Fire disturbance and vegetation dynamics - analysis and models

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0. SUMMARY

On considering the immense changes in the natural environment due to human activity, the need to understand patterns and processes in natural ecosystems, above all succession and vegetation dynamics, is especially important. The large variety of literature, published since the beginning of the 20^{th} century, discusses theories of ecosystem dynamics in general, and of succession in particular, reflects the different understanding of the topic and at the same time the influence of ecosystem-specific observations on theory development. According to the Clementsian theory of succession (Clements 1928; 1936; Weaver & Clements 1938), natural landscapes are characterised by climax systems that result from regular successional sequences. Observations of small vegetation changes in temperate forests supported this theory. Meanwhile authors like Aubreville in 1938, Watt in 1925 and Poole in 1937 (see citation in Shugart 1984) recognised the mosaic structure of landscapes in their case studies. Gleason's individualistic concept (1926; 1927; 1939, cited in Burrows, 1990) suggests that plant species live together not because of strong interdependence among species, but due to their ecological niche overlap, which form a vegetation continuum. In his understanding of vegetation as temporary and fluctuating units implies succession cannot be an orderly and hierarchical process. More observations of changes in diversity, vegetation composition and in ecosystem productivity stimulated both discussions on stability and constancy of ecosystems, and studies on the mechanisms behind vegetation change to explain mechanisms of succession (see review in Shugart 1984, and Burrows, 1990). One of the mechanisms, cyclic disturbance of vegetation, was proposed by Loucks (1970) to contribute to temporal heterogeneity in vegetation and thus plant diversity. Ecosystems, which face frequent disturbance, are characterised by plant communities that cannot be explained by site characteristics alone (White 1979). He outlined the gradient of disturbances relative to their size and effects on plant communities and the biotic role in disturbances – not only abiotic, but also biotic factors can cause disturbance – to show that existing concepts on plant communities had to be re-examined. Small-scale disturbance of plant populations, their temporal and spatial variability as well as long-term climate changes, contradicted the linear pathway of succession of an entire forest ecosystem and underlined the non-equilibrium nature of vegetation (Burrows 1990; Shugart 1984). The scientific insight to these observations was renewed decades later in theories of Shifting-Mosaic (Steady State) concept (Bormann & Likens 1979; Mueller-Dombois 1986; Remmert 1991) and of vegetation dynamics (Burrows 1990; Glenn-Lewin *et al.* 1992).

Starting from the recognition of wave-generated heterogeneity in landscapes (e.g. Loucks 1970; Sprugel 1976) the causes of succession, especially their external forces came into focus. The evolving understanding of disturbance pattern and processes and their function in ecosystems can be drawn from the various definitions of disturbances that have been published in the past. They do not only destroy biomass (Grime 1979), they change community or ecosystem composition, structure and cycles (Mooney & Auclair 1983), resources, substrate availability and even the physical environment (White & Pickett 1985). Being a spatially, temporally discrete event of a given magnitude (the dimensions of disturbance) disturbance opens sites for regeneration and therefore leads to the rejuvenation of vegetation (Sousa 1984; van der Maarel 1993). The remaining resources in the opened sites determine plant competition as well as modus and rate of regeneration. In the case of stable environmental conditions, this might maintain pre-disturbance conditions or as a consequence of changed site conditions result in a new vegetation composition (Grime 1979; Remmert 1992; Sousa 1984). These characteristics of disturbance events allow us to distinguish them from stress and environmental fluctuations (see discussion in Burrows 1990). In summary, all characteristics examined by the authors define natural disturbances. A natural disturbance is therefore a discrete event that - characterised by its temporal, spatial and magnitudinal dimension - changes the status of a patch in an ecosystem with the effect that, depending on the remaining resources, the competition about it is again initiated and rate and modus of regeneration are newly determined. The mean characteristics of the disturbance dimensions allow the definition of a certain disturbance regime.

Studies of the role of disturbance in vegetation or ecosystems showed that disturbances are an essential and intrinsic element of ecosystems that contribute substantially to ecosystem health, to structural diversity of ecosystems and to nutrient cycling at the local as well as global level (Booysen & Tainton 1984; Gill 1981b; Goldammer 1992; Mooney *et al.* 1981; Pickett & White 1985; Reynolds & Tenhunen 1996; Sousa 1984; Suffling *et al.* 1988; Watt *et al.* 1990). For a more detailed description of disturbances in different vegetation zones see Thonicke (1998) and Walker (1999). By starting new successional cycles in newly opened areas vegetation is rejuvenated, the variability in the spatial, temporal and magnitudinal characteristics create a mosaic of different environmental conditions that offer plant communities, which fit into these environmental niches, a base for establishment at the specific site and persistence in the ecosystem. In ecosystems with dominant disturbance agents, such as fire, plants developed adaptation strategies to mitigate or survive. This special adaptation is essential in the context of environmental changes, when climate or human impacts force changes in the disturbance regime that can have an effect on vegetation composition and its dynamics.

First attempts to model the impact of disturbances on vegetation have been made in gap models. Studies on the frequency and intensity of disturbances outlined the influence disturbance has on vegetation composition and dynamics (see models in Keane *et al.* 2001; Shugart 1984). Disturbances were considered as static events of a relatively constant regime, first not specified, but later adapted to a certain disturbance agent such as fire (e.g. Bergeron & Dansereau 1993; Johnson & Gutsell 1994; Suffling 1995). As Clark (1989) analysed for fire disturbance in North America, disturbance regimes are not constant, but variable over time, as the driving environmental conditions vary. This gave the objective to investigate processes of specific disturbance agents to develop process-oriented, agent-specific models. Physical models of fire behaviour were developed already in the 1970s (Albini 1976; Rothermel *et al.* 1986) to support the decision-making process for fire fighting. These instruments have been influencing the development of many process-oriented fire models (see review of fire models in Gardner *et al.* 1999). Fire has been modelled as single events on small spatial and temporal scales using physical models. When modelled at larger scales, compromises between statistical and mechanistic approaches have been made. Some investigations into the effects of changing environments on fire considered only one aspect of the fire regime (lightning (Price & Rind 1994), fire season (Flannigan & van Wagner 1991), fuel loads (e.g. Bessie & Johnson 1995; Keane *et al.* 1996b) or fire weather (e.g. Flannigan *et al.* 1998)). Models describing the fire regime by combination of stochastic and mechanistic fire-process functions were applied to climate change scenarios (e.g. Gardner *et al.* 1996).

