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A stochastic fire spread model for north Patagonia based on fire occurrence maps

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ABSTRACT

Understanding fire spread in different ecosystems is of fundamental importance for conservation, management and anticipating the effects of environmental changes. Tailoring existing fire spread models to particular landscapes is challenging because it demands a substantial data collection effort. Here we develop an objective way to fit simple stochastic fire spread models based on readily available data from documented fire events (i.e. approximate ignition point, preexisting vegetation, final perimeter, topography, and average wind direction). We use a simulation-based approach founded on Approximate Bayesian Computation, which allows for a thorough exploration of parameter space as well as the quantification of uncertainty around best estimates. As illustration, we use data from nine fire events that occurred during dry years in northern Patagonia, Argentina. We found that fire spreads readily in shrublands, while forests tend to act as firebreaks. Topography has a strong effect not only because fire moves easily upslope but also because it modulates wind direction. Finally, aspect affects fire spread mainly in forests, probably due to its effects on fuel moisture. Simulating fire spread sampling parameters from the approximated joint posterior distribution resulted in individual fires roughly similar to the ones used for model fitting. Furthermore, the fitted model was able to produce simulated fire-size distributions in good agreement with the historical record for dry years in Nahuel Huapi National Park, Patagonia. The approach presented here can be used in places where standard fuel models have not yet been developed.

1. Introduction

Wildfire is a global contagious spatial process with important ecosystem and societal effects as well as biosphere feedbacks. Rapid shifts in fire regimes, amplified by global environmental change, call for an assessment and understanding of the suite of processes that govern fire spread across different ecosystems (Bowman et al., 2009; Stephens et al., 2013). Despite the fact that fire is governed by common biophysical processes, its occurrence and spread in particular landscapes are usually hard to predict. This is because fire spread is controlled by complex system-specific interactions of fuel quantity and quality modified by environmental heterogeneity, climate, weather and the occurrence of both natural and anthropogenic ignitions (McKenzie et al., 2011; Stephens et al., 2013).

There have been numerous studies about the physics that rule fire spread (Rothermel, 1972; Albini, 1976; Van Wagner, 1977), and a lineage of fire modeling tools based on these mechanisms starting with BEHAVE in the late seventies (see Andrews, 2014 for a review). More recently, complex mechanistic models of fire propagation such as FARSITE and PROMETHEUS have been developed and adopted by governmental agencies (e.g. Finney, 1998; Tymstra et al., 2010). These models require detailed information such as hourly or daily weather data (temperature, humidity, wind speed and direction) as well as estimates of the load and spatial arrangement of different fuel classes (Tymstra et al., 2010). Other software such as FlamMap (Finney, 2006) are based on these mechanistic models but require few GIS layers to run but still need assigning each pixel to one of several possible "fuel model" categories as well as determining canopy height and crown bulk density. Furthermore the user has to set fixed fire spread times.

Relatively simpler fire-spread models are used in Landscape Fire Succession Models where the goal is to understand the relationship between vegetation dynamics and the fire regime and to predict the consequences of climate and land use changes (reviewed in Keane et al., 2004 and Cary et al., 2006). These models vary in whether simulated fire size is pre-determined or an emergent property of fire spread probabilities (e.g. Baker, 1993; Yassemi et al., 2008). In any case, some sort of ad hoc procedure is required to obtain fires

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similar in size, shape and the type of vegetation burned, to those occurring in the target ecosystem or cases. An objective and general methodology to calibrate fire spread probabilities is needed. Here we develop an objective way to estimate parameters of a simple stochastic fire spread model based on readily available data from documented fire events (i.e. ignition point, preexisting vegetation, final perimeter, topography, and average wind direction).

Stochastic models where fire propagate from cell to cell of a raster map according to some probability have been used extensively to study fire size distributions and landscape memory as emergent properties of both the contagion process (the spread probability) and the shape of the curve describing how fire susceptibility in a cell changes with time since the last fire (e.g. Zinck and Grimm, 2009; Kitzberger et al., 2012). Recently, Kennedy and McKenzie (Kennedy and McKenzie, 2010; McKenzie and Kennedy, 2012) fitted a stochastic fire spread model to fire-scar data from semi-arid mountain systems in Washington State, USA. Their model included a single parameter for fire propagation representing bottom-up controls such as fuel types, topography and so on, and a parameter for mean fire size aimed at capturing top-down (climatic) controls on the fire regime. A third parameter controlled the probability of trees getting scarred during fire events. The work we present here extends their approach by explicitly including the effect of several covariates on fire-spread probability. Another difference is that we do not impose an effective upper limit to the size of simulated fires. We only considered fires that occurred during dry years, when fire-spread is maximal and with minimum differences in top-down controls on fire size. Finally, we use a simulation-based approach founded on Approximate Bayesian Computation, which allows for a thorough exploration of parameter space as well as the quantification of uncertainty around best estimates (Beaumont, 2010; Hartig et al., 2011).

