
dg	3516.49	189.14	27.04	240.34	2143.69	140.13	21.69	140.68
BAL	1.03	0.22	0.00	0.23	1.34	0.29	0	0.29
dbh/dg	2.92	0.99	0.35	0.29	2.49	1.01	0.28	0.34
g/G	0.05	0.002	0.0001	0.004	0.05	0.00	0.0001	0.003

2.2 Methods

2.2.1 *Modelling Mortality with Logistic Regression (General Approach)*

The occurrence of stem death in a sample plot over a given period of time is a binomial outcome that may be modeled by logistic regression. Moreover, the logistic function is mathematically flexible, easy to use, and has a meaningful interpretation (Hosmer and Lemeshow 2000). The logistic model predicts a probability of an occurrence ranging continuously between 0 and 1. The dependent variable is dichotomous (e.g. death or no death). A cut-point may be defined and compared to each estimated probability (Hosmer and Lemeshow 2000) in order to assign '1' to the event of death and a '0' to the no death event. The logistic regression model may be presented as:

$$p = \frac{1}{1 + e^{-(\beta_0 + \beta_1 x_1 + \cdots \beta_n x_n)}} \quad (4)$$

where the variable p is a measure of the total contribution of all the independent variables used in the model, x_1 to x_n are independent variables, β_0 is the intercept and β_1 to β_n, are estimated parameters or regression coefficients.

The logistic function was used to model stand-level damage and tree-mortality caused by wildfires. The Proc logistic procedure of SAS 9.1 (SAS Institute, Cary, NC) that estimates the parameters of the logistic equation with maximum likelihood method was used in all three steps of the proposed approach to develop the post-fire stand damage and tree mortality models. The information obtained from applying the stepwise variable selection method was combined with an understanding of the process of mortality.

2.2.2 *Predicting Whether Mortality Will Occur in a Stand after a Wildfire*

In order to predict whether mortality will occur in a stand after a wildfire, a stand-level binary variable was created. This variable takes the value '1' if death occurs and the value '0' if no death occurs in a stand. This modeling approach thus provides information to filter the stands where some mortality would occur out of the whole set of stands that also includes those where all the trees survive. A number of stand-level variables (e.g. plot characteristics, biometric variables (Table 1)) were used for estimating the probability of mortality occurrence. Model building considered both the ecological consistency of predictors (i.e. signs of coefficients that are biologically reasonable) and its statistical significance (i.e. 0.05 significance level and no systematic errors in the residuals).

2.2.3 *Estimating Stand-Level Damage Caused by a Wildfire*

In order to quantify mortality caused by wildfires in stands where mortality did occur, two stand-level variables were created. These variables indicated the number of trees that died as a consequence of a wildfire (i.e. number of events) and the total number of trees in the stand (i.e. number of trials). Then SAS Proc logistic procedure used these numbers to fit the logistic regression. The average proportion of dead trees in stands where mortality occurred as a consequence of wildfires was 40% in the case of eucalypt stands.

A number of stand-level variables related to topography (e.g. slope), biometric variables (e.g. mean diameter) and structure (e.g. standard deviation of tree heights) were used for estimating the probability of stand-level mortality caused by a wildfire (Table 1). All predictors had to be logical and significant at the 0.05 level without any systematic errors in the residuals.

2.2.4 *Estimating Post-Fire Tree Mortality*

We tried to find the best fitting and biologically reasonable model to describe the relationship between the response variable i.e. the tree status (alive or dead), and a set of explanatory variables (Table 1). For modeling purposes, a tree-level binary categorical variable was created. This variable takes the value '1' if death occurs, and a '0' if the tree survives.

As this is a two-stage model, a variable indicating the proportion of dead trees in the stand (Pdead) estimated with the stand level model (section 2.2.3) – was used to predict the post fire tree mortality. Therefore only trees present in stands where mortality was predicted were used to fit the tree mortality model. In total, 942 eucalypt trees were inventoried in burned plots, of which 771 were present in stands where mortality was predicted. Further predictors were selected by testing whether they improved the model. Selection considered the importance of the variable in terms of forest inventory and management as well as its simplicity, its ecological consistency and its statistical significance (i.e. 0.05 significance level and no systematic errors in the residuals). The "receiver operating characteristic (ROC)" curve was further used to test the model sensitivity. The ROC curve plots the probability of detecting true signal (sensitivity) and false signal (specificity) over all possible threshold values of the marker.

3 Results

The logistic model to predict whether mortality will occur in a eucalypt (Eq. 5) stand is:

$$\mathrm{Psd} = \frac{1}{1+e^{-(\beta_0+\beta_1\frac{\mathrm{Sd}}{dg})}} \qquad (5)$$

where Psd is the probability of stand death to occur, Sd is the standard deviation of trees' diameters at breast height (cm), dg is the quadratic mean diameter (cm) of trees. The predictor Sd/dg expresses the relative variability of tree diameters.

The model indicates that higher values of Sd/dg (i.e. variability of tree diameters) increase the probability of death to occur in the stand (Eq. 5). All model coefficients were significant, at least at the 0.05% level as judged by the Wald χ^2 sta-

Table 2. Parameter estimates, standard errors (SE), Wald χ^2 statistics and p-values for the model predicting whether mortality will occur in a stand (Eq. 5).

Effect	Estimate	SE	Wald χ^2	$p>x^2$
β_0	−1.1742	0.4716	6.1994	0.0128
β_1	3.8942	1.4944	6.7906	0.0092

Table 3. Parameter estimates, standard errors (SE), Wald χ^2 statistics and p-values for the model predicting degree of damage caused by a wildfire (Eq. 6).

Effect	Estimate	SE	Wald χ^2	$p>x^2$
β_0	0.4654	0.0495	88.5417	<0.0001
β_1	0.00119	0.000133	88.4201	<0.0001
β_2	0.0214	0.00278	59.2655	<0.0001
β_3	0.00401	0.00520	59.3581	<0.0001
β_4	−0.1027	0.0103	100.3593	<0.0001

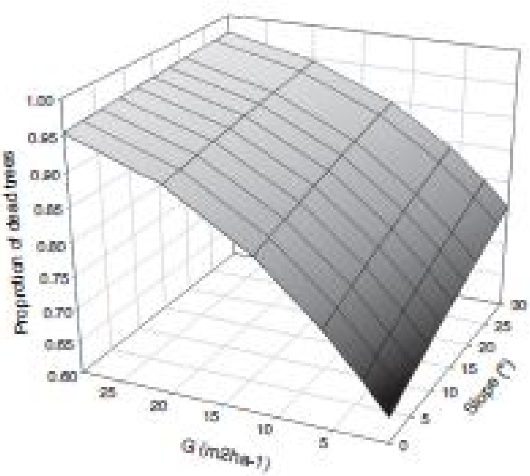

Fig. 2. Effect of stand basal area (G) and slope on the degree of damage in the stand, i.e. the proportion of dead trees (Eq. 6). The values were calculated with an altitude = 200 m and Sd = 3.47 cm.

tistic (Hosmer and Lemeshow 2000) (Table 2). The model was successful in predicting whether mortality did occur after the wildfire in 63% of eucalypt stands (i.e. percentage of concordant pairs).

The model to quantify mortality caused by wildfires in eucalypt (Eq. 6) stands where mortality did occur is:

$$\text{Pdead} = \frac{1}{1+e^{-(\beta_0+\beta_1 Alt+\beta_2 Slope+\beta_3 G+\beta_4 Sd)}} \qquad (6)$$

where Pdead gives the proportion of dead trees in the stand, Alt is altitude (meters), Slope is measured in (°), G is the stand basal area (m^{2}ha^{-1}) and Sd is the standard deviation of the diameter of trees (cm).

All coefficients in Eq. 6 were significant, at least at the 0.05% level as judged by the Wald χ^2 statistic (Hosmer and Lemeshow 2000) (Table 3), while 65% of the data were successfully identified by the model as to whether death had or had not occurred for eucalypt. Collinearity was assessed by adding new variables to the model and observing the effect on the slope coefficients and the estimated standard errors (Hosmer and Lemeshow 2000). This assessment showed no collinearity among variables included in the model.

The model indicates that the proportion of dead trees increases when stand basal area increases (Fig. 2). Moreover, steep slopes and higher altitudes contribute to increase this proportion (Eq. 6).

A tree-level mortality model predicting the probability of a tree to die due to a forest fire was developed:

$$\text{Ptm} = \frac{1}{1+e^{-(\beta_0+\beta_1 dbh+\beta_2 G+\beta_3 Sh)}} \qquad (7)$$

where Ptm is the probability of a tree to die, dbh is the tree diameter at breast height (cm), G is the stand basal area (m^{2}ha^{-1}) and Sh is the standard deviation of the tree heights in the stand (cm). The higher the value of this variable the more irregular the stand is.

All coefficients in Eq. 7 were significant, at least at the $p<0.05$ level (Table 4) as judged by the Wald χ^2 statistic (Hosmer and Lemeshow 2000). The model predicted the right outcome (death after the wildfire) in the case of 79.6 % of inventoried dead trees. The area under the ROC curve (0.798; Fig. 3) indicates excellent discrimination (Hosmer and Lemeshow 2000), thus showing that the selected model performs well.

Table 4. Parameter estimates, standard errors (SE), Wald χ^2 statistics and p-values for the tree-model predicting the probability of a tree to die due to a forest fire (Eq. 7).

Effect	Estimate	SE	Wald χ^2	$p>\chi^2$
β_0	3.6381	0.321	128.4757	<0.0001
β_1	–0.217	0.0251	74.8686	<0.0001
β_2	–0.1747	0.0312	31.4322	<0.0001
β_3	0.4311	0.0698	38.1911	<0.0001

The model (Eq. 7) indicates that trees with high diameters are less prone to die when a wildfire occurs (Fig. 4). Trees in stands with higher basal area have also lower mortality probability. Moreover, trees located in stands with higher variability in tree heights (Sh) are expected to have higher mortality probability (Fig. 5).

The odds ratio was further used to help interpret results as it provides an intuitive and easily understood way to capture the relationship between the independent and dependent variables. (Hosmer and Lemeshow 2000, Kleinbaum 1994). The odds ratio gives the increase or decrease in probability that a unit change in the independent variable has in the probability that the event of interest will occur. However, the change in odds for some amount other than one unit is often of greater interest. Exponentiation of the parameter estimate(s) for the independent variable(s) in the model by the number c yields the odds ratio, where c is the increase in the corresponding independent variable.

Results show that a 5 $m^2 ha^{-1}$ increase in stand basal area has an odd ratio of 0.418 which means that the probability of a tree to die would decrease by 58.2%. An increase in 5 cm in the dbh of the tree has an odd ratio of 0.338 which means that probability of death would decrease by 66.2%. The effect of an increase in one unit in height standard deviation would increase the probability in 53.9% (i.e. odd ratio of 1.539), which means that variability in tree sizes have a high impact on the probability of tree mortality.

Cut-point was calculated for the model predicting whether mortality will occur in a stand (Table 5). If the value that maximizes the correct classification rate (CCR=63.5%) was used as

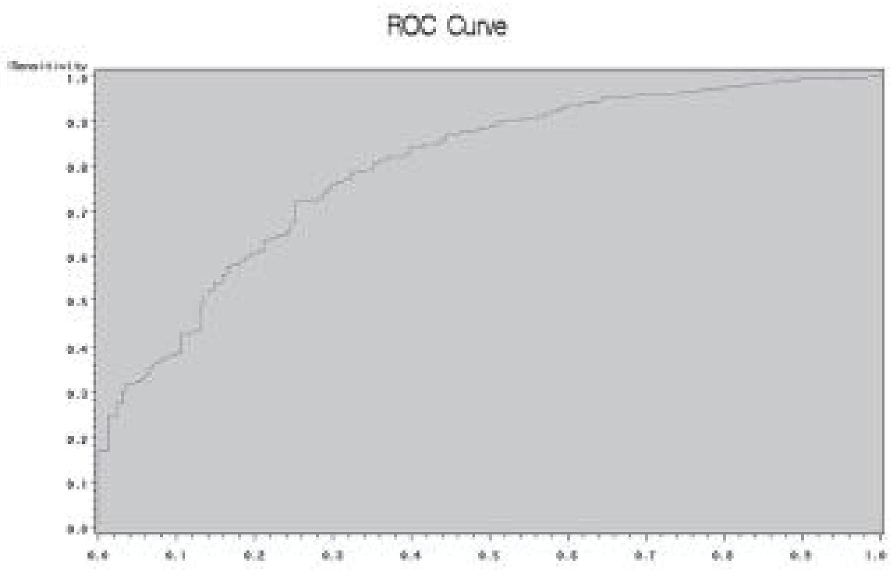

Fig. 3. ROC curve for Eucalypt tree-mortality model, showing an acceptable discrimination between tree mortality occurrence and non-occurrence (0.798) Models with ROC values 0.7 are considered to have an acceptable discrimination, ROC values 0.8 have excellent discrimination, and ROC values 0.9 are considered to have outstanding discrimination

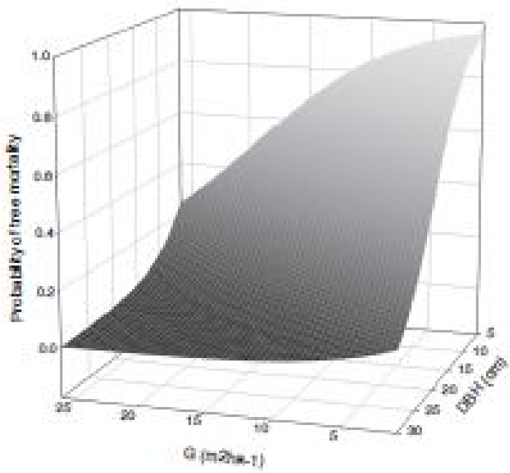

Fig. 4. Effect of tree-diameter (dbh) and basal area (G) on the probability of tree mortality using Eq. 7. The values were calculated using Sh=2 which is the mean value in our dataset.