A combined model to describe effects and feedbacks between fire and vegetation became relevant as changes in fire regimes due to land use and land management were observed and the global dimension of biomass burnt as an important carbon flux to the atmosphere, its influence on atmospheric chemistry and climate as well as vegetation dynamics were emphasized (Crutzen & Goldammer 1993; Gardner *et al.* 1999). The existing modelling approaches would not allow these investigations. As a consequence, an optimal set of variables that best describes fire occurrence, fire spread and its effects in ecosystems had to be defined, which can simulate observed fire regimes and help to analyse interactions between fire and vegetation dynamics as well as to allude to the reasons behind changing fire regimes (e.g. FIRE-BGC, Keane *et al.* 1996b). Limited in application to different spatial and temporal scales and to other study regions, a general model that can support estimates of fire regimes and biomass burning at the global scale was needed. Possible strategies and requirements have been discussed Gardner (1999) and

Fosberg (1999). One approach would be to take existing fire models and generalise them in terms of process description that are still relevant at broader scales, but also valid in a variety of ecosystems that would allow global simulations of fire and vegetation. However, dynamic links between vegetation, climate and fire processes are required to analyse dynamic feedbacks and effects of changes of single environmental factors. The possible strategy to combine fire history and fire hazard function, as discussed in Fosberg (1999) appeared to be less flexible for the investigation into changes of the fire-vegetation interaction.

This leads us to the point, where new fire models have to be developed that allow the investigations, mentioned above, and can help to improve our understanding of the role of fire in global ecology. The following questions are the focus of this thesis, its major findings are then summarised:

- What forms a minimal set of major fire drivers, dominating fire occurrence and effects, that allow adequate simulation of global historic fire pattern, and thus suitable for climate change studies?
- What are important fire processes that drive human-dominated fire regimes at the regional scale, especially their temporal pattern?
- How does vegetation composition and fuel characteristics impact fire? What is the influence of fire management on fire regimes?
- What are the climate change effects on fire regimes and fire-vegetation interactions in major ecosystems? What is the role of vegetation dynamics for changes in fire regimes under climate change conditions?

A fire model was developed that forms a compromise between fire history and physical models and, after incorporation into a Dynamic Global Vegetation Model, is sufficiently general for applications at the global scale thus alluding to the influence and impact of fire on other components of the Earth system (**see chapter 1**). The spatial and temporal scale required, define the level of explicitness for the model development. The critical status of soil moisture and its persistence over time (fire season) together with vegetation-specific moisture of extinction and the resistance of vegetation types to survive a fire form the basis of the **Global Fire Model (Glob-FIRM)**. These elements were identified as the minimal set to describe fire pattern at the global scale. Simulation experiments have been performed under historic climate conditions and are validated against observations. The results show that it is a suitable tool to simulate fire at the global scale, with minor restrictions for regions where the soil moisture status, as provided by the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM Sitch *et al.* 2002), is not adequate or strong human impacts modified vegetation composition and thus the associated fire regime. A mechanistic fire module incorporated into a dynamic vegetation model allows impact studies of inter-annual climate variability and vegetation status on fire, also under climate-change conditions, but also the impact of vegetation change on fire regimes in certain ecosystems.

Many ecosystems have human-driven fire regimes. Fire management associated with a certain land use type have caused dramatic shifts in vegetation composition towards selection of fire-adapted plants. These plants have specific features in post-fire establishment and increased fire risk through high content of volatiles. This requires an explicit modelling of fire causes, fire risk and spread to allow in-depth analysis at the regional scale, a research field, where up to now little efforts have been made. The developed tool **Reg-FIRM (Regional Fire Model, see Chapter 2)** was developed to consider human and lightning-caused fires explicitly. Human-caused fires can be classified according to land management type and associated life style, a pattern, which is assumed to change with human population density and thus a suitable descriptor of human-caused ignition potentials. Reg-FIRM was incorporated into the LPJ-DGVM and applied to the Iberian Peninsula to validate the model concept per se and its parameterisation. The assessment of both temporal and spatial dynamics shows that it is possible to describe human-caused fires at the regional level using a mechanistic approach. The description of fire-relevant processes is precise enough to simulate the observed pattern. Application to other regions should be easily possible with only minor re-parameterisations of regionally specific variables.

European forests and shrublands, although human-dominated, experience different magnitudes of fire regimes. The underlying fire patterns reflect different attempts of fire management and fire suppression that have developed under historic land use systems and forest policies. These differences in management become apparent, when Reg-FIRM is applied to Brandenburg, Germany, and Peninsular Spain, regions that are characterised by different climate, vegetation and historic fire management **(see Chapter 3)**. The importance of fire-related processes to simulate observed area burnt at the regional scale is outlined in a sensitivity analysis. Human and natural causes, climatic fire danger, fuel characteristics, which depends also on vegetation composition, and wind speed are of particular importance, when inter-annual variability, variance of fire are to be simulated at the same scale as the observations. Bi-directional feedbacks between fire and vegetation are investigated in simulation experiments with different vegetation compositions. They show that vegetation composition has a larger influence on fire spread than fire ignition and change area burnt, but has less influence on the inter-annual variability and variance in number of fires and area burnt. A comparison of simulated fire pattern of the two study regions reveals the potential of Reg-FIRM to interpret causes and effects of fire management on the fire pattern that cannot easily be concluded from national fire statistics. The dominance of using fire as a land management tool in Peninsular Spain as opposed to fire suppression result in a different pattern of discrepancies between simulated and observed fire pattern as compared to only accidentally ignited forest fires that are immediately suppressed such as in Brandenburg.

Both developed fire models, optional running inside the LPJ-DGVM, were used to explore **global and regional implications of climate change (see Chapter 4)** on vegetation dynamics and fire regimes. Fire can not only influence the magnitude of net carbon exchange, but also its sign as an analysis of the components that constitute net ecosystem production has shown. This underlines the importance of including fire in carbon balance calculations under climate change conditions. A qualitative analysis of the combined effects of climate and vegetation productivity on biomass burning emissions showed how vegetation can buffer climate change effects on fire in some regions, but lead to increases in fire-related emissions in semiarid regions, where climate favours vegetation growth regularly. Here, a reduction in the length of fire season implies changes in vegetation growth from episodic to a regular pattern, which no longer limits fire spread. Increases in the length of fire season do not automatically lead to increases in biomass burnt, as seen in many ecosystems in the Northern Hemisphere. Regional climate change applications outlined the importance to consider the entire causal chain in such studies. Temporal changes in climatic fire danger can be delayed as compared to changes in annual area burnt due to the direct impact of vegetation composition and moisture conditions on fire spread. Climate driven changes in vegetation composition feed back on annual area burnt as seen in Peninsular Spain, where increases in grass cover caused dramatic increases in annual area burnt in subsequent years.

In conclusion of the thesis, one can state that moisture conditions, its persistence over time and fuel load are the important components that describe global fire pattern. If time series of a particular region are to be reproduced, specific ignition sources, fire-critical climate conditions and vegetation composition become additional determinants. Vegetation composition changes the level of fire occurrence and spread, but has limited impact on the inter-annual variability of fire. The importance to consider the full range of major fire processes and links to vegetation dynamics become apparent under climate change conditions. Increases in climate-dependent length of fire season does not automatically imply increases in biomass burnt, it can be buffered or accelerated by changes in vegetation productivity. Changes in vegetation composition as well as enhanced vegetation productivity can intensify changes in fire and lead to even more fire-related emissions.