Our study area is located in northern Patagonia where the vegetation has been strongly influenced by both natural and anthropogenic fires (Veblen et al., 1999, 2003; Gowda et al., 2012). Biotic and abiotic variables strongly influence the occurrence of fires across northern Patagonian landscapes but their effects on fire spread have not been yet modeled nor formally quantified. Our goal here is twofold; we want to develop ways to fit stochastic fire-spread models to available fire map data, and we aim at increasing our understanding of the determinants of fire regimes in northern Patagonian forests so that we can eventually build appropriate Landscape Fire Succession Models for this area.

2. Methods

2.1. Study area

The study area includes Nahuel Huapi and southern Lanin National Parks, located in the northern Patagonian Andean region, at 40°–41°30′ S latitude (Fig. 1). Terrain is mountainous, with typical U-shaped valleys and steep slopes formed by glacial activity. Soils are derived from volcanic ash deposits overlaying the glacial topography. Average annual precipitation varies between 3000 mm in the western limit to 800 mm in the eastern area, with approximately 60% of the total precipitation falling during the winter season (from May to August). Wind direction is highly constant during the fire season with ca. 78% of days coming from NW (55.4%) or WNW (22.8%; 2010–2013 data Bariloche Airport.) Water deficits are severe in late spring and summer (Paruelo et al., 1998).

In areas with more than 1000 mm of annual rainfall, hillsides and valley bottoms up to the tree line (ca. 1600 m.a.s.l.) are dominated by dense and continuous forests and shrublands. Subalpine forests of the deciduous *Nothofagus pumilio* occur above 1000–1100 m.a.s.l., and the lowland rainforests are dominated by the evergreen *Nothofagus dombeyi* which forms monospecific, mesic forests or mixed stands with the conifer *Austrocedrus chilensis* at drier sites. Dense, tall shrublands occur at sites that are either edaphically unsuitable for the development of tall forest or as successional communities that develop after burning of tall forest (Veblen et al., 1992). In the wet western district, tall shrublands and dense woodlands of the small deciduous tree *Nothofagus antarctica* occur mainly in valley bottoms where soil drainage is poor. In the central and eastern areas, tall shrublands *of N. antarctica*, the bamboo *Chusquea culeou*, and numerous other small trees and tall shrubs (*Schinus patagonicus*, *Embothrium coccineum*, *Maytenus boaria*, *Diostea juncea*, *Lomatia hirsuta*, and *Berberis* spp.) occur from low to high elevations, and around tall forests. Despite the fact that forest and shrublands can vary in species composition, we consider them as the main types of vegetation in the area, showing very different flammability (Mermoz et al., 2005).

2.2. Fire maps

For model parameterization and testing we compiled mapped fires that matched the following conditions: (1) approximated location of point of ignitions were known, (2) fire occurred during dry years (rainfall between October and December below 0.5 SD the historic mean 1950–2008), and (3) 30 m resolution vegetation maps of the areas affected by fire were available. Only nine recorded fires matched these criteria and were used for parameterization. Elevation, aspect and slope maps were obtained from an ASTER GDEM digital elevation model (30 m resolution, http://gdem.ersdac.jspacesystems.or.jp/). Pre-fire vegetation maps were available from ortho-rectified aerial photos (scale 1:20,000 and 1:60,000) (Mermoz et al., 2005). Fire maps were obtained by comparing pre and post fire Landstat TM images, except for Tristeza 1957, which was mapped from aerial photos (Mermoz et al., 2005).

2.3. Modeling fire spread

Our goal was to model how topography, fuel type and other factors such as wind direction modify fire spread across the landscape. All landscape variables were discretized into raster maps of 30 by 30 m square cells. Once ignited, a landscape cell could propagate fire to each of its 8 neighboring cells. The probability of propagation (p) was determined by a function of fuel type, aspect, relative wind direction and slope of the target cell:

$$p = \frac{0.5}{1 + \exp(-(\beta_0 + I_f + \beta_2\Psi + \beta_3\omega + \beta_4\sigma))} \tag{1.1}$$

where I_f is an indicator variable equal to 1 for cells occupied by forest and 0 otherwise, so that β_0 is then the baseline (untransformed) fire-propagation probability for shrubland cells and β_1 measures the difference between shrubland and forest. If fire is less likely to propagate in forests than in shrublands, then β_1 should be less than zero. Parameters β_2, β_3 and β_4 modify propagation probabilities according to aspect (ψ), wind direction (ω), and slope (σ) respectively. We set the upper limit for this logistic function to 0.5 because larger propagation probabilities produce unrealistic fire patterns (Hargrove et al., 2000).

Aspect is known to affect fire occurrence in the study area (Mermoz et al., 2005), probably through its effects on fuel moisture. The driest conditions are on sites facing toward the NW while those facing to the SE are the moistest. Thus, for landscape cells with a slope greater than 5 degrees we computed $\Psi = \cos(\theta - 315°)$, where θ is the untransformed aspect in degrees. This relative aspect takes values of 1 for NW facing sites and −1 for those facing SE. Thus, values of $\beta_2 > 0$ indicate that fire propagate more readily in drier slopes. We did not have access to detailed wind speed and direction during