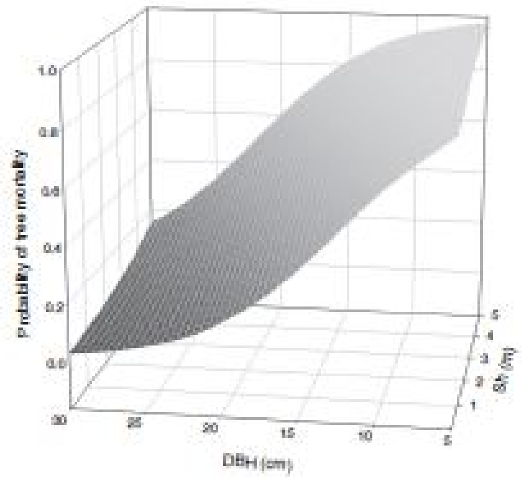

Fig. 5. Effect of tree-diameter (DBH) and standard deviation of the tree height (Sh) on the probability of tree mortality using Eq. 7. The values were calculated using G=6 which is the mean value in our dataset.

Table 5. Prediction parameters depending on the cut-points used to transform a continuous probability into a 0–1 dichotomous value predicting whether there is mortality in a stand or not.

Cut point	CCR (%)	Sensitivity (%)	Speficitity (%)	False dead (%)	False alive (%)	Classified as dead (%)	Classified as alive (%)
0.41	52.9	75.6	31.8	49.2	41.7	72	28
0.42	58.8	73.2	45.5	44.4	35.5	64	36
0.43	58.8	70.7	47.7	44.2	36.4	61	39
0.44	55.3	63.4	47.7	46.9	41.7	58	42
0.45	52.9	53.7	52.3	48.8	45.2	51	49
0.46	55.3	51.2	59.1	46.2	43.5	46	54
0.47	52.9	46.3	59.1	48.6	45.8	44	56
0.48	51.8	43.9	59.1	50	46.9	42	58
0.49	52.9	43.9	61.4	48.6	46	41	59
0.5	57.6	43.9	70.5	41.9	42.6	36	64
0.51	56.5	36.6	75	42.3	44.1	31	69
0.52	57.6	36.6	77.3	40	43.3	29	71
0.59	58.8	26.8	88.6	31.3	43.5	19	81
0.6	60	26.8	90.9	26.7	42.9	18	82
0.61	58.8	24.4	90.9	28.6	43.7	16	84
0.62	58.8	24.4	90.9	28.6	43.7	16	84
0.63	60	24.4	93.2	23.1	43.1	15	85
0.64	61.2	24.4	95.5	16.7	42.5	14	86
0.65	62.4	24.4	97.7	9.1	41.9	13	87
0.66	63.5	24.4	100	0	41.3	12	88

CCR, Correct Classification Rate 63.5. The percentage of observed plots where tree mortality occurred was 48%. Sensitivity indicates the percentage of predictions where true signal was predicted correctly (i.e. stands with some mortality) and specificity refers to false signal (i.e. stands not showing mortality were correctly predicted).

criteria as suggested by Ryan (1997), the cut-point would be 0.66. According to this value, mortality would occur in 12% of the plots while mortality did actually occur only in 48%. This cut point did not show any false positive (i.e. stands that did not have any dead trees but were classified as if mortality had occurred) but 40% were false negatives (i.e. stands that had dead trees but where classified as if mortality had not occurred). Thus another criterion was tested to select the cut-point: the value where the sensitivity and the specificity curves cross. It provided a 0.45 cut-point value. Using this value led to a CCR of 59% and the percentage of stands classified as having mortality was 50%. If still another criteria was used e.g. the average observed percentage of event occurrence as suggested by Monseroud and Sterba (1999), the cut point value would be the same (i.e. 0.45). Using this cut-point, in 45% of the stands classified as not having mortality, some trees had actually died (i.e. false alive). If a cut point of 0.43 was chosen in 36% of the stands classified as not having mortality, some trees had actually died (i.e. false alive) and 70% of the stands showing mortality were well classified.

In the case of the individual tree mortality model, the cut-point value where the sensitivity and the specificity curves intersect was 0.83 (CCR of 73%). This would result in a sensitivity of 73% and specificity of 70%.

4 Discussion and Conclusions

A variety of methods have been used to study post-fire mortality (e. g. Fowler and Sieg 2004). Most of them have been used to predict which trees will survive a fire after the event has occurred. Further, post-fire tree survival models have been mainly used to study the effects of prescribed burning on trees (Ryan and Reinhardt 1988) or to provide guidelines to post-fire salvage logging operations (Rigolot 2004). Yet the use of these methods in forest management planning is constrained by its cost-effectiveness. Further, variables used to predict post-fire mortality (e.g. weather conditions, tissue damage) are seldom available.

Logistic regression has been used earlier for predicting tree-mortality as a consequence of prescribed fire (Botelho et al. 1996, Linder et al. 1998) and wildfire (Regelbrugge and Conard 1993, Harrington 1993, Stephens and Finney 2002, Beverly and Martell 2003, McHugh and Kolb 2003, Rigolot 2004, González et al. 2007). Yet, these models used variables that are seldom available for long-term forest planning. This explains why few studies have used post-fire mortality models in forest planning (Reinhardt and Crookston 2003, González et al. 2007, Hyytiäinen and Haight 2009). González et al. (2007) demonstrated the potential for the development and use of a damage model within a forest planning context. This model did not use tissue injury indicators or direct fire behavior parameters. Yet no management planning friendly damage models were available for Eucalypt stands in Portugal.

The proposed logistic modeling approach overcomes these obstacles and provides mortality models that may be readily used in stand or forest-level management planning e.g. mortality models that do not depend on direct descriptors of fire damage that are never available within a management planning framework. The proposed model rather provides information about the impact on mortality of variables whose future value may be estimated with reasonable accuracy. Further, these variables are under the control of forest manager (Pollet and Omi 2002, Agee and Skinner 2005, González et al. 2007). In this context, post-fire models may be used to develop alternatives that reduce expected losses due to fire. This model may be also used in post-fire mortality assessment when tree damage is no more visible or when trees have been cut in post fire salvage operations.

The advantage of the three-step methodology used in this study when compared to other traditional approaches is that it enables the identification of stands where no mortality occurs. Traditional models always generate some mortality for all plots (Fridman and Ståhl 2001). This research confirmed the potential of the proposed approach to develop mortality models that may be used in forest planning (Reinhardt and Crookston 2003, González et al. 2007, Hyytiäinen and Haight 2009).

The proposed approach used a large dataset encompassing 1858 trees in 92 plots located in 24 fire areas in Portugal. As fire areas are inventoried after each fire season by the Portuguese public administration, these 24 fire areas may result from more than one wildfire. Model fitting quality as assessed by concordance and area under the ROC curve suggests that the model has a good ability to discriminate post-fire mortality in eucalyptus stands in Portugal. In the framework of forest management planning, Eq. 5 may be used to predict whether mortality may occur in a stand after a wildfire. If mortality is predicted to occur, Eq. 6 may be used to estimate the degree of damage in the stand, i.e. the proportion of dead trees. Finally, the mortality at stand level may be then distributed among trees using Eq. 7. As these models are developed to support management planning, Eq. 6 may be used to estimate the number of trees that will die in the stand (i.e. percentage of trees), after a wildfire (if mortality indeed occurs). Equation 7 may then be used to predict the probability of mortality of each tree in the stand and to build a list of all trees in the stand ordered according to this probability (trees with higher probability of mortality are ranked first in the list). The management planning model may then select the trees that will be assumed to die for planning purposes by going down the list and stopping when it reaches the number of trees that are estimated to die (from Eq. 5). For this reason no threshold value is needed to transform the estimated probability into a dichotomous variable (e.g. death or no death). An illustrative example is given: assume Eq. 6 indicates that 50% of the trees in one stand will die if a wildfire occurs, assume we have 300 trees in the stand, then Eq. 7 would be used to calculate each tree probability to die and the 150 trees with higher probability to die will be selected and classified as dead for planning purposes. As suggested by González et al. (2007) the tree mortality equations can be used to generate mortality variation if a stochastic component corresponding to the residual variation of the stand level damage model is added to the prediction.

Prediction and classification do not follow the same pattern, so a compromise must be reached between good classification and good prediction of mortality when choosing a threshold level (cut-point) (Crecente-Campo et al. 2009). In our study, a cut-point of 0.43 for the model predicting whether mortality occur in a stand (Eq 5) seems appropriate. In the model predicting probability of a tree to die, a threshold value is not needed, however if a cut value of 0.83 is used, the sensitivity and specificity would be 73 and 71%, respectively.

Biometric variables selected for estimating post-fire mortality included the tree diameter (dbh), the relative variability of tree diameters (Sd/dg, Sd and Sh), and stand basal area (G). Other significant variables were related to fire behavior (i.e. slope) and stand location (i.e. altitude). In the stand-level damage model, steeper slopes increase the expected mortality. This is in concordance with other studies and may be explained by an easier transfer of heat uphill (Agee 1993, González et al. 2007, Hyytiäinen and Haight 2009). In our case, altitude correlates positively with the degree of mortality in burned areas. This is because most of the burned stands were located in high altitudes.

Eucalypt models indicate that in even-aged stands with higher tree diameters, fire damage is expected to be lower than in irregular stands with trees with smaller dimensions. This confirms results presented by Guinto et al. (1999) who found that eucalypts' resistance to fire was highly correlated with the thickness and the extent of protective bark tissue on the stem. These generally increase with the size of the individual. If bark is sufficiently thick eucalyptus trees may even survive also crown fires (Gill 1977). Moreover, the eucalypt mortality models also showed that in stands with larger trees, fire damage is expected to be lower. These results are in concordance with findings from other studies (Guinto et al. 1999, Pollet and Omi 2002, González et al. 2007). Extensive model testing led to the rejection of other biometric variables as predictors of stand-level damage after a wildfire.

At tree level, tree diameter (dbh) was found to be negatively related with tree mortality. In addition, trees located in stands with high tree height variability had higher probability of dying after a wildfire event. This is because irregular structures may facilitate crown fires. The combination of dbh and tree height variability indicates that dominant trees (e.g. trees with high diameters located in irregular stands) have lower probability of dying. This is in concordance with other studies (Ryan and Reinhardt 1988, Linder et al. 1998, González et al. 2007). This finding is also coherent with other studies (Monserud and Sterba 1999, Van Mantgem et al. 2003, González et al. 2005), dominant trees experiencing less competitive stress than smaller ones. The use of prescribed burning to reduce the potential wildfire intensity in European forests has been acknowledged by several authors (e.g. Vega et al. 1994, Mutch and Cook 1996). Moreover, when planning prescribed fire, sound prescriptions are required to constrain tree damage and mortality to acceptable levels (Botelho et al. 1996). Our results may help understand the effects of stand structure and tree-size on mortality and may thus help to define prescribed burns.

When no pre-fire inventory was available, reverse engineering was needed to reconstruct the stand. Thus the quality of the models is dependent on the quality of the equations used for that purpose. Further, this research considered mortality within a fixed period after the wildfire (i.e. each plot was measured only once one year after the wildfire event). In some cases, this may lead to an underestimation of mortality caused by the wildfire. Yet in doing so we avoided the situation where stands might have been harvested after the wildfire leading to a loss of data needed for the development of the model. Nevertheless, the development of the first post-fire mortality models in Portugal took into account all available data and information. No evaluation data were available, therefore model evaluation was made with the fitting data. It is never easy to select the best way to validate a model. The authors are aware of the advantages and disadvantages of splitting the data set for model validation purposes well discussed for instance in Kozak and Kozak (2003).