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1. THE ROLE OF FIRE DISTURBANCE FOR GLOBAL VEGETATION DYNAMICS: COUPLING FIRE INTO A DYNAMIC GLOBAL VEGETATION MODEL

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Abstract

Disturbances from fire, wind-throw, insects and other herbivory are, besides climate, CO_2, and soils, critical factors for composition, structure and dynamics of most vegetation. To simulate the influence of fire on the dynamic equilibrium, as well as on potential change, of vegetation at the global scale, we have developed a fire model, running inside the modular framework of the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM). Estimated litter moisture is the main driver of day-to-day fire probability. The length of the fire season is used to estimate the fractional area of a grid cell which is burnt in a given year. This affected area is converted into an average fire return interval which can be compared to observations. When driven by observed climate for the 20th century (at a 0.5° longitude/latitude resolution), the model yielded fire return intervals in good agreement with observations for many regions (except parts of semi-arid Africa and boreal Siberia). We suggest that further improvement for these regions must involve additional process descriptions such as permafrost and fuel/fire dynamics.

Keywords: fire, disturbance, fire model, fire return intervals, fire season, dynamic global vegetation model, vegetation dynamics

Introduction

Disturbance, defined as irregularly occurring destruction of vegetation structure, whether natural or human-induced, plays an integral part in shaping global vegetation (Glenn-Lewin & van der Maarel 1992; Sousa 1984). Disturbance usually acts as an additional component, which together with climate and soil conditions drives vegetation structure and functioning. Sites opened by disturbance allow regeneration of vegetation, thereby often maintaining vegetation composition and successional cycles. Depending on the post-disturbance environment, disturbance can also accelerate changes in vegetation composition, possibly resulting in different vegetation dynamics and altered biodiversity. Consequently, inclusion of disturbance in vegetation models is required to correctly simulate vegetation dynamics.

In this study we consider fire as a primary disturbance agent, because of its overwhelming importance in many ecosystems – other agents could be considered using a similar approach. The long history of fire ecology and fire modelling studies provides process understanding and suitable algorithms for including fire disturbance in a vegetation model (see, e.g. Gardner *et al.* 1999).

The role of fire for natural vegetation

Fire is one of the major disturbance agents on the global scale, affecting biogeochemical cycling, playing an important role in atmospheric chemistry and the global carbon cycle. Globally, approximately 3.9 Gt of carbon (Gt C) are released annually into the atmosphere through biomass burning (Andreae 1991), equivalent to over 70% of the annual anthropogenic fossil fuel emissions. With its inherent sensitivity to climatic conditions, and with the prospect of rapid future climate change, fire has been a focus of intensive investigation in recent years (Levine 1996). Vegetation composition and biogeochemical cycling in many European ecosystems are strongly affected by fire and therefore must be included in any European regional study (Sykes *et al.* 2001).

The season of burning and the time between recurring fires determine the plant species composition through selection in most ecosystems. In semi-arid and arid grasslands the season of burning determines the final assemblage of C_3 and C_4 grasses and forbs, whereas the length of the fire return intervals governs the invasion of woody plants on a site, given sufficient precipitation (Bragg 1995). By initiating successional cycles, a larger number of species can often be maintained, as is observed in tropical rainforests (Goldammer 1992), Mediterranean-type ecosystems (Keeley & Keeley 1988; Walter & Breckle 1991a), boreal forests (Treter 1993; Walter & Breckle 1991a) and in the transition zone between boreal and temperate deciduous forest (Burrows 1990).

The nutrient cycle is accelerated by fire, with the formation of ash, rapidly mobilising nutrients for the plants regenerating in the newly opened understory. This is important in ecosystems with long decomposition cycles, e.g. in the boreal forest (Treter 1993), or where soils are nutrient poor, e.g. in South-African fynbos (Kruger & Bigalke 1984), Californian chaparral (Rundel 1981; Walter & Breckle 1991a) or the African savanna (Scholes & Walker 1993). A changing fire regime may cause an overall loss of nutrients in the ecosystem.

Fire can influence the hydrological cycle and soil chemistry by transferring heat into the soil, thereby changing microclimate and soil conditions. In the boreal forest, removal of the litter layer leads to intensified soil heating during the growing season, increasing the melting depth in permafrost soils, and thus tree rooting depth (Treter 1993). In Mediterranean-type shrublands, oxidation and erosion of soil organic matter due to intense fire affects water holding capacity and erosivity (Christensen 1994).

In the long history of human intervention in nature, different and changing interests have influenced fire regimes, and hence vegetation. The existence of ecosystems such as the African savannas, the Brazilian cerrado, many Mediterranean-type ecosystems and Prairie grasslands can not be adequately explained without the consideration of human burning activities, which shifted the periodicity of fire, consequently changing vegetation composition (Bourlière & Hadley 1983; Coutinho 1990; Davis & Richardson 1995; Schüle 1990). After a change in management, previously adapted plants may no longer be adapted to the new fire regime, as changes in pastoralism (e.g. Davis & Richardson 1995), or North-American fire suppression policy have shown. It was initially believed that suppression would reduce fire hazard and economic loss, but in fact merely demonstrated how fire is a key driver in vegetation dynamics. Fire suppression reduced the number of medium to large-scale fires, with the effect that a small number of huge fires account for 98% of the total area burnt (Stocks 1991). Such a change in fire dynamics has important influences on stand-age structure, e.g. in the western Canadian boreal forest (Johnson *et al.* 1998).

Simulating fire within a Dynamic Global Vegetation Model

Many fire models have been developed for various spatial and temporal scales and application purposes. The state-of-the art process-oriented fire models (e.g. Albini 1976; Keane *et al.* 1996b) include processes of topography-dependent lateral fire spread and are designed for GIS systems at relatively fine scales (1 km or less). They are therefore not directly applicable on the coarse 0.5° (≈50 km) scale, i.e. cannot be directly incorporated into Dynamic Global Vegetation Models (Cramer et al., 2001). Modelling fire at this coarse spatial scale requires a top-down approach, which considers only major regional characteristics but is sufficiently general to be applicable to every grid cell on the global scale.

To link fire regime and its effects on vegetation dynamics we have developed a general fire model Glob-FIRM (Global FIRe Model) which has been incorporated into the LPJ-DGVM (Sitch *et al.* 2000; Smith *et al.* 2001). Our approach is a compromise between the fire history concept (a statistical relationship between the length of the fire season and area burnt) and a process-oriented model methodology (estimation of fire conditions based on soil moisture). A relationship between the daily litter moisture status and the length of the fire season is derived from observed data to develop the function for probability of fire occurrence. Observed data is also used to calibrate a function which describes the relationship between the length of the fire season and annual area burnt. The fire module is validated against observations of typical fire return intervals.

Human-changed fire regimes as well as other land use impacts are not considered in the LPJ-DGVM at the present stage. As a first step, the impact of simulating fire is studied for natural conditions to prove the applicability of this model before combining impacts of fire and land use on vegetation. The coupled model (i.e. the DGVM including the fire module) is used to study the interaction between fire and global vegetation over the historic period 1901-1998.

Model description

The Lund-Potsdam-Jena Dynamic Global Vegetation Model

The LPJ-DGVM (Sitch *et al.* 2000; Smith *et al.* 2001) was constructed in a modular framework. Individual modules describe key ecosystem processes, including vegetation establishment, resource competition, growth and mortality. Vegetation structure and composition is described by nine plant functional types (PFTs) which are distinguished according to their plant physiological (C_3, C_4 photosynthesis), phenological (deciduous, evergreen) and physiognomic (tree, grass) attributes. The model is run on a grid cell basis with input of soil texture, monthly fields of temperature, precipitation and percentage sunshine hours. These fields are interpolated to obtain daily values for processes, which are calculated on a daily time-step, such as evapotranspiration and soil moisture status. Each grid cell is divided into fractions covered by the PFTs and bare ground. The presence and fractional coverage of an individual PFT depends on its specific environmental limits, and on the outcome of resource competition with the other PFTs.

The two layer soil water balance model is based on Haxeltine and Prentice (1996a). Moisture in each layer, expressed as a fraction of water holding capacity, is updated daily. Percolation from the upper to the lower layer, and absolute water holding capacity are soil texture dependent.

Establishment and mortality are modelled on an annual basis. Plant establishment, in terms of additional individuals, depends on the fraction of bare ground available for seedlings to successfully establish. Natural mortality is taken as a function of plant vigour, and corresponds to an annual reduction in the number of individuals.

Dead biomass enters the litter pool, and the soil pools. Mortality also occurs due to disturbance (explained in the fire model section below).

The fire module Glob-FIRM in the LPJ-DGVM

Fire is modelled as the combination of fire occurrence and its effects. In order to model both processes on a global scale some simplifying hypotheses are made. Firstly, fire occurrence is dependent only upon fuel load and litter moisture (i.e. the amount of dry material available), which combines both the influence of climate and vegetation. Ignition is assumed to take place sooner or later, without specific consideration. Secondly, fire effects are mainly driven by the length of the fire season and the PFT dependent plant resistances only.

Fire occurrence

To start a fire the fuel has to reach a minimum temperature at which it ignites and combustion will start if there is sufficient fuel present at the site. If fuel moisture is above a certain level, then all available energy in the pre-heating process is consumed to vaporise water, thus ignition temperature is not reached and ignition, either spontaneously or due to fire spread, fails (Viegas 1997b). The threshold of fuel moisture content above which a fire would not spread is named the 'moisture of extinction'. Albini (1976) reports moisture of extinction for dead fuel in the range of 15 to 30% in temperate ecosystems, similar to values reported by Kauffman & Uhl (1990) in the tropical rainforest, Viegas (1998) in the Mediterranean Basin, and De Ronde (1990) in tropical industrial pine plantations.

Even if the fuel is dry enough, a fuel load of less than approx. 200 gC m^{-2} reduces fire spread to zero (Schultz 1988). In ecosystems with high climate variability and consequently fluctuating vegetation productivity, periods with a discontinuous fuel bed, i.e. insufficient fuel load, are characterised by little or no area burnt despite favourable climatic conditions. Above the threshold, fuel load is sufficient to sustain fire and is therefore not limiting fire spread. Instead climate conditions become more important. This is in agreement with observation from the boreal forests (Bessie & Johnson 1995; Schimmel & Granström 1997), where climate conditions vary more over time than available fuel load (which is in ample supply) and have a larger impact on fire behaviour and fire spread.

In our model, the influence of fuel load on fire spread and thus the final area burnt, is described by a threshold function. Below a threshold of 200 gC m^{-2} fire is not permitted. If a sufficient amount of dead fuel exists with a moisture content below the moisture of extinction, then both live and dead fuel will start to burn. As a first approximation it is assumed that an ignition source is always available, either natural or anthropogenic.

Experimental data on dead fuel (pine needles) moisture have been used to develop a function to estimate the probability of fire in a grid cell. Measurements were made by Viegas et al. (1992) in three zones in Central Portugal, differing in both climate and land use. The results of these experiments express the probability of at least one fire occurring on a day with a given moisture content. They show that, for this part of the world, the daily moisture content of dead fuel has similar dynamics and similar extinction values in all three zones. The measurements of dead fuel moisture content were corrected for a consistent bias, associated with the specific measurement technique (Viegas *et al.* 1992), using the fire behaviour model BEHAVE (Rothermel *et al.* 1986). The resulting relationship between measured and predicted dead fuel moisture content, $\tilde{m}$ and m respectively, is:

$$m = 0.4994 * \tilde{m} + 1.02 \qquad (1),$$

with $r^2 = 0.852$, where $\tilde{m}$ and m are percentages (see equation 8 in Viegas *et al.* 1992). In our fire model, m is taken as the daily moisture status in the upper soil layer.

The exponential power function (eq. 2) is used to approximate the probability of the occurrence of at least one fire in a day in a grid cell:

$$p(m) = e^{-\pi * \left(\frac{m}{m_e}\right)^2} \qquad (2),$$

where m_e is the moisture of extinction. Statistically, fire is considered to be absent when m exceeds m_e with probabilities lying within the 95% confidence interval (see eq. 3). The smooth non-linear function provides the better fit and continuity over the entire range for possible daily moisture status (m ranging from 0.0 to 1.0), than a simple linear function.

$$1 - p(m_e) = 1 - e^{-\pi} \approx 0.956 \qquad (3).$$

Eq. 2 yields good fit to the experimental data for the three zones in Central Portugal with the moisture of extinction equal to approx. 30% (see Fig. 1-1), which is well in the range of the published data.

Although the fuel composition and its surface-to-volume ratio are important parameters for fuel ignition (Albini 1993; Viegas 1997a), fuel flammability is species dependent (Heinselman 1981; Walter & Breckle 1991a). The present PFT definition is too general with PFTs not characterised in terms of fuel flammability.

This could be implemented in a future definition of fire-related subsets of PFTs. As a first approximation, m_e is kept constant at 30 % for the set of woody PFTs (e.g. trees and large shrubs) and 20 % for herbaceous PFTs in the model. The latter value is obtained by averaging observed values for grasses of different height and small shrubby vegetation according to Albini (1976).