Fire damage models (e.g. Beverly and Martell 2003, González et al. 2007) are key to evaluate forest prescriptions and yet, again, no such models have been developed for eucalypt stands in Portugal. This research encompassed the development of post-fire *Eucalyptus globulus* stand damage and tree mortality models for enhanced forest planning in Portugal. They provide information about the impact of forest fires under alternative forest conditions.

These models are management-oriented; they provide information needed to quantify the effect of different management options on the expected fire damage thus further providing a more realistic estimation of future incomes. These models are instrumental to designing silvicultural strategies that may decrease the damage caused by wildfires. For example, developing silvicultural strategies at stand level aiming to maintain stands with lower densities and high tree diameters, performing earlier and heavier low thinnings may decrease post fire mortality.

The characteristics of the models provide opportunities for several applications e. g. integration of fire risk into forest management planning either at stand level (e.g. González et al. 2005a, 2007, 2008, Ferreira et al. Submitted) or at landscape level (González et al. 2005b). These models can easily be implemented in decision support systems that may allow the manager to minimize the expected losses due to wildfires when developing management plans.

The usefulness of post fire models in forest planning depends on the information they may provide about the impact on mortality of variables whose future value may be estimated with reasonable accuracy. Eucalypt post-fire stand damage and tree mortality models are based on variables that are under the control of forest managers (e.g. forest density, mean diameter). Thus we may further conclude that they can be used to integrate effectively fire risk into forest management planning.

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CHAPTER IV

A THREE-STEP APPROACH TO POST-FIRE MORTALITY MODELING IN MARITIME PINE (*PINUS PINASTER* AIT.) STANDS FOR ENHANCED FOREST PLANNING IN PORTUGAL

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A three-step approach to post-fire mortality modelling in maritime pine (*Pinus pinaster* Ait) stands for enhanced forest planning in Portugal

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Summary

Maritime pine (*Pinus pinaster* Ait) is a very important timber-producing species in Portugal with a yield of ~67.1 million m^3 $year^{-1}$. It covers ~22.6 per cent of the forest area (710.6×10^3 ha). Fire is the most significant threat to maritime pine plantations. This paper discusses research aiming at the development of post-fire mortality models for *P. pinaster* Ait stands in Portugal that can be used for enhanced integration of forest and fire management planning activities. Post-fire mortality was modelled using biometric and fire data from 2005/2006 National Forest Inventory plots and other sample plots within 2006–2008 fire perimeters. A three-step modelling strategy based on logistic regression methods was used. Firstly, the probability of mortality to occur after a wildfire in a stand is predicted and secondly, the degree of mortality caused by a wildfire on stands where mortality occurs is quantified. Thirdly, mortality is distributed among trees. The models are based on easily measurable tree characteristics so that forest managers may predict post-fire mortality based on forest structure. The models show that relative mortality decreases when average d.b.h. increases, while slope and tree size diversity increase the mortality.

Introduction

Post-fire mortality has been studied using a variety of methods (e.g. Fowler and Sieg 2004; Sieg *et al.*, 2006) that may be classified into two main groups. The first includes indirect approaches for prediction of tree mortality based on fire behaviour parameters. The second includes direct approaches based on the measurements of tree tissue injury (Keyser *et al.*, 2006; Sieg *et al.*, 2006). Indirect approaches require the use of fire behaviour simulators (e.g. Finney, 1998, 2006) which include models to calculate fire rate of spread (Rothermel, 1972; Albini, 1976; Rothermel and Rinehart 1983), fire shape (Anderson, 1983; Alexander, 1985), spot fire distance (Albini 1979, 1983) and crown fire spread rate (Van Wagner, 1977; Rothermel, 1991). However, these systems are seldom implemented in stand level simulators because information about weather conditions in a specific fire ignition day, fuel moisture (e.g. 1- and 10-h fuel moisture contents) and fuel accumulation (e.g. shrubs growth, deadwood) are necessary and hard to predict over long planning periods, e.g. 60 years (Rothermel, 1991; Finney, 1999; He and Mladenoff, 1999; González *et al.*, 2007). On the other hand, direct approaches require measurements of tree tissue injury and fire intensity. These methods can be used for a variety of situations, e.g. setting acceptable upper and lower fuel moistures for conducting prescribed burns, determining number of hectares that may be burned on a given day and developing timber salvage guidelines following fire (Reinhardt, 1997). Yet direct methods are hardly practical in a forest management planning context as they require input data that are not available to forest managers when developing forest plans.

The usefulness of post-fire models in forest planning depends on the information these models may provide about the impact on mortality of variables whose future value may be estimated with reasonable accuracy and are under the control of forest managers through management (e.g. forest stand density, species composition, mean diameter). Many studies demonstrate the relationships between these variables and post-fire mortality (Pollet and Omi, 2002; Hély *et al.*, 2003; McHugh and Kolb, 2003). Stand structure is related to fire intensity (Fernandes, 2009), fire severity (Fernandes *et al.*, 2010) and with damage/mortality (Agee and Skinner, 2005; González *et al.*, 2007). The amount of shrubs biomass may further increase fire severity. However, information about the evolution of forest fuels and/or shrubs over planning periods longer than 5–10 years is limited.

Stand-level prescriptions provide the biological framework for fire activity and damage (Weaver, 1943; Agee and Skinner, 2005; Peterson *et al.*, 2005; González *et al.*, 2005, 2007). Management may thus effectively modify stand conditions to control expected levels of fire damage (Pollet and Omi, 2002; González *et al.*, 2007; Fernandes *et al.*, 2010). Thus, the use of post-fire models oriented to forest planning, i.e. using predictor variables controllable by the manager, may help anticipate the outcomes of different management alternatives, thus reducing uncertainty (Gadow, 2000). It also helps to identify management alternatives that reduce the expected losses due to fire.

Many studies have addressed fire effects on maritime pine (*Pinus pinaster* Ait) stands. Some of them concentrated on fire ecology (e.g. Fernandes and Rigolot, 2007) and fire behaviour (Fernandes *et al.*, 2004). Other analysed the influence of fire severity on the recruitment of maritime pine (e.g. Martínez *et al.*, 2002; Fernández *et al.*, 2008). Further studies have been focused on competition-induced mortality or drought-induced mortality (Martínez-Vilalta and Piñol, 2002). Botelho *et al.* (1996) and Botelho *et al.* (1998) presented a mortality model for prescribed fires in maritime pine stands in Portugal. Basically, the existing mortality models have been mostly developed to serve as guidelines for timber salvage following fire or to be used for prescribed fires or to make post-fire management decisions (Botelho *et al.*, 1996; Reinhardt, 1997; Rigolot, 2004; Sieg *et al.*, 2006). Nevertheless, the development and/or use of a post-fire mortality model in forest planning have not attracted much attention. Few studies have used or developed post-fire mortality models in forest planning (Peterson and Ryan, 1986; Ryan and Reinhardt, 1988; Reinhardt *et al.*, 1997; Reinhardt and Crookston, 2003; González *et al.*, 2007; Hyytiäinen and Haight, 2009). González *et al.* (2007) further considered its application within a forest planning context without using tissue injury indicators neither direct fire behaviour parameters. Yet no such models have been developed for maritime pine stands in Portugal, even though maritime pine covers ~22.6 per cent of the forest cover, totalling 710.6×10^3 ha with a yield of ~67.1 million m^3 $year^{-1}$ (DGRF, 2006) and that 48 per cent of the forested area in Portugal that burned in the 1990s consisted of pure maritime pine stands (Pereira and Santos, 2003).

In this context, this study aims at developing post-fire mortality models for maritime pine that may be used for generating optimal management plans taking into account fire. The occurrence of tree death in a sample plot over a given period of time is a binomial outcome that may be modelled by logistic regression (Hosmer and Lemeshow, 2000). Logistic regression methods have been previously used to predict the probability of a single tree to survive or die due to different causes (Regelbrugge and Conard, 1993; Botelho *et al.*, 1996; Rigolot, 2004; Keyser *et al.*, 2006; Eisenbies *et al.*, 2007; González *et al.*, 2007).

In this research, a three-step modelling strategy was used to develop the post-fire stand damage and tree mortality models (Woollons, 1998; Fridman and Stahl, 2001; Álvarez González *et al.*, 2004). The three-step approach consists of (1) estimating whether mortality occurs in a stand after wildfire, (2) quantifying the degree of damage in terms of proportion of dead trees in the stand and (3) estimating the probability of mortality of a tree after a wildfire which serves to distribute the mortality among individual trees. Logistic regression was used in all three steps. Data from over 124 plots and 1174 trees were used for modelling purposes. Models with good ecological behaviour were preferred over models with purely good statistical fit.

Materials and methods

Materials

The fire data used in this study consisted of perimeters of 2006–2008 wildfires in Portugal that were larger than 5 ha. Burned area mapping in 2006–2008 was obtained by automated classification of high-resolution remote sensing data (i.e. Landsat Thematic Mapper (TM) and Landsat Enhanced TM+). In this period, ~125 000 ha burned in 3436 fire events. Data acquisition further encompassed the collection of the 2006 National Forest Inventory (NFI) plots. By the overlay of NFI plots and fire perimeters using GIS tools (ArcGIS 9.2), it was possible to identity plots that had been measured before the wildfire occurrence. This analysis showed that 18 maritime pine plots of the 12 237 NFI plots were burned between 2006 and 2008. In the same period, 106 additional maritime pine burned plots were considered. These plots were measured in areas where the fire perimeter was known and trees had not been harvested. They were located all over the country and were inventoried (after the fire) at the same time as the burned NFI plots. In total, data acquisition encompassed the post-fire inventory of 124 plots from 2007 to 2009. In all these plots, no trees had been harvested after the wildfire.

The post-fire inventory involved, in the case of all 124 plots, both the measurement of biometric variables for trees with diameter larger than 7.5 cm (e.g. height, diameter at breast height, bole char height, crown killed height) and the characterization of the plot (e.g. elevation, aspect, slope, presence of soil erosion, shrubs species). However, because the objective of the model was to predict fire mortality if a fire occurs over long planning horizons (i.e. over 60 years), biometric variables tested for the model were limited to easily measurable tree and stand characteristics, which permit the forest manager to predict the effect of stand structure and species composition on the expected mortality (Table 1).

In the case of plots that had not been measured before the wildfire occurrence, regression models were used to reconstruct the forest before the fire. Pre-fire d.b.h. of standing burned trees was assumed to be unaffected by fire and pre-fire height was estimated using an equation developed by Tomé *et al.* (2007) for maritime pine (equation 1).

$$h = 0.0795\left(1+\left(0.0795+0.211\frac{N}{100}-e^{0.0254h_d}\right)\right)\cdot\left(1-e^{-1.1658\frac{\mathrm{DBH}}{h_d}}\right), \quad (1)$$

where d.b.h. is the tree diameter at breast height (centimetre), N is the stand density (number of trees per hectare) and h_d is the dominant height (metre).

Methods

Modelling mortality with logistic regression (general approach)

The occurrence of stem death in a sample plot over a given period of time is a binomial outcome that may be modelled by logistic regression (Hosmer and Lemeshow, 2000). Moreover, the logistic function is mathematically flexible, easy to use and has a meaningful interpretation (Hosmer and Lemeshow, 2000). The logistic model predicts a probability of an occurrence ranging continuously between 0 and 1. The dependent variable is dichotomous (e.g. death or no death). The logistic regression model may be presented as:

$$Y = \frac{1}{1+e^{-(\beta_0+\beta_1 x_1+\ldots+\beta_p x_p)}}, \quad (2)$$

where Y is the dependent variable (dichotomous), x_1 to x_p are independent variables, β_0 is the intercept and β_1 to β_p, are parameters.