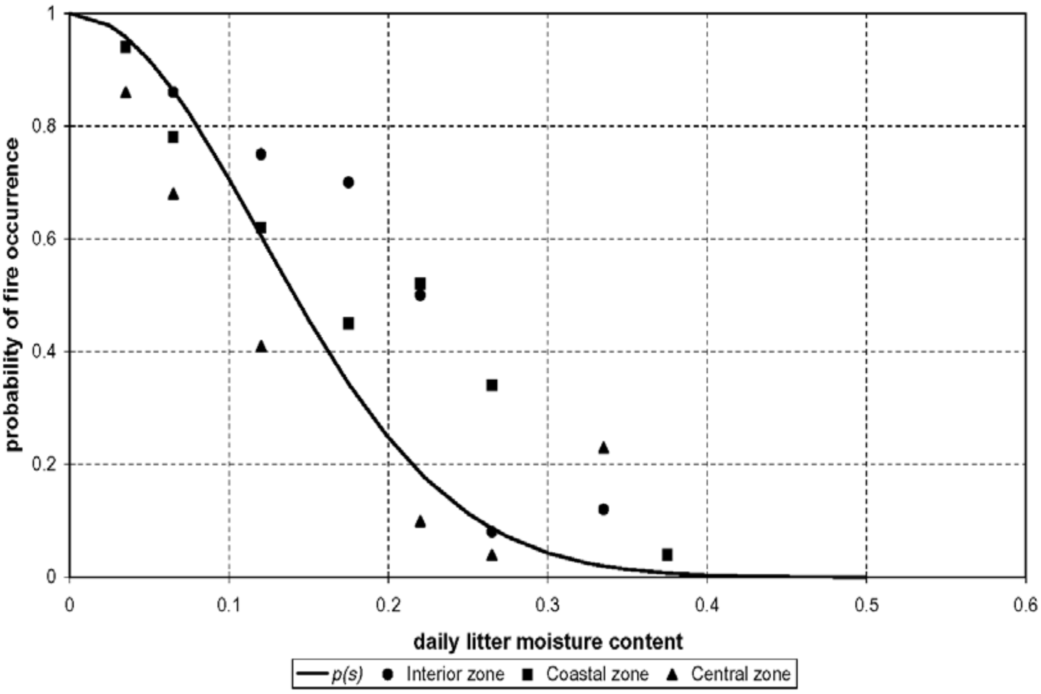

Fig. 1-1 The probability of at least one fire occurring on a day with a given litter moisture content. The points represent the experimental data for each investigated zone in Central Portugal (Viegas *et al.* 1992), averaged over the four years of investigation.

The annual length of the fire season is estimated by adding the probability of at least one fire in a day over the whole year:

$$N = \sum_{n=1}^{365} p(m_n) \qquad (4).$$

Fire effects

After ignition, the temporal and spatial change in burning conditions determine fire spread, and thus fire size. The lower the moisture content of the fuel, the less energy is consumed in the preheating process, the faster the fire can propagate. The amount of area burnt is inversely related to the daily moisture content of dead fine fuels as field measurements in Portugal have shown (Viegas 1998; Viegas *et al.* 1992). Secondly, the longer the burning conditions persist, the larger the fire can grow (Viegas 1998). This leads to the assumption that, on the meso-scale of our grid cells, the same relative annual sum s of days with particular fire conditions N (see eq. 4) with $s = N/365$ gives the same annual fractional area burnt A, expressed in terms of a fraction of the grid cell. We assume that this relationship is the same in all geographic regions.

The annual fractional area burnt A is zero, when fire conditions were absent during the year, and reaches 1 when fire conditions continue throughout the entire year. As a non-linear function, $A(s)$ can be expressed as follows

$$A(s) = s * f(s) \qquad (5),$$

where $f(s)$ is equal to 1, when $s = 1$. Assuming that $f(s)$ is

$$f(s) = e^{((s-1)*\alpha(s-1))} \qquad (6),$$

$\alpha(s-1)$ is a function, which provides the rate of exponential growth in annual fractional area burnt with increasing length of fire season. The shape of *f(s)* defines the rapid change of behaviour in *A(s)* from an almost linear slow increase of the annual fractional area burnt, when the length of fire season is less than half a year, to a rather rapid increase, when the length of fire season exceeds half a year (see Fig. 1-2).

If we combine eqs. 5 and 6, then $\alpha(s-1)$ can be expressed in terms of *A(s)* and *s*,

$$\alpha(s-1) = \frac{\ln(A(s)) - \ln(s)}{s-1} \qquad (7).$$

The observed data of area burnt, converted into fraction $\tilde{A}$, with corresponding length of fire season $\tilde{s}$, were used in eq. 7 to obtain estimates of $\alpha(s-1)$. A non-linear regression of these values satisfying the least mean square criteria gives the following formulation:

$$\alpha(s-1) = \left(\frac{1}{0.45*(s-1)^3 + 2.83*(s-1)^2 + 2.96*(s-1) + 1.04} \right) \qquad (8),$$

which provides the fractional area burnt with length of the fire season (see Fig. 1-2),

$$A(s) = s * e^{\left(\frac{s-1}{0.45*(s-1)^3 + 2.83*(s-1)^2 + 2.96*(s-1) + 1.04} \right)} \qquad (9).$$

This formulation is based on the hypothesis that the annual fractional area burnt increases at first slowly, when the length of fire season *s* is relatively short, but increases more rapidly, when *s* approaches the entire year. The hypothesis was tested against observations, containing both annual areas burnt and length of fire season in four large regions on different continents, southern California (Minnich

1998), Portugal (Viegas 1998) and central Portugal (Viegas *et al.* 1992), and northern Australia (Russell-Smith *et al.*, 1997). The data from northern Australia is considered in two ways. First, data obtained from the Plateau in Kakadu National Park is assumed to be the most representative for natural conditions, with the least human influence on vegetation through fire management. The second is an average value for the entire National Park and for all years thereby excluding the specific influence of local vegetation. Observed areas burnt are converted into fractions of 0.5° x 0.5° longitude / latitude grid cell and were used to calibrate function (6) (see Fig. 1-2).

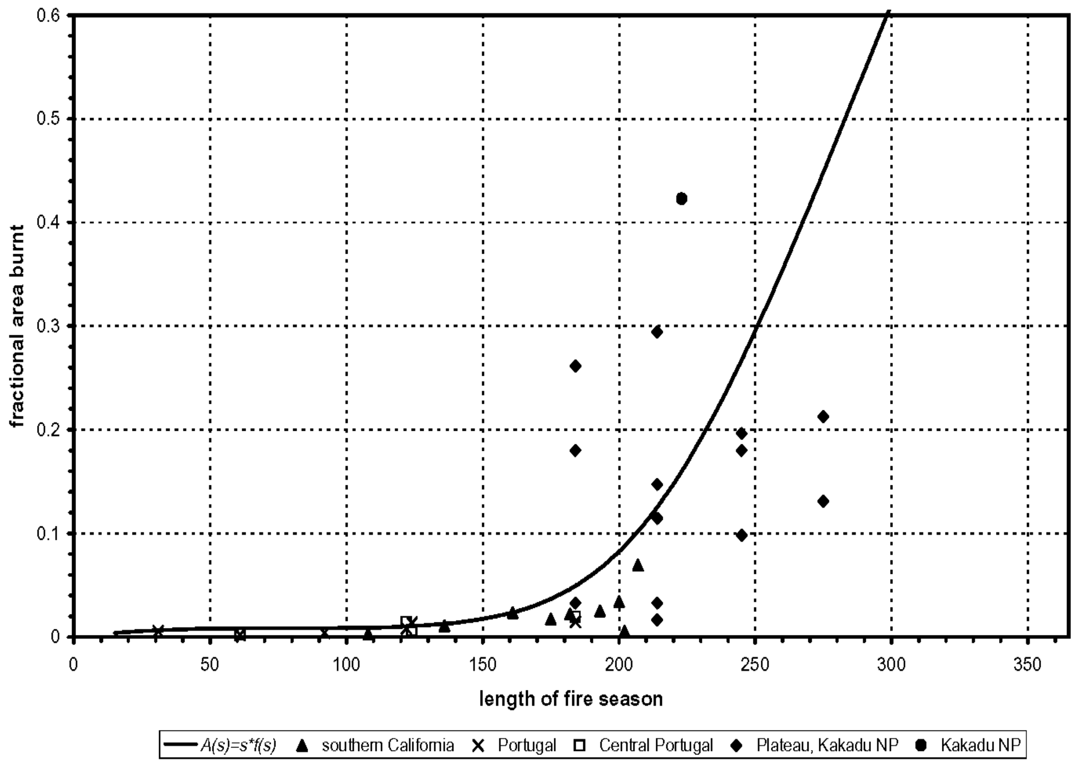

Fig. 1-2 Length of the fire season in relation to area burnt (expressed as fire fraction). Points are experimental data from Portugal (×) 1987-94, Central Portugal (□) 1987-1990, southern California (▲) 1970-80, Plateau, Kakadu National Park in northern Australia (◆) 1980-1994 and entire Kakadu National Park averaged over 1980-1994 (●), representing the length of a fire season against area burnt (expressed as the fraction of a 0.5° x 0.5° longitude / latitude grid cell). The line presents the fitted function A, see eq. 9.

The resulting fire effects on vegetation in the fractional area burnt are calculated for each PFT. The fraction of individuals killed depends upon the prescribed PFT fire resistance, which represents the PFT survivorship during a fire (see Table 1-1). In the fire model, grasses and litter are fully consumed. Interannual variability in climate, area burnt and therefore vegetation mortality provides bi-directional feedback between vegetation and fire.

Data

For the historical simulation we used the CRU05 1901-1998 0.5° x 0.5° longitude / latitude monthly climate data, provided by the Climate Research Unit, University of East Anglia, UK. Historical CO_2 concentrations were derived from ice core and atmospheric measurements (Enting *et al.* 1994). Soil texture information was obtained from the FAO soil data set (FAO 1991).

PFT	Fire resistance (%)
Woody	
Tropical broad-leaved evergreen	12.0
Tropical broad-leaved raingreen	50.0
Temperate needle-leaved evergreen	12.0
Temperate broad-leaved evergreen	50.0
Temperate broad-leaved summergreen	12.0
Boreal needle-leaved evergreen	12.0
Boreal summergreen	12.0
Grasses	
C3 grass	0.0
C4 grass	0.0

Table 1-1. PFT parameter values for fire resistance

A global validation of length of fire season or area burnt over long time series is rather problematic due to the limited data available. Most of the fire statistics are gathered over administrative units, which often integrate different vegetation types. Therefore, data given for a specified vegetation type are preferred. An indirect method of validation is to convert the fractional area burnt into fire return intervals (FRI), which have been observed for a broad range of ecosystems. The fire return interval, according to the fire history concept of Johnson and Gutsell (1994), is the expected return time of the fire. It is defined as the time required to burn an area equal in size to the study area. Their annual percent burn, here equivalent to average fractional area burnt, is the inverse of the fire return interval (averaged over a sufficiently long time period).

In using the FRI one must bear in mind its limitations and the difficulties associated with the implicit assumption of stable ecosystem conditions (see Clark 1989; Turner & Romme 1994). In reality, the observed fire regimes are a consequence of interannual variability in climate, of human influence and of changes in vegetation dynamics over different time scales.

Results

Validation of the fire model within the LPJ-DGVM

The first step is to validate the complete fire model Glob-FIRM within the LPJ-DGVM for sample regions in California, Portugal and Australia. The observations used to construct functions (4) and (9) are valid only on the short term and the fire-vegetation relationships are themselves dependent on not only the present but also the past climate and ecosystem conditions.

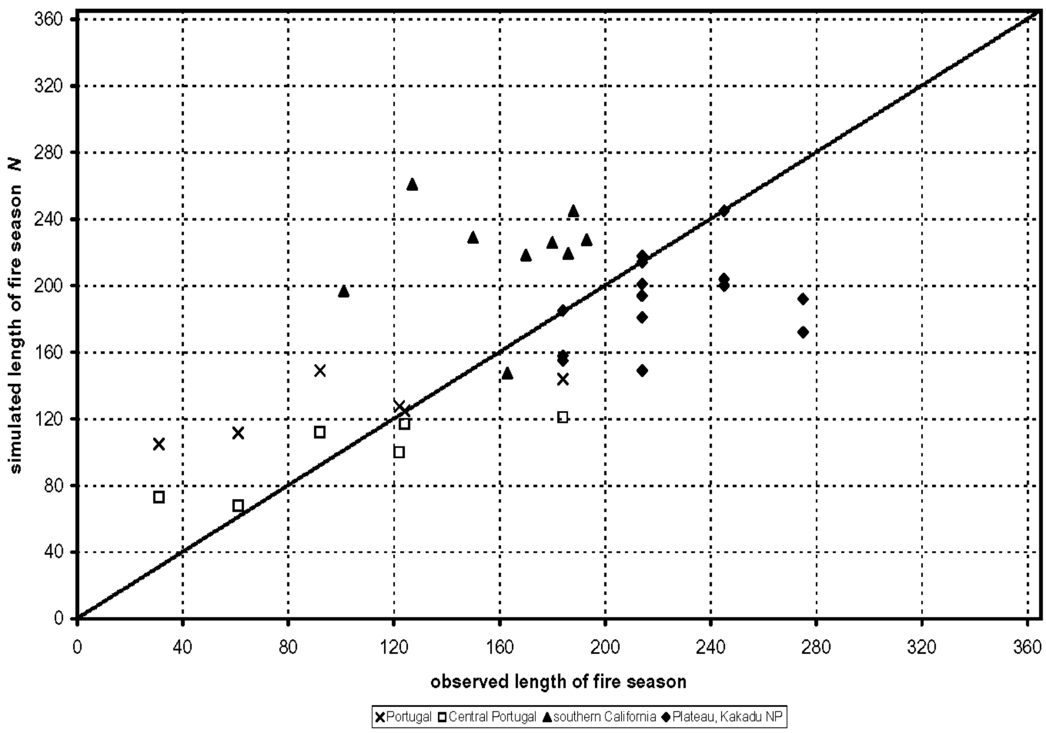

Fig. 1-3 Verification of the simulated length of fire season using LPJ-DGVM against observations in the sample regions.