Table 1: Descriptive statistics for variables tested as model predictors at stand level

	Stand level							
	Stands without dead trees = 31				Stands with dead trees = 93			
Variable	Max	Min	Average	SD	Max	Min	Average	SD
Altitude (m)	931	0	324.80	298.75	940	0	344.98	193.62
Slope (°)	27	0.60	12.64	6.10	32	0	13.13	7.71
avgDBH (cm)	34	5.36	17.31	7.70	29.33	4.6	13.55	5.94
N (tree/ha)	578	20	142.83	135.27	1539	20	278.06	295.82
G (m^2 ha^{-1})	21.36	0.08	4.73	5.84	38.15	0.08	7.03	8.35
Dg (cm)	37.14	7	18.40	7.92	32.69	7	16.34	6.91
Avgh (m)	19	5.30	11.77	4.06	25.75	3.47	12.82	5.88
sd (cm)	17.26	0	5.094	4.69	17.67	0	4.70	3.86
sh (m)	6.20	0	1.70	1.63	8.41	0	1.85	1.56
G/Dg	0.84	0.02	0.22	0.23	1.73	0.02	0.38	0.38
Sd/Dg	0.64	0.01	0.25	0.18	0.69	0.01	0.26	0.14
Pd (%)	0	0	0	0	0.99	0.05	0.82	0.31
Ndead (tree/ha)	0	0	0	0	1537	6	213.34	259.91

	Tree level							
	Live trees = 234				Dead trees = 940			
Variable	Max	Min	Avg	SD	Max	Min	Avg	SD
DBH (cm)	45.50	7	19.30	8.66	43.50	7.00	14.71	7.56
h (m)	28.10	3.44	13.98	4.71	23.60	3.80	11.30	4.12
g (m^2 ha^{-1})	0.16	0.00	0.04	0.03	0.15	0.00	0.02	0.02
BAL (m^2 ha^{-1})	5.17	0.00	1.20	1.15	5.40	0.00	1.35	1.27
Dg (cm)	207.03	4.90	44.73	41.18	189.23	4.90	27.36	29.01
DBH/Dg	2.21	0.24	1.03	0.33	2.22	0.33	0.94	0.27
g/G	0.01	0.00	0.01	0.01	0.11	0.00	0.01	0.01

G is stand basal area; Dg is the quadratic mean diameter; N, number of trees per ha; Pd, proportion of dead trees in the stand; Ndead, number of dead trees per ha; avgDBH, mean tree diameter of the stand; avgh is the average tree height; SD, standard deviation of tree diameters and Sh, standard deviation of tree heights of the trees in the stand; G/Dg is a density measure related to the number of trees per hectare. The predictor Sd/Dg expresses the relative variability of tree diameters. Altitude is measured in metres and slope is measured in degrees; DBH is the tree diameter at breast height; *h* is the tree height; *g* is basal area of the tree; BAL is the basal area of the trees higher than the studied tree, DBH/Dg and g/G are competition indexes. Max, maximum; min, minimum; Avg, average.

Models to predict stand-level damage and tree-mortality caused by wildfires were developed using the logistic procedure of SAS 9.1 (SAS Institute, Cary, NC). This procedure estimates the parameters of the logistic equation with maximum likelihood methods.

An analysis of the relationships between each individual independent variable and response variables was performed for a preliminary assessment of the relative importance of each variable on post-fire damage and tree mortality. The final multivariate model was obtained by testing all possible combinations of variables. If the resulting mortality model is not biologically correct, it cannot be expected to perform well outside the data range (Hamilton, 1986; Crecente-campo *et al.*, 2009). Thus, model building considered ecological consistency of predictors (i.e. signs of coefficients), importance of the variable in terms of forest inventory and management as well as its simplicity and its statistical performance and significance (e.g. 0.05 significance level, receiver operations characteristic (ROC) parameters, index of concordance and correct classification rate (CCR)). Collinearity was assessed by adding new variables in the model and observing the effect to the slope coefficients and estimated standard errors (Hosmer and Lemeshow, 2000).

Standard tests and statistics for logistic regression, namely the likelihood ratio test and Wald's test, were used. Hosmer—Lemeshow goodness-of-fit statistics and ROCcurve analysis from the logistic model were also used (Hosmer and Lemeshow, 2000). The ROC curve plots the probability of detecting true signal (sensitivity) and false signal (specificity) over all possible cut-points. To evaluate the discriminatory ability of a cut-point, it is common to summarize the information of the ROC curve into a single global value or index (e.g. area under the ROC curve). Models with area under ROC curve values higher than 0.7 are considered to provide an acceptable discrimination between wildfire occurrence and non-occurrence (Hosmer and Lemeshow, 2000). The concordance analysis procedure was further used to help interpret results (Kleinbaum, 1994; Hosmer and Lemeshow, 2000).

A way to summarize the results of a fitted logistic regression model is to use a classification table. This is a result of cross-classifying the outcome variable (e.g. death occurrence) with a dichotomous variable whose values are derived from the estimated logistic probabilities (Hosmer and Lemeshow, 2000). The logistic model predicts a probability of an occurrence ranging continuously between 0 and 1. Thus to obtain this dichotomous variable (e.g. death or no death), a cut-point must be defined and compared to each estimated probability (Hosmer and Lemeshow, 2000). Different selection criteria have been proposed, e.g. the average observed survival rate of the dataset and the value that maximizes the sum of sensitivity and specificity (Monserud and Sterba, 1999; Crecente-Campo *et al.*, 2009).

In this study, three different criteria were used to define the cut-point: (1) the value that maximizes the CCR (e.g. Ryan, 1997), (2) the value where the sensitivity curve and the specificity curve cross each other (Hosmer and Lemeshow, 2000) and (3) the average observed percentage of event occurrence in the original data (Monserud and Sterba, 1999). Tables with classification error rates associated with different criteria to define cut-points were constructed to help select the best cut-point value. Due to the relatively small number of plots, no specific dataset was set aside for evaluation. Thus, evaluation of the model was done calculating ROC curves and classification tables for the fitting dataset.

Modelling whether mortality will occur in a stand after a wildfire

In order to predict whether mortality will occur in a stand if a wildfire occurs, a stand-level binary variable was created. This variable takes the value '1' if mortality occurs within the stand (mortality of trees bigger than 7.5 cm) and the value '0' if no death occurs. Thus, this model would filter the stands where some mortality would occur from those where all the trees survive. A number of stand-level features (e.g. site conditions, biometric variables) were tested (Table 1). The dataset showed that mortality had occurred in 75 per cent of burned stands (93 of 124 stands).

Estimating stand-level mortality caused by a wildfire

In stands where mortality did occur (93 over 124 stands), two stand-level variables were created; the number of trees that died after fire (i.e. number of events) and the total number of trees in the stand (i.e. number of trials). Then SAS logistic procedure used these numbers to fit the logistic regression. This model would quantify mortality caused by a wildfire in terms of proportion of dead trees in the stand. The average proportion of dead trees in stands where mortality occurred was 80 per cent (940 dead trees of 1174) (Table 1). A number of stand-level variables related to topography, biometric variables and structure were tested (Table 1).

Estimating post-fire individual tree mortality

The predicted variable was the probability of a tree to die. For modelling purposes, a tree-level binary categorical variable was created. This variable takes the value '1' if death occurs, and a '0' if the tree survives.

As this is a two-stage model, a variable indicating the proportion of dead trees in the stand (Pd) predicted with the stand-level model (estimating stand-level mortality caused by a wildfire) was tested as a predictor. For this reason, only trees present in stands where mortality was predicted were used to fit the tree mortality model (i.e. 940 trees). Further predictors were selected by testing whether they improved the model (Table 1).

Results

The logistic model to predict the probability of mortality occurring in a stand if fire occurs is

$$\text{StandMort} = \frac{1}{1+e^{-(2.1231+2.3943\frac{G}{\text{Dg}})-0.1134\,\text{avgDBH})}}, \quad (3)$$

where StandMort is the probability of tree death to occur in the stand (i.e. it differentiates the stands where all the trees survive from the stands where some or all the trees die), G is the basal area (square metre per hectare) and Dg is the quadratic mean diameter (centimetre) of trees. The predictor G/Dg is a density measure and avgDBH is the average diameter at breast height (centimetre). Higher densities contribute to a higher probability of death to occur in a stand, whereas this probability decreases with higher average diameter at breast height (see equation 3 and Table 2). The model was successful in predicting whether mortality did occur after the wildfire in 73.8 per cent of stands (i.e. percentage of concordant pairs). The area under the ROC curve (0.74) indicated good discrimination (Hosmer and Lemeshow, 2000).

The model to quantify stand-level mortality caused by wildfires where mortality did occur has the following form:

$$\text{Pd} = \frac{1}{1+e^{-(0.7065+0.00491\,\text{Alt}+0.1158\,\text{Slope}-0.1649\,\text{avgDBH}+0.1456\,\text{Sh})}}, \quad (4)$$

where Pd stands for the proportion of dead trees in the stand, Alt is altitude (metres), Slope is measured in degrees, avgDBH is the average diameter at breast height (centimetre) and Sh is the standard deviation of the height of trees (metre). The relative mortality at stand-level caused by a wildfire (equation 4) decreases with higher average diameter at breast height (Table 3). Conversely, higher variability in tree heights (Figure 1) and steep slopes increase the stand-level mortality. The model showed a percentage of concordant pairs of 80 per cent) and the area under the ROC curve (0.846) indicated excellent discrimination (Hosmer and Lemeshow, 2000).

The tree-level mortality model that best predicted the probability of an individual maritime pine tree to die if a forest fire occurs was:

$$\text{Ptd} = \frac{1}{1+e^{-(-3.1958-0.0244\,\text{DBH}+0.2601\,\text{BAL}+6.3382\,\text{Pd})}}, \quad (5)$$

where Ptd is the probability of an individual tree to die, DBH is the tree diameter at breast height (centimetre), BAL is the basal area of trees higher than the studied tree (square metre per hectare) and Pd is the proportion of dead trees in the stand. The model indicates that trees with large DBH are less prone to die due to a wildfire (Figure 2 and Table 4). Conversely, trees suppressed (high BAL) and located in stands with higher expected stand damage (Pd) have higher mortality probability (equation 5). The model was successful in predicting whether mortality did occur after the wildfire in 86 per cent of trees (i.e. percentage of concordant pairs). The area under the ROC curve (0.85) indicated excellent discrimination (Hosmer and Lemeshow, 2000). The model shows a CCR of 85.1.

Table 2: Parameter estimates, standard errors (SE), Wald X^2 statistics and P-values for the model predicting whether mortality will occur in a stand (equation 3)

			Wald	
Variables*	Estimate	SE	X^2	$P > x^2$
Intercept	21.231	0.5497	14.9161	<0.0001
avgDBH	−0.1134	0.0344	10.8796	0.0010
G/Dg	2.3943	0.9150	6.8474	0.0089

*For the parameter definitions see Table 1.

Table 3: Parameter estimates, standard errors (SE), Wald X^2 statistics and P-values for the model predicting degree of damage caused by a wildfire equation 4 (i.e. proportion of dead trees in the stand)

			Wald	
Variables*	Estimate	SE	X^2	$P > x^2$
Intercept	0.7065	0.0687	105.8	<0.0001
Altitude	0.00491	0.000106	21.5592	<0.0001
Slope	0.1158	0.00272	18.0577	<0.0001
avgDBH	−0.1649	0.00426	14.9658	<0.0001
sh	0.1456	0.0177	67.5690	<0.0001

*For the parameter definitions see Table 1.

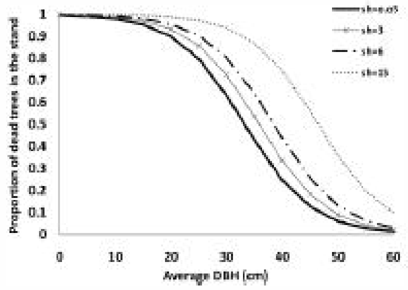

Figure 1. Effect of average diameter (avgDBH, centimetre) and standard deviation of height (sh, metre) on the proportion of dead trees according to equation 4 for a stand located at 500 m above sea level with a slope of 20°.

The most appropriate cut-points were calculated for the model predicting whether mortality will occur in a stand (Table 5). If the value that maximizes the CCR (75.8 per cent) was used as criteria to choose the cut-point, its value would be 0.36 (Table 5). According to this value, mortality would occur in 96 per cent of the plots (classified as dead), while inventories after wildfire events showed that mortality did occur only in 75 per cent (93 plots over 124). Around 24 per cent of the predictions were false positives (i.e. stands that did not have any dead trees but were classified as if mortality had occurred) and 40 per cent were false negatives (i.e. stands that had dead trees but were classified as if mortality had not occurred). The cut-point at which

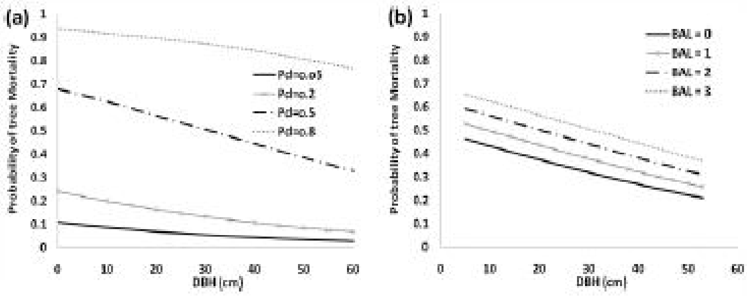

Figure 2. Effect of diameter at breast height (d.b.h., centimetre), stand-level mortality (Pd) and BAL (m^2 ha^{-1}) on the probability of tree mortality using equation 5 for a BAL of 3 m^2 ha^{-1} (a) and a Pd of 0.5 (b).