The four sample regions differ in climate (interannual and seasonal variabilities), vegetation composition and flammability. For all four, the simulated length of the fire season is generally in good agreement with the observations (Fig. 1-3). The largest differences between model and observation are at sites with a length of fire season greater than 160 days, with model over- and under-estimates at the southern California and Kakadu National Park sites, respectively. Minnich (1998) describes a self-regulated fire regime for the Californian Chaparral where fire risk is reduced during the early-successional stage. The underestimated length of fire season for northern Australia in some years can be explained by the high flammability of the dominant *Eucalyptus* and *Spinifex* species, which promotes fire a feature not considered in the model.

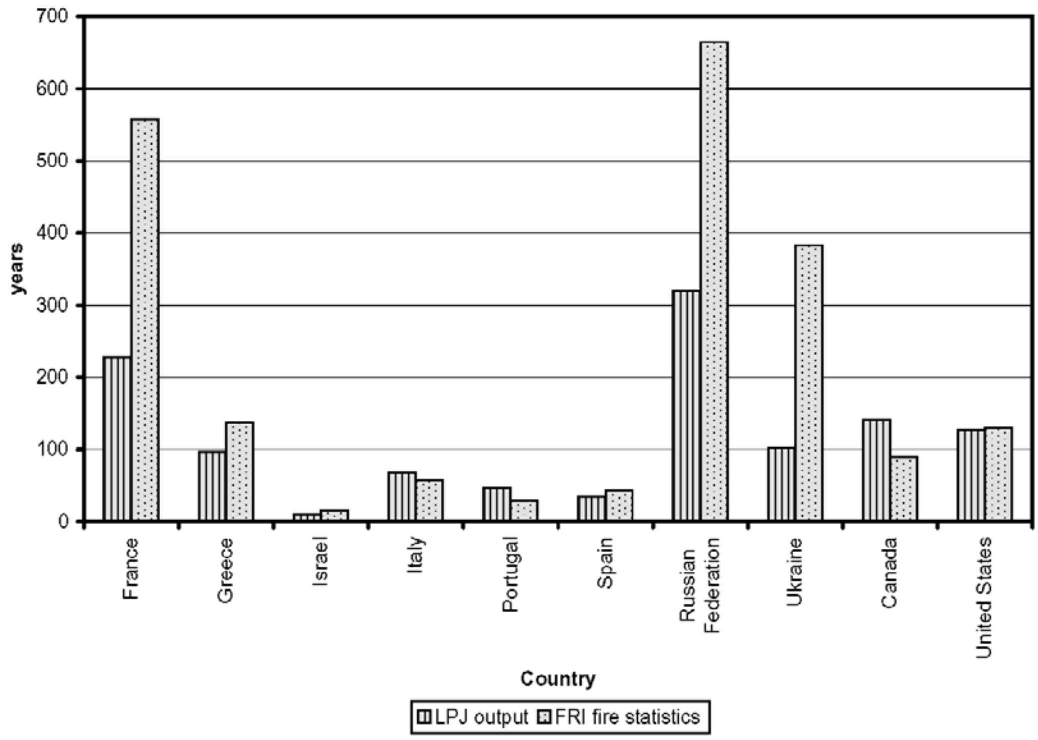

Fig. 1-4 Fire return intervals for the period 1987-1996 derived from the national fire statistics of forest services and simulated for the same period by the LPJ-DGVM (The data for the Russian Federation and Ukraine are given for the period 1991-1996).

Broad-scale observations of fire regimes around the world are still rare. The observation methods used over the last decades are remote sensing or on-ground fire inventories conducted by national forest services. Application of remote sensing to identify areas burnt and numbers of fires has only recently started and are still in development, therefore results vary markedly (1.5 - 2 times, e.g. Barbosa et. al., 1999). Hence, national fire statistics for forested and open woodland lands were used (FAO 1999) to check the credibility of our fire model. This data source contains total areas of fires in hectares for forests and other woodlands over the period 1987-1996 (for republics of the former Soviet Union over the period 1991-1996). The forest fire statistics for ten countries of Europe and North America, where the annual average of total area burnt over the entire period exceeds 10^3 ha (i.e. significant for the spatial resolution of the LPJ-DGVM) were taken for validation purposes. The average annual fractional area burnt over the period 1987-1996 (1991-1996 for Russia and Ukraine) for an entire forested land by country $A_{country}$ was derived using FAO data on national forest cover for 1995 (FAO, 1999). The average annual fractional area burnt for the ten countries was converted into fire return intervals (FRI) and then compared with the values produced by the LPJ-DGVM for the same time period (Grid cells with non-forested lands were masked out and $A_{country}$ was calculated by spatial averaging of the average annual fractional area burnt over the period 1987-1996 in the remaining grid cells and then converted to the FRI). The observed versus simulated FRI by country is shown in Fig. 1-4.

Global historical fire simulation

The simulated historical fire return intervals, averaged over the period 1901 to 1998, are in reasonable agreement for most regions with observations given in the literature and compiled in Fig. 1-5. Regions, which are unsuitable to carry fires (e.g. deserts) are shown in Fig. 1-6 as having intervals of more than 900 years. The transition between non-frequent to frequent fires is well captured by the model in Africa (Gillon 1992; Scholes & Walker 1993; Trollope 1984; van Wilgen *et al.* 1990), in South and Southeast Australia (Gill 1994; Walker 1981; Walter & Breckle 1991b), South America (Soares 1990) and in northern North America. In the following sections a few regions, covering a range of global ecosystems and ecotones, will be discussed in more detail.

North America

An almost complete picture of observed FRI is available for the North-American continent (Fig.1-5). The shape of the simulated FRI follows the climate gradients for boreal North America. Long intervals with almost no fire occur around the Hudson Bay, with a shortening, i.e. more fires, going westward in the boreal zone. Relatively short FRI are observed in central Alaska and neighbouring regions in the Canadian Northwest Territories, but intervals were much longer (implying almost no fire) in the coastal zone of the Pacific Northwest. In both of these regions the model is in reasonable agreement with observations.

It was more difficult to model the FRI for temperate grasslands (prairie) and for the boreal Labrador peninsula, where the modelled FRI were too long, whereas for some parts of the Appalachian Mountains fires should be less frequent. Due to sparse vegetation very long FRI would be expected in the North American deserts instead of intervals between 12 and 50 years (Fig. 1-6).

The transition between no fire occurrence and frequent fires in the tundra near the tree-line, e.g. near St. James Bay, Western Quebec, is modelled further south than reality in the boreal forest in Quebec, North West Ontario and Labrador (compare Fig. 1-5 and Fig. 1-6). Here, simulation of the fire season was less successful – the impact of other environmental factors on its length, e.g. the influence of permafrost and/or snow cover on the soil moisture regime needs to be investigated further.