Table 4: Parameter estimates, standard errors (SE), Wald X^2 statistics and P-values for the tree-model predicting the probability of a tree to die due to a forest fire (equation 5)

			Wald	
Variables*	Estimate	SE	X^2	$P > x^2$
Intercept	−3.1958	0.4237	56.9008	<0.0001
DBH	−0.0244	0.0109	5.0261	0.0250
BAL	0.2601	0.0754	11.8973	0.0006
Pd	6.3382	0.4276	219.7140	<0.0001

*For the parameter definitions see Table 1.

the sensitivity and specificity curves crossed was ~0.76. Using this value led to a CCR of 66.9 per cent and the percentage of stands classified as having mortality was 58.1 per cent (classified as dead). Using this cut-point, in 41.9 per cent of the stands classified as not having mortality (classified as alive), some trees had actually died (i.e. false negative). On the other hand, when the average observed percentage of event occurrence (Monserud and Sterba, 1999) was used, a cut-point of 0.70 would be chosen. This cut-point classified 26.6 per cent of stands as stands where no mortality did occur (classified as alive); this value was very close to the real observed rate which is 25.5 per cent (i.e. 31 plots over 124). However, in this case, the number of false negatives was 54.5 per cent and the CCR was 72.6 per cent. Analysing these different options and having in mind that a compromise has to be found between classification of dead trees and good prediction of mortality and survival rates, a cut-point value of 0.7 is recommended as the predicted stands with mortality is the closest with the observed in the inventoried data.

Discussion and conclusions

Post-fire mortality has been studied using a variety of direct and indirect methods (e.g. Fowler and Sieg, 2004; Sieg *et al.*, 2006). However, they need information that is seldom available to forest managers beforehand (e.g. tissue damage, fire intensity). Fire simulators may provide information about tissue damage or fire intensity; however, they need information about specific weather conditions and fuel accumulation at the time of fire that are hard to predict over long planning horizons (Rothermel, 1991; Finney, 1999; He and Mladenoff, 1999; González *et al.*, 2007). The unavailability of this information constrains the applicability of these methods in long-term forest management planning. Thus, both approaches are hardly practical for forest planning.

The proposed logistic modelling approach to post-fire mortality for enhanced forest planning has been used earlier for predicting tree-mortality as a consequence of wind damage (Lohmander and Helles, 1987; Jalkanen and Mattila, 2000), prescribed fire (Botelho *et al.*, 1996) and wildfire (Regelbrugge and Conard, 1993; McHugh and Kolb, 2003; Rigolot, 2004; González *et al.*, 2007). This approach has been also used to model natural tree mortality (Fridman and Stahl, 2001; Álvarez-González *et al.*, 2004). Our research confirmed the potential of the proposed approach to develop mortality models that may be used in forest planning (Reinhardt and Crookston, 2003; González *et al.*, 2007; Hyytiäinen and Haight, 2009).

The proposed approach was tested using a dataset encompassing 1174 trees in 124 plots located in 26 fire perimeters in Portugal. Results suggest that the models may predict accurately post-fire mortality in maritime pine stands in Portugal. An advantage of the three-step methodology used in this study compared to other traditional approaches is the possibility of detecting stands where no mortality occurs.

Otherwise, traditional models always generate some mortality for all plots (Fridman and Stahl, 2001). This is especially important in species that have demonstrated a good fire resistance as the case of maritime pine (Ryan *et al.*, 1994; Fernandes *et al.* 2008).

Prediction and classification do not follow the same pattern, so a compromise must be reached between good classification of dead trees and good prediction of mortality and survival rates when choosing a cut-point (Crecente-Campo *et al.*, 2009). In our study, a cut-point of 0.7 for the model predicting whether mortality occur in a stand (equation 3) was selected. To determine this cut-point, the

Table 5: Prediction parameters depending on the cut-points used to transform a continuous probability into a 0–1 dichotomous value predicting whether there is mortality in a stand or not

Cut-point	CCR (%)	Sensitivity (%)	Specificity (%)	False positive* (%)	False negative† (%)	Classified as dead (%)	Classified as alive (%)
0.36	75.8	97.8	9.7	23.5	40.0	96.0	4.0
0.38	75.8	96.8	12.9	23.1	42.9	94.4	5.6
0.40	75.8	96.8	12.9	23.1	42.9	94.4	5.6
0.42	75.0	94.6	16.1	22.8	50.0	91.9	8.1
0.44	75.0	94.6	16.1	22.8	50.0	91.9	8.1
0.46	75.0	94.6	16.1	22.8	50.0	91.9	8.1
0.48	75.0	94.6	16.1	22.8	50.0	91.9	8.1
0.50	74.2	93.5	16.1	23.0	54.5	91.1	8.9
0.52	75.0	93.5	19.4	22.3	50.0	90.3	9.7
0.54	75.0	93.5	19.4	22.3	50.0	90.3	9.7
0.56	73.4	90.3	22.6	22.2	56.3	87.1	12.9
0.58	73.4	90.3	22.6	22.2	56.3	87.1	12.9
0.60	74.2	89.2	29.0	21.0	52.6	84.7	15.3
0.62	74.2	88.2	32.3	20.4	52.4	83.1	16.9
0.64	74.2	86.0	38.7	19.2	52.0	79.8	20.2
0.66	72.6	83.9	38.7	19.6	55.6	78.2	21.8
0.68	71.8	82.8	38.7	19.8	57.1	77.4	22.6
0.70	72.6	80.6	48.4	17.6	54.5	73.4	26.6
0.72	71.0	77.4	51.6	17.2	56.8	70.2	29.8
0.74	69.4	74.2	54.8	16.9	58.5	66.9	33.1
0.76	66.9	66.7	67.7	13.9	59.6	58.1	41.9
0.78	63.7	62.4	67.7	14.7	62.5	54.8	45.2

The percentage of observed plots where occurred tree mortality was 75%.

* Stands that did not have any dead trees but were classified as if mortality had occurred.

† Stands that had dead trees but were classified as if mortality had not occurred.

observed percentage of stands with mortality was used as suggested by Monserud and Sterba (1999). After a wildfire, the number of stands where at least some mortality occurs is usually much greater than the number of stands where no mortality occurs, so errors that result in underestimating the number of stands where mortality occurs could have more impact. Thus, cut-point of 0.7 presented the best compromise between underestimating the number of stand where mortality occurs (the case of cut-point = 0.76) and overestimating mortality that occurs if cut-point that maximizes the number of CCR is used (0.36).

In the framework of forest management planning, equation 3 may be used to predict whether mortality may occur in a stand after a wildfire. As these models are developed to support management planning, equation 4 estimates the number of trees that will die in the stand (i.e. percentage of trees) after a wildfire (if mortality indeed occurs). Equation 5 may then be used to distribute that mortality among trees. Thus, equation 5 may be used to predict the probability of mortality of each tree in the stand and to build a list of all trees in the stand ordered according to this probability (trees with higher probability of mortality are ranked first in the list). The management planning model may then select the trees that will be assumed to die for planning purposes by going down the list and stopping when it reaches the number of trees that are estimated to die (from equation 4). For this reason, no cut-point is needed to transform the estimated probability into a dichotomous variable (e.g. death or no death). Equation 5 is especially important when the growth and yield simulation uses an individual tree model (which means that every tree may have different characteristics). As suggested by González *et al.* (2007), the tree mortality equations can be used to generate mortality variation if a stochastic component corresponding to the residual variation of the stand-level mortality model is added to the prediction.

Our models are developed to predict mortality if a fire occurs in a forest management planning context. Thus, unlike former models for post-fire tree mortality that were developed to assess mortality after a wildfire occurrence, our models do not use tissue damage or fire severity as predictors. This is in concordance with the approach presented by González *et al.* (2007). However, some of the variables included in our models have a clear correlation with fire behaviour. This is the case of slope as steeper slopes increase the expected mortality. Biometric variables that impacted post-fire mortality included tree diameter (average d.b.h. of the stand and d.b.h. of the tree), variation of heights (Sh) and indicators of density such as basal area (G) and competition index (BAL). Other significant variables were related to fire behaviour (i.e. slope) and stand location (i.e. altitude). This agrees with findings of Fernandes *et al.* (2008), who stated that the level of injury and mortality for a given species is a combined outcome of fire behaviour, tree size and stand structure. In addition, Fernandes (2009) presented a study where combined forest structure data and fuel modelling to classify fire hazard in Portugal. He concluded that forest structure is highly related to fire intensity. Based in previous studies and according to the purpose of this model, no direct measurements of fire behaviour were included in the model. This is because the purpose of this model is to predict mortality for long-term planning horizons (i.e. over 60 years planning periods), where data needed to use fire behaviour models is limited or even not possible to calculate for small scale areas located in Portugal (e.g. bush development, 1–10 h fuel moisture content, specific weather conditions in a specific day for long periods). However, dataset of fire occurrences which cover many different fire events was used, in addition, indirect variables that may be related to fire behaviour as can be the slope or the vertical structure of the stands were included in the analysis.

The need for an individual-tree mortality model for long-term planning is justified by the fact that growth simulation may be done with individual tree-growth models. Thus, individual tree-mortality models even in long-term planning periods help to distribute stand mortality over trees with different tree sizes.

In concordance with other studies, in our stand-level mortality model, steeper slopes increase the expected proportion of dead trees in the stand; this may be explained by an easier transfer of heat uphill (Agee, 1993; González et al., 2007; Hyytiäinen and Haight, 2009). In our case, altitude correlates positively with the degree of mortality in burned areas because most of the burned stands were located in high altitudes.

The coefficients of biometric variables in stand-level mortality models indicate that even-aged stands with higher tree diameters have lower stand mortality than irregular stands with trees with smaller dimensions. Moreover, in stands with higher densities and smaller diameters, stand mortality is expected to be higher than in stands with lower densities. This is in concordance with studies in North-American conifer dry forests (Pollet and Omi, 2002; Agee and Skinner, 2005; Ritchie et al., 2007) which indicate that fire severity is lower in open stands, especially when thinning is concurrent with surface fuel treatment. Also in Portugal Fernandes et al. (2005, 2010) and in southern Spain Gallegos et al. (2003) indicated that dense maritime pine stands have higher crown fire potential and tend to experience higher fire severity which results in higher post-fire tree mortality. They indicate that high densities favour death of the lower canopy branches which are retained, establishing continuity with the live crown and, consequently, implying high crowning potential. In our case, variability of tree heights (Sh) is highly related to vertical continuity of fuels and thus with high crowning potential and higher mortality.

At tree level, tree diameter (d.b.h.) was found to be negatively related with tree mortality. This is in concordance with other studies (Ryan and Reinhardt, 1988; Hély et al., 2003; González et al., 2007). Moreover, a competition index (BAL) was found to be positively related with tree mortality; the more suppressed is the tree (i.e. higher BAL) the more probability of death. This is in concordance with findings by González et al. (2007) and Van Mantgem et al. (2003), who concluded that a suppressed tree is more prone to die than dominant trees due to both, the fire damage and the stress before the fire event.

When no pre-fire inventory was available, reverse engineering (i.e. regression models) was needed to reconstruct the stand. Thus, the quality of the models is dependent on the quality of the equations used for that purpose. Stands where burned trees had been harvested were not used in the model fitting process. Further, this research considered mortality within a period extending between 1 and 2 years after the wildfire, a time period between fire and the inventory that has been already used by other authors (Botelho et al., 1998; Fernandes et al., 2008). In some cases, this may lead to an underestimation of mortality caused by the wildfire. Nevertheless, the development of the first maritime pine post-fire mortality models in Portugal took into account all available data and information.

Validation of the models was done through studies of the performance of the functions. No specific validation data sets were set-aside and later used for that purpose. This was for two main reasons. Firstly, the relatively small number of observations in the stand dataset. Secondly, the best possible parameter estimates were of greater interest. There are advantages and disadvantages of splitting the dataset for model validation purposes as discussed by Kozak and Kozak (2003). They concluded that that cross validation by data splitting and double cross validation provide little, if any, additional information in the process of evaluating regression models. Other authors have the same opinion, for instance, Picard and Cook (1984).

Post-fire mortality models are a valuable forest management planning tool (González et al., 2007). Their usefulness in forest planning depends on the information they may provide about the impact on mortality of variables whose future value may be estimated with reasonable accuracy. This research encompassed the development of maritime pine post-fire stand and tree mortality models for enhanced forest planning in Portugal. These models are based on variables that are under the control of forest managers (e.g. forest density, mean diameter) and provide information about the impact of forest fires under alternative forest conditions. Thus, these models are instrumental to designing silvicultural strategies that may decrease mortality caused by wildfires and that they can be used to effectively integrate fire risk into forest management planning.

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A STOCHASTIC, CELLULAR FOREST HARVESTING MODEL INTEGRATING WILDFIRE RISK AND DISPERSION

CHAPTER V - A STOCHASTIC, CELLULAR FOREST HARVESTING MODEL INTEGRATING WILDFIRE RISK AND DISPERSION (WORK IN PROGRESS)

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V.1 - Abstract

Understanding a disturbance like a fire on forest landscapes is a challenge because of complex interactions over a range of temporal and spatial scales. We present a stochastic, cellular multi-objective forest harvest scheduling model incorporating a mechanistic model of fire risk probability based on the state of a cell and the fire risk in neighboring cells.

V.2 - Introduction

Fire is a major disturbance in the Mediterranean landscape (Rundel 1998, Ferreira et al. 2011), and in recent decades its incidence has increased dramatically in Southern Europe (Rego 1992, Moreno 1999, Pereira et al. 2006, Velez 2006, Pausas 2008), especially in Portugal where nearly 40% of the forest area has burned during the last three decades (Marques et al. 2011). This problem has been further aggravated by the absence of adequate measures to control and avoid wildfires. Such fires have devastating effects on the landscape, affecting the ecological balance of the forest environment, emitting large quantities of stored carbon to the atmosphere, and causing substantial loss of human lives. Moreover, the lack of proper forest management has added to the problem.

Fire behavior is influenced by three factors: fuel, weather and topography. Of these, only fuel can be actively managed. Fire managers are tasked with reducing the flammability of the landscapes by applying fuel treatments to modify fuel quantities, patterns and distribution (Martell 2007, Minas et al. 2013). The challenge of creating and maintaining desired forest conditions has been discussed by a large number of authors who have suggested that fire and fuel conditions in certain forests would be improved by creating a forest with densities and age structures that emulate historical conditions or natural processes. Scheduling removals to create and maintain such a forest presents forest managers with a formidable management planning problem.

Harvesting can reduce the ability of fire to spread across a landscape, and the spatial distribution of harvesting activities can be a key factor in reducing in the risk of large fires (Johnson et al. 1998, Gustafson et al. 2004, Gonzalez et al. 2005, Palma et al. 2007). Thinning and other fuel management practices have been shown to be effective in reducing fire hazards (Stephens 1998, Graham et al. 1999, Pollet and Omi 2002).

Finney (2005) addressed the importance of incorporating the probability of fire occurrence, fire behavior, and values at risk (to which we would add fire suppression effectiveness) in strategic or long-term fire management planning. Sampson and Sampson (2005) noted that "all wildland areas share wildfire risks with their surroundings," but the development of spatial fire risk assessment procedures has yet to receive the attention it deserves.

The use of mathematical models for managing fires has a rich history in many regions of the world (e.g., Hof and Omi 2003, Wei et al. 2008). Due to the complexity of the problem, finding the optimal combination of stand management alternatives to maximize or minimize a landscape metric often requires numerical optimization techniques. As most landscape metrics are spatial, the computational complexity of many planning problems calls for the use of heuristic search techniques (Borges et al. 2002, Pukkala 2002). These techniques are generally more flexible and more capable of addressing complicated objective functions and constraints than exact algorithms (Reeves 1993, Borges et al. 2002).

This paper presents a timber harvest allocation model whose objective is to maximize the expected value of a forest that is subject to the risk of burning. It assumes that information regarding the probability that a fire will burn any given portion on the landscape can be obtained from fire simulation and behavior models. The paper presents a preliminary version of the model that focuses on how such information can be used to develop an optimization model to efficiently allocate fuel treatments across a landscape.

V .3 - The model

The state space of the forest is defined as an m×n grid (Figure V.1) where each cell in the grid is a management unit with an initial age at time 0 of α_{i0}. Corresponding to each age is a timber yield, y_α, and a flammability index, $f_\alpha \in (0,1)$. Timber yield is a monotonically increasing, concave function of stand age (i.e., $y'_\alpha>0$ and $y''_\alpha<0$). The flammability index is low when stands are young and when they are old and highest when stands are of intermediate age (i.e., $f'_\alpha>0$ for $\alpha<\alpha^*$, $f'_\alpha=0$ for α^*, $f'_\alpha<0$ for $\alpha>\alpha^*$ and $y''_\alpha<0$, where α^* is the age at which flammability is highest).

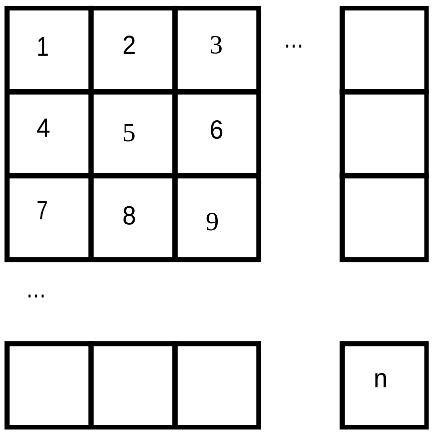


Figure V.1 – State space example

The planning horizon consists of a set of T periods, t = 1,...,T, each τ years in length. The model's decision variables represent the decision to harvest management unit i in period t. Let X_{it} be 1 if management unit i will be harvested in period t and 0 if management unit i will not be harvested in period t. A management unit can only be harvested once during the planning horizon. Hence:

$$\sum_{t=0}^{T} \mathbf{X}_{it} = \mathbf{1} \ \forall \ \mathbf{i} \qquad [\boldsymbol{Eq}.\mathbf{V}.\mathbf{1}]$$

The most important and challenging component of this model is determining the probability that a given stand will burn in any given period. In any given period there are three possibilities: either the stand burns because a fire started in it, the stand burns because a fire in an adjacent cell spread to it, or the stand does not burn. We assume that the probability that a given stand will burn in a given period is a function of the flammability of the stand, the probability that each stand around it will burn, and the probability that a fire will spread from an adjacent stand. To make the model more tractable, when we calculate the probability that a stand will burn, we assume that none of the stands have burned. This tends to over-estimate the probability that a stand will burn, since if an adjacent stand has already burned then it is assumed to no longer be flammable. Thus,

$$p_{it} \cong p_{it}^{I}\left(f_{a(it)}\right) + \left(1 - p_{it}^{I}\left(f_{a(it)}\right)\right) \sum_{k \in Adj_i} \left[\prod_{m \in \{Adj_i \setminus k\}} (1 - p_{mt}) \right] p_{kt} \times p_{kit}^{S}\left(f_{a(it)}\right) \quad [Eq.\mathrm{V}.2]$$

Where:

p_{it} = the probability that the stand i will burn in period t; (similarly for p_{kt} and p_{mt});

$p_{it}^{I}\left(f_{\alpha(it)}\right)$= the probability that a fire will start in stand i in period t, which is a function of the flammability of the stand in that period;

$p_{kit}^{S}\left(f_{\alpha(it)}\right)$ = the probability that a fire will spread from stand k to stand i in period t, which is a function of the flammability of stand i in that period, and

Adj_i = the set of stands that are adjacent (including corner adjacencies) to stand i.

This formula assumes that whether or not each of the adjacent cells burns are independent events, and that the fire can spread to cell i from only one of the neighboring cells. So if came from cell, couldn't have come from B and C

$$P(A_a \cap \overline{A_b} \cap \overline{A_c}) = P(A_a)(1 - P(\overline{A_b}))(P(\overline{A_c}))$$

For each cell k (adjacent of i) we have the product of probabilities of the others adjacent cells not having been burned.

So:

$$m \in Adj\ i\ \cap m \neq k$$

A key problem with the above equations is that the probability of each stand burning is a function of the probability that every other stand will burn, which creates seemingly intractable circularities in the calculation of the probability that any one stand will burn. To get around this, we separate the burn probabilities into four independent cases based on wind direction. The cases are that the wind is from the NW, the NE, the SE, and the SW. For notational purposes, let D be the set of wind directions: D = {NW, NE, SE, SW}. Thus, the probability that a stand i will burn in period t can be written:

$$\mathrm{p_{it}} = \sum_{\mathrm{d \in D}} \mathrm{p_d} \times (\mathrm{p_{it}}|\mathrm{d}) \qquad [Eq.\mathrm{V.3}]$$

Equation V.3 can now be revised as follows:

$$p_{it}|d \cong p_{it}^{I}(f_{a(it)}) +$$

$$\left(1 - p_{it}^{I}(f_{a(it)})\right) \sum_{k \in Adj_i^d} \left[\prod_{m \in \{Adj_i^d \backslash k\}} (1 - p_{mt}|d) \right] p_{kt}|d \times p_{kit}^{S}|d(f_{a(it)}) \qquad [Eq.\mathrm{V.4}]$$

Where:

$p_{it}|d$ = the probability that the stand i will burn in period t given that the wind is from direction d; (similarly for $p_{kt}|d$ and $p_{mt}|d$);

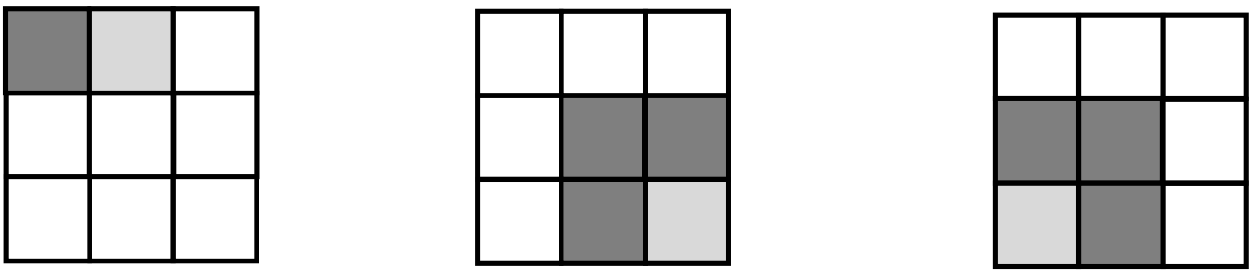
$p_{kit}^{S}|d(f_{\alpha(it)})$ = the probability that a fire will spread from stand k to stand i in period t, given that the wind is from direction d, and

Adj_i^d = the set of stands that are adjacent to and upwind from stand i.

It is reasonable to assume that the probability of the wind blowing from a given direction, pd, does not change over time. Furthermore, we assume that the probability that a fire will start in stand i is independent of the wind direction; this assumption would be easy to relax.

The advantage of separating the probabilities into independent cases based on wind direction is that if we also assume that the probability of fire spreading upwind is zero, then $p_{it}|d$ only depends on the flammability of stand i and the probabilities that an upwind stand will burn and that the fire will spread to stand i. This eliminates the circularity in the calculation of the probabilities and allows us to calculate the probability of each stand burning by starting in the upwind corner of the grid and working downwind. For example, if the wind is from the NW, processing would start in the upper left corner and proceed to the right through the top row; when the top row is done, processing would continue with the second row, moving from left to right, and so on.

For every cell, the "set of connected cells" is defined as the neighborhood of immediately adjacent cells, considering each wind direction. In the simplest case each set would be composed of all immediately adjacent cells (Figure V.2). In alternative case is where sets are constructed to take into account heterogeneous landscape features such as the prevailing wind directions associated with severe burning conditions.



a) Simplest case, with north-westerly prevailing wind direction where the I cell in light green is connected to the cell in dark green.

b) Illustrative with north-westerly prevailing wind direction where the right bottom cell is connected to three other cells

c) Illustrative with north-easterly prevailing wind direction where the left bottom cell is connected to three other cells

Figure V.2. Illustrative examples of sets of connected cells.

The goal of this model is to schedule harvesting treatments to produce a fragmented landscape fuel complex with a view to inhibiting fire spread. Other similarly structured models involving dynamic natural systems with spatially explicit management objectives in the forest operations research literature include Martell et al. (1998), Weintraub et al. (2000), Weintraub and Romero (2006), and Bjørndal et al. (2012). While our model employs some concepts from this earlier work (i.e., adjacency, connectivity and inter-temporality), on the whole, fuel management problems tend to differ significantly.

The objective of the model is to maximize the expected present value of the economic return from the forest. To calculate the probability that a stand has not burned before it is harvested, we have to make an assumption about whether the harvest event for a given period happens before or after the fire event in the same period. Since our probability of a stand burning in a given period is based on the state of the stand and its neighbors before any of them are harvested, it makes sense to assume that the fire event happens first. In this case, the probability that a stand has not burned before it is harvested in period 1 is (1-pi1). The probability that a stand has not burned before it is harvested in period 2 is (1-(pi1 + (1-pi1) pij2)). In general, that a stand has not burned before it is harvested in period t, p'it, is:

$$p'_{it} \cong 1 - \sum_{t'=1}^{t} \left(\prod_{t''=1}^{t'-1} (1 - p_{it''} \right) p_{kt'} \qquad [Eq.\text{V}.5]$$

Essentially, we want to optimize:

Maximize the present value of the economic return from the forest, assuming that it does not burn:

$$\text{NPV} = \sum_{i} \sum_{t=0}^{T} r_{it} X_{it} \qquad [Eq.\text{V}.6]$$

Where:

rit = the present value of the economic returns from stand i, plus its ending value if it is harvested in period t (i.e., rit = δtπγα(it) + δTϕit, where δ is the discount term, π is the price of timber, α(it) is the age of stand i if it is harvested in period t, and ϕit is the ending value of stand i if it is harvested in period t).

Now the objective function can be rewritten as:

$$E(NPV) = \sum_{i}\sum_{t=0}^{T} p'_{it} r_{it} X_{it} + \sum_{i}\sum_{t=1}^{T} p_{it}\theta_{it} \qquad [Eq.V.7]$$

Where:

θit is the present value of stand i if it burns in period t.

Note that the value of the second term depends indirectly on the values of the decision variables because the pit's depend on the management decisions that are made.

If the model has multiple planning periods, it should have some kind of even flow constraints, and these constraints need to recognize that at least some of the harvests that are planned will not occur because the stands will burn first. This can be addressed by using accounting constraints that give the expected yield for each period. This procedure reduces expected yields for future periods more than expected yields in early periods because the probability that a stand will be burned before it is harvested increases with the length of time before the stand is harvested. The expected yield in period p can be calculated using the following accounting constraint:

$$\sum_{i}\sum_{i} p'_{it} y_{\alpha(it)} X_{it} - E(H_t) = 0 \qquad [Eq.V.8]$$

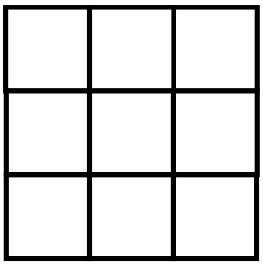
Now one can formulate any of the standard flow or harvest target constraints using the $E(H_t)$ variables.

The key assumption that a stand can burn only once was made to improve the tractability of the specification and calculation of the burn probabilities. As long as the planning horizon is short or the probability of any given area burning is low, then this should be a reasonable assumption. If the probability that stands will burn is high, then it makes sense not to have too long a planning horizon, as it will be necessary to re-plan frequently as it becomes known which areas have actually burned.

We now have a fully-specified harvest scheduling model. The problem is that it is highly non-linear. However, there are many heuristic solution algorithms for solving non-linear problems like this: simulated annealing, tabu search, sequential quenching and tempering, genetic algorithms, etc. Furthermore, with a small problem it is possible to solve it using complete enumeration.

V.4 - Case example - The problem

The example case forest is a 3×3 matrix as represented in Figure V.3. Timber age classes can be assigned to each cell, corresponding to important seral stages for wildfire. Table V.1 shows the yields and flammability indexes for each age class. The table also identifies the ignition probability for the landscape, I. While other assumptions can easily be modeled, all cells are assumed to have the same ignition probability. The probability that a fire will start in a cell and that the cell will actually burn – $p_{it}^{I}\left(f_{a(it)}\right)$ from Equations V.3 and V.5 – is given by the product of the ignition probability and the flammability index.



FigureV.3– The 9-cell example planning area.

Table V.1 –Yield (yα) and flammability index (fα) for each timber age class (α) and ignition probability (I) for all cells.

α	yα	f α
0	0	0
1	1	0,5
2	3	0,8
3	5	0,6
4	6	0,3
	I	
	0,02	

Because fires often start in one cell and spread to others, fire risk in a cell is a function of the fire ignition risk within the cell. Examples of models for predicting ignition risk have been developed by Catry et al. (2008, 2009), Vasconcelos et al. (2003), Botequim et al. (2013), Garcia-Gonzalo et al. (2012), Marques et al. (2011), and Gonzalez et al (2005). A model of the risk of fire entering a cell from adjacent cells (spread risk) has been developed by Chou et al. (1993).

When the wind is blowing from one of the corners of the landscape grid – i.e., either from the NW, the NE, the SE, or the SW – a fire can spread downwind either horizontally, vertically, or diagonally. This is illustrated in Figure 4. The figures show a parameter used in the model to determine the probability that a fire will spread from one cell to another, which we refer to as the spread propensity. The spread propensity, which depends on the wind direction, is multiplied by the flammability index of the cell into which the fire might spread to determine the probability that the fire will spread to that cell. Figure 4 shows how the spread propensity varies depending on the wind direction. In general, we have assumed a spread propensity of 0,5 for vertical and horizontal adjacencies and 0,75 for diagonal adjacencies.

A	B
C	D

0	0.5
0.5	0.75

Northwest wind direction

B	A
D	C

0.5	0
0.75	0.5

Northeast wind direction

C	D
A	B

0.5	0.75
0	0.5

Southwest wind direction

D	C
B	A

0.75	0.5
0.5	0

Southeast wind direction

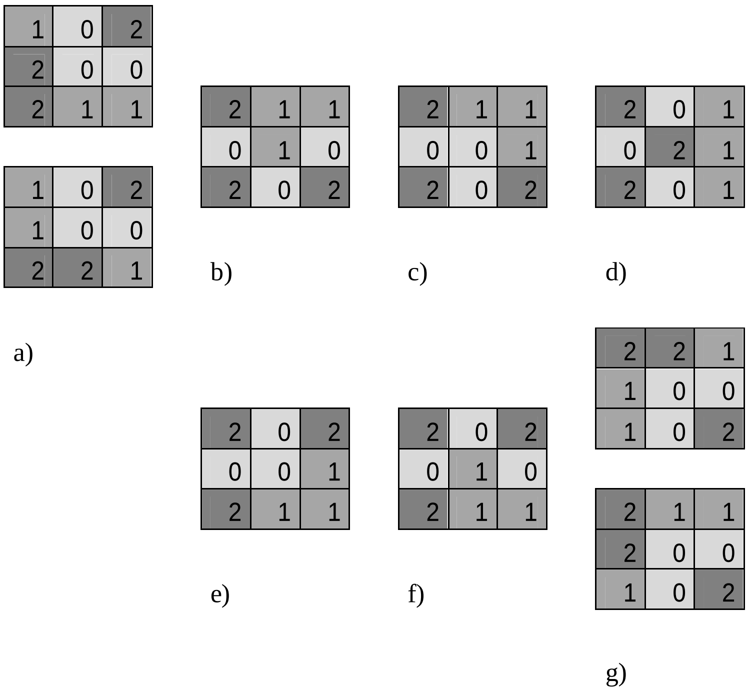
Figure V.4– Fire spread propensities according to different wind directions

V.5 – Results and discussion

V.5.1 - Static problem

V.5.1.1 – Regulated forest

In order to demonstrate its functionality, we applied the model to a number of hypothetical landscapes with all possible combinations of timber age classes composing an even age forest. In this example and for simplicity the treatment cost and timber value are set at a constant value of one unit per cell across the entire landscape. No treatment restrictions or ecological constraints are imposed.



FigureV.5– a) Optimal landscape pattern for a 9-cell forest with three cells each in three age classes when the predominant wind is from the SW (Case SW-1), b) Solution when the wind is from the SW 98.7% of the time and from the NW – 1.3% of the time (Case SW-0.987; NW-0.013), c) Solution for case SW-0.633; NW–0.367, d) Solution for case SW-0.59; NW–0.41, e) Solution for case SW-0.409; NW – 0.591, f) Solution for case SW-0.366, NW–0.634, g) Optimal landscape when the predominant wind is from the NW (Case NW-1).

Thus in this homogeneous landscape with no ecological constraints or treatment restrictions the spatial solution amounts to the creation of an initial pattern and then the maintenance of this pattern through a recurring treatment cycle.

v.5.1.2 – Uneven aged forest

In practice, most forest landscapes will not have perfectly regulated age-class distributions. To demonstrate how the model handles this, we assume that the optimal rotation is divided into five timber age classes, so that it is no longer possible to have an even age-class distribution (nine cells cannot be divided evenly into five age classes). The treatment costs, annual budget and set of connected cells definitions remains the same as in the previous example.

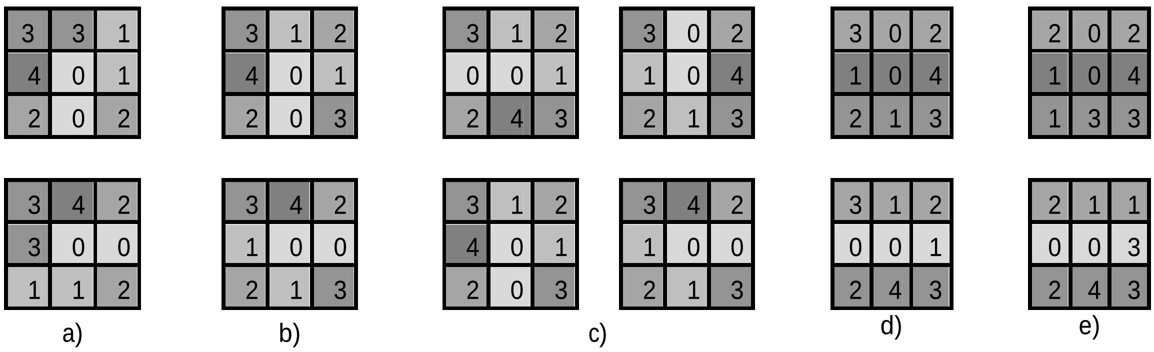


Figure V.6 – a) Optimal landscape pattern for a 9-cell forest with two cells each in four age classes and one cell in the fifth age class when NW is the predominant wind b) Solution when having wind with the following percentages of predominance NW- 0.83 and SE – 0.17 c) Solution when having wind with the following percentages of predominance NW and SE of 0.5 d) Solution when having wind with the following percentages of predominance NW- 0.16 and SE – 0.64 e) Optimal landscape when SE is the predominant wind

V.5.2 – Dynamic problem

V.5.2.1 – Even aged forest

Starting with the optimal solution from the static problem, when we have two wind directions (NW and SW), with the wind blowing from the NW 53 percent of the time, considering two 10-yr periods, and an interest rate of 3% per period, timber price and costs set a unit for all landscape, the solution is presented in figure V.7, below.

t_0			Action			t_1			Action			t_2		
2	0	1	1	0	0	0	1	2	0	0	0	1	2	3
0	2	1	0	1	0	1	0	2	0	0	1	2	1	0
2	0	1	1	0	0	0	1	2	0	0	0	1	2	3

Figure V-7 – Optimal solution and action to perform within each period in an even-aged forest. 1 in figure "Action" means that the cell in same position in our 9 cell landscape should be harvest. Figure t1 and t2, shows the optimal solution after action being performed.

.V.5.2.2 – Uneven aged forest

Starting with the optimal solution from the static problem, with the wind from the NW of the time and the same conditions as in previous example, the solution is presented in figure V.8, below.

t_0			Action			t_1			Action			t_2		
3	4	2	1	1	0	0	0	3	0	0	1	1	1	0
1	0	0	0	0	0	2	1	1	0	0	0	3	2	2
2	1	3	0	0	1	3	2	0	1	0	0	0	3	1

FigureV.8– Optimal solution and action to be performed within each period in an uneven-aged forest. 1 in figure "Action" means that the cell in same position in our 9 cell landscape should be harvest. Figure t1 and t2, shows the optimal solution after action being performed.

v.6 – Discussion

There is a recognized need to apply and maintain timber harvest to reduce catastrophic wildland fires in forests (Pollet and Omi 2002; Agee and Skinner 2005; Prichard et al. 2010, Chung et al. 2013). However, treating all forest lands considered at risk would be costly and impractical. Forest managers who are faced with limited budgets, narrow burning windows, air quality issues, and concerns about treatment effects on other critical forest resources,must establish priorities for where, when and how to implement fuel treatments. Science-based as well as field-applicable guidelines are necessary to strategically locate, schedule and apply fuel treatments to effectively reduce catastrophic fire and restore ecosystem health on landscapes over time (Collins et al. 2010).

Although fuel treatments may not stop wildfires (Finney and Cohen, 2003 and Wei et al, 2003), they can alter fire behavior and reduce intensities across a landscape. By carefully planning the spatial location of fuel treatments in a landscape, forest managers can increase the effectiveness of these treatments by fragmenting the fuel complex, thus increasing the likelihood that suppression will be effective (Finney, 2001).

The direction of fire spread is an important factor to be considered in designing the spatial distribution of fuel treatments. This study assumed that the main factor determining fire spread patterns was the wind directions. However, spread direction can also depend on topography. Future enhancements could define fire spread directions using a spread direction matrix that reflects the combined effects of both wind direction and topography. Such matrix can store a specific spread direction for each cell. Therefore, fuel treatment locations could also consider landscape topography.

Fuel treatment is one of many components in an integrated fire management system. Fuel treatments need to be combined with other fire program components, such as suppression and prevention, to improve the overall efficiency of fire management. While fuel treatments may not always stop the spread of fire, they can improve suppression productivity. Basic information such as the spatially explicit landscape fire risk distribution can potentially be used to tie different fire management components together.

V.7 – Conclusions

The optimal location of fuel treatments across a landscape remains a central challenge in fire management decision-making processes. Landscape conditions are usually heterogeneous, and scheduling fuel treatments by following regular patterns may not be an efficient strategy for reducing expected fire losses. Multiple spread directions may further complicate the problem by requiring the prevention of the spread along multiple pathways.

The model developed here illustrates a potential approach to integrate these practical management concerns to improve the overall effectiveness of landscape level fuel treatments. By implementing the complete enumeration approach, we separated the probability of fire in each cell into the probability that the fire would start in that cell and the probability that the fire would spread to that cell from an adjacent cell. The integer programming model uses the calculated probabilities based on the interrelationships between cells regarding fire risk for different wind directions. The values at risk at each specific location of a landscape can be acquired through either expert opinion or through other kinds of economic analysis. The model can then incorporate the potential fire-loss information into the decision-making process to determine the optimal spatial distribution of fuel treatments on the landscape. In addition to the benefits of protecting the forest from potential loss from fires, the cost and feasibility of scheduling certain types of fuel treatments at specific locations of a landscape can also be considered.

The resulting forest removals problem is difficult because it is intrinsically nonlinear. The assumptions necessary to make such problem linear are rather heroic. Accounting for the spatial nature of fire itself is difficult because fire origins and behavior can be quite random and unpredictable. An approach that focuses on spatial fuel pattern, per se, might show promise, but guidelines for desirable patterns are not apparent. Monte Carlo approaches that simulate many fires might show promise in accounting for the uncertainty of fire origin and behavior, but heuristics for finding near-optimal solutions have yet to be developed and the basic computing time necessary to simulate an adequate number of fires may be prohibitive. Clearly, much additional work is needed on all aspects of the spatial and dynamic management of fuels at the landscape scale.

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CHAPTER VI

FINAL REMARKS

Alexandrian et al. (2000) reported that both the number and area of forest fires throughout the Mediterranean basin almost doubled since the 1970's. This is largely a Northwest Mediterranean problem where recent socio-economic and demographic trends have led to an increasing severity of forest fires (Velez 2006). Climate change scenarios suggest the reinforcement of this severity (Kjellstrom 2004) Portugal has been the country most affected by forest fires in the Mediterranean. In the period from 1980 to 2004 there was a fire for every 20 ha of land and forest fires burned the equivalent of one third of the territory (Pereira et al. 2006). Fire suppression costs have increased substantially and yet apparently have not been able to reverse this trend. Thus the development of an approach and of tools to help integrate forest and fire management and optimize forest-wide socioeconomic and ecological objectives while sustaining effective fire protection levels will have a country-wide impact. These tools will contribute to enhance current forest plans for all Portuguese regions and current forest fire protection plans for all local municipalities. The information provided will further contribute to enhance national fire prevention policies that impact all regions.

As its knowned, forest fire is nowadays an important problem in Portugal. For this reason several studies have already addressed the characterization of wildfires and have tried to model fire ignition probabilities (Vasconcelos et al., 2001; Catry et al., 2009; Moreira et al., 2009; Marques et al., 2011a, 2011b). Most studies focused on variables that either are uncontrollable by forest managers (e.g. climate, topography) or that may mostly support strategic land allocation decision-making (e.g. cover type zoning). Even though these studies provide valuable information, forest management requires further information, i.e. models that may help foresters design prescriptions to decrease the probability of wildfire occurrence. Our study develops a planning oriented model that includes variables that are easily measurable and controllable by forest managers (e.g. shrub fuel load, basal area and quadratic mean diameter). It further extended former studies developing a model for different species and stand structures. This model can help delineate adequate prevention policies (e.g. shrub cleanings, thinning, final cuttings). This research confirmed the potential of the logistic regression approach to develop a fire probability model for pure maritime pine stands in Portugal, like Leiria National Forest.

Forest and fire management planning are usually carried out independently one to each other. This research has developed a model to predict fire probability, which gives the possibility to integrate forest practices to reduce stand level fire risk. Thus, this model may have practical application for management to decrease fire hazard, by promoting less fire-prone stands. Moreover, the results are instrumental to understand the influence of certain variables on the probability of wildfire occurrence in pure maritime pine stands. For instance, the control of biomass load and stocking allows the forest manager to include reduction of the fire risk as an alternative objective in forest management planning. In addition to the reduction in fire risk, several studies also indicate that fuel treatments (i.e. reduction of fuels in forests) may change wildfire behaviour and enhance the effectiveness of fire suppression tactics (e.g. Mercer et al., 2008).

The proposed model may be integrated within a growth and yield model that predicts the stand development (Hanewinkel et al., 2010). This model may be used to predict the probability of a wildfire to occur if there is an ignition. Thus, it should be applied after using a wildfire ignition model such as the ones developed by Catry et al. (2009) or Vasconcelos et al. (2001). In the framework of forest management planning, first the growth and yield model would be used to predict stand development. At each step of the growth simulation, if an ignition occurs the simulator estimates the probability of wildfire occurrence. At that point there are two options: a dichotomous variable (i.e wildfire occurs or does not occur) or just information about the probability. If a dichotomous variable is needed then the probability of wildfire occurrence is compared to a cut-point to decide whether the wildfire occurs or not (e.g. González-Olabarria and Pukkala, 2010). However, if only information on the probability of wildfire occurrence is required, no cut-point would be used (e.g. Pasalodos-Tato et al., 2010; Garcia-Gonzalo et al., 2011b).

If the approach followed needs to calculate whether a wildfire occurs or not over the planning horizon a cut-point must be defined and compared to each estimated probability (Hosmer and Lemeshow, 2000). This cut-value then transforms a continuous probability in a dichotomous value (0 or 1). In any case, risk quantification provided by the selected model may help forest managers design adequate fuel and stand management strategies.

During model development, a wide range of variables that influence the probability of the occurrence of forest fires was considered. However, more variables such as wind speed or maximum temperatures could be addressed by future research. In addition, it is important to underline that wildfire occurrence in a stand is not only dependent on the intrinsic characteristics of the stand, but it is also influenced by the landscape structure. Stands are often burnt by wildfires that started in neighbouring stands, thus it would be important to also take into account the biometric variables of neighbouring stands (Moreira et al., 2009; Silva et al., 2009).

Post-fire mortality has been studied using a variety of direct and indirect methods (e. g. Fowler and Sieg 2004). Yet the use of these methods in forest management planning is constrained by its cost-effectiveness and the difficulty to predict accurately the variables they use (e.g. tissue damage, relative humidity of air at the time of ignition). The proposed approach follows the recommendations presented by González et al. (2007) to develop models that are very suitable for forest management planning.

The logistic modeling approach has been used earlier for predicting tree-mortality as a consequence of wind damage (Lohmander and Helles 1987, Jalkanen et al. 2000), prescribed fire (Botelho et al. 1996) and wildfire (Regelbrugge and Conard 1993, Harrington 1993, Stephens and Finney 2002, Beverly and Martell 2003, McHugh and Kolb 2003, Rigolot. 2004, Gonzalez et al. 2007). The modeling approach in different steps has been also used to model natural tree mortality (Fridman and Stahl 2001, Alvarez Gonzalez et al. 2004). This research confirmed the potential of the proposed approach to develop mortality models that may be used in forest planning (Reinhardt and Crookston 2003, González et al. 2007, Hyytiainen and Haight 2009).

Our models are developed to predict mortality if a fire occurs. Compared with previous models for post-fire tree mortality, our models do not use tissue damage or fire severity as predictors. This follows the approach presented by Gonzalez et al. (2007). However, some of the variables included in our models have a clear correlation with fire behavior. This is the case of slope; steeper slopes increase the expected damage. This is in concordance with other studies and may be explained by an easier transfer of heat uphill, “chimney” effects, and lower fuel moisture (Gonzalez et al. 2007, Hyytiainen and Haight 2009).

Biometric variables that influenced post-fire mortality included average tree size (average diameter, or quadratic mean diameter or cm), indicators of density as basal area (G), a variable non linear related to stand density (G/Dq) and a measure of competition (BAL). The coefficients of biometric variables in stand-level mortality models are also in concordance with findings from other studies (e.g. Pollet and Omi 2002, Gonzalez et al. 2007). Extensive model testing led to the rejection of other biometric variables as predictors of stand-level damage after a wildfire.

Prediction and classification do not follow the same pattern, so a compromise must be reached between good classification of dead trees and good prediction of mortality and survival rates when choosing a threshold level (cut-point) (Crecente-Campo et al. 2009). The advantage of the three-step methodology used in this study, compared to other traditional approaches is the possibility of detecting stands where no mortality occurs. Otherwise, traditional models always generate some mortality for all plots (Fridman and Stahl 2001).

Post-fire mortality models are a valuable forest management planning tool (Gonzalez et al. 2007). This research encompassed the development of post-fire stand damage and tree mortality models for improved forest planning in Portugal. They provide information about the impact of forest fires under alternative forest conditions. Thus, we may conclude that these models are instrumental to designing silvicultural strategies that may decrease the damage caused by wildfires. The usefulness of post fire models in forest planning depends on the information they may provide about the impact on mortality of variables whose future value may be estimated with reasonable accuracy. The presented post-fire stand damage and tree mortality models are based on variables that are under the control of forest managers (e.g. forest density, mean diameter).

Thus we may further conclude that the presented models (fire risk probability and mortality models) they can be used to integrate effectively fire risk into forest management planning and used to design landscapes more resistant to fires.

The optimal location of fuel treatments across a landscape remains a central challenge in fire management decision-making processes. Landscape conditions are usually heterogeneous, and scheduling fuel treatments by following regular patterns may not be an efficient strategy for reducing expected fire losses.

The model developed here illustrates a potential approach to integrate these practical management concerns to improve the overall effectiveness of landscape level fuel treatments. By implementing the complete enumeration approach, we separated the probability of fire in each cell into the probability that the fire would start in that cell and the probability that the fire would spread to that cell from an adjacent cell. The integer programming model used the calculated probabilities based on the interrelationships between cells regarding fire risk for different wind directions. The model can then incorporate the potential fire-loss information into the decision-making process to determine the optimal spatial distribution of fuel treatments on the landscape. In addition to the benefits of protecting the forest from potential loss from fires, the cost and feasibility of scheduling certain types of fuel treatments at specific locations of a landscape can also be considered. Clearly, much additional work is needed on all aspects of the spatial and dynamic management of fuels at the landscape scale.

This Ph'D working plan facilitated information sharing among researchers with different expertise in forest and fire management, forest managers and forest owners so that decision support tools may be developed to help integrate forest and fire management planning activities currently carried out mostly independently of each other. This was instrumental to develop fire prevention activities and policies that may effectively address those critical trends. Fire risk and fire hazard models provided information for forest cover type's vulnerability analysis. They further enabled the assessment of forest operations impacts on this vulnerability. Forest-level models provided information to help the public administration and the forest industry layout forest activities in their properties so that the resulting landscapes are more resistant to fires. They will further enhance collaborative planning by forest owners to address the risk of fire in forested landscapes where property fragmentation is predominant. Moreover, they will provide needed information for regional forest planning. Public policy acknowledges that Portuguese forest area should expand from 1/3 to 2/3 of the national territory. Fire risk impacts negatively the attractiveness of forest investment and it contributes to the lack of active management on the current area. This will be instrumental to promote active management in the current forest area and to sustain an adequate expansion. It will thus contribute to decrease the levels of destruction of urban and rural property and of damage of forest resources by wildfires. It will contribute to sustain the forest industry and the Portuguese economy. It will contribute to decrease the threat to human lives by wildfires.

CHAPTER VII

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