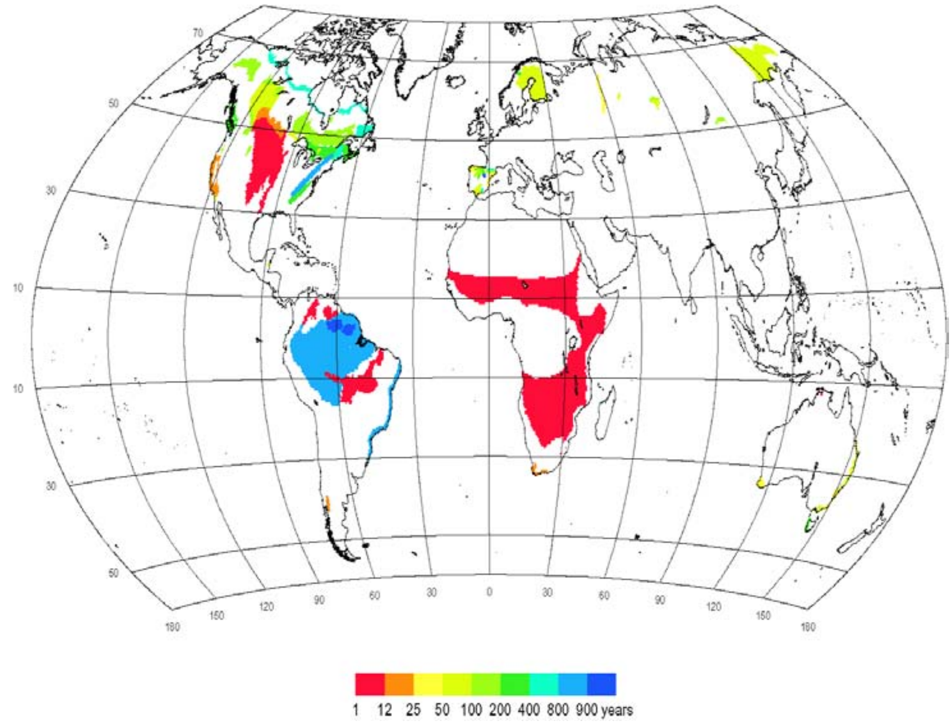

Fig. 1-5 Observed fire return intervals. Values are either given as an average for a vegetation type or estimated for a specific study area. According to the two categories of data sources, boundaries of the areas shown here correspond to the extension of the study area or were defined according to the defined extension of the major vegetation zone after Walter & Breckle (1983). (Bragg 1995; Briggs & Knapp 1995; Collins & Gibson 1990; Delisle & Hall 1987; Goldammer 1990; Goldammer & Furyaev 1996; Green 1986; Groves 1994; Johnson & Larsen 1991; Kellman & Meave 1997; Larsen & MacDonald 1998; Lavoie & Sirois 1998; Mooney *et al.* 1981; Niklasson & Granström 2000; Runkle 1985; Scholes & Walker 1993; Suffling *et al.* 1982; Swain 1973; Taylor & Skinner 1998; Turner & Romme 1994; Vázquez & Moreno 1998; Veblen *et al.* 1999; Whelan 1995)

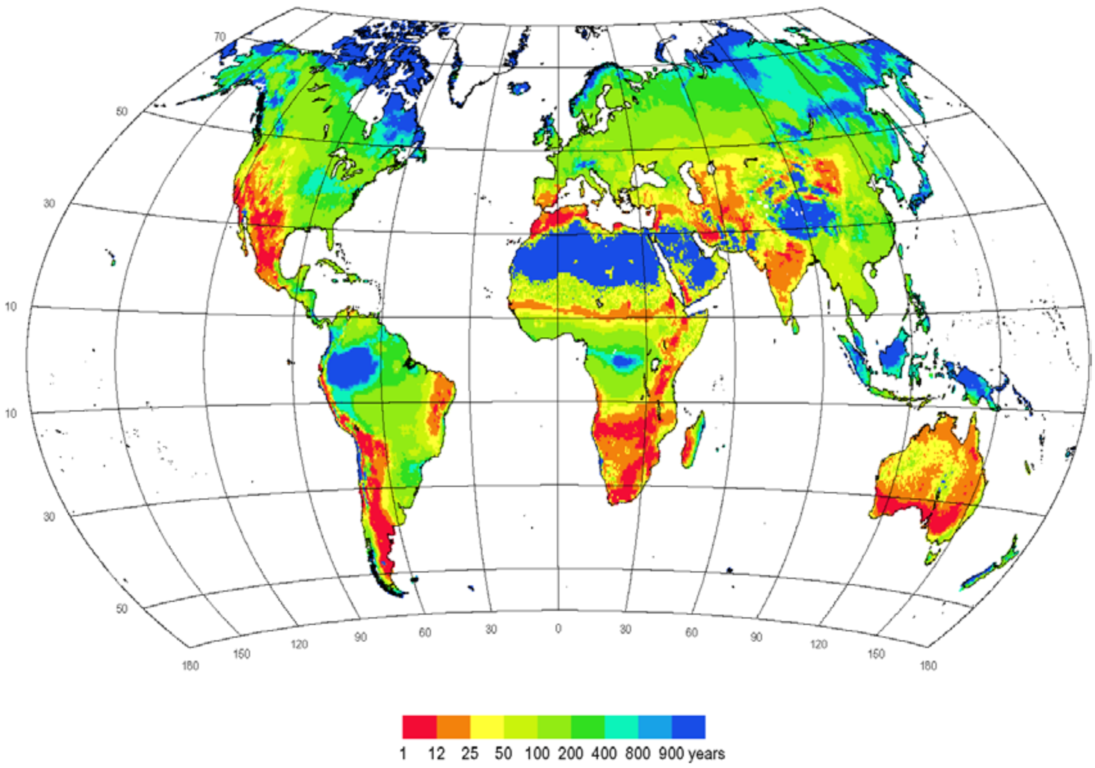

Fig.1-6 Historical fire intervals simulated by the LPJ-DGVM (averaged over the period 1901 to 1998).

The fire regime in the mixed-conifer northern hardwood forest in eastern North America is successfully modelled (e.g. Abrams & Orwig 1996; Grimm 1983; Swain 1973), although the longer FRI reported for the northern Appalachian Mts. are not captured. These intervals maybe a result of intense land use or fire suppression policy (Elliott *et al.* 1999).

North American tall-grass prairie and Aspen-parkland are burnt every 10 to 20 years, and in the other prairie types fires are even more frequent (Fig. 1-5). During the last centuries fire management practice of native Americans together with a favourable abiotic environment have created self-perpetuating fire conditions, leading to very frequent fires. This has led to the exclusion of primarily non-resistant woody plants in areas, where the climate would allow forests or at least woodlands (Bragg 1995; Grimm 1984). The complex interactions between drought, soil conditions, topography, grazing and fire (Collins & Wallace 1990; Joern 1995) make the correct simulation of the FRI very difficult. Addressing all of these factors, although important, is beyond the current scope of the analysis.

Central and South America

The impact of fire has been observed in many South American ecosystems, ranging from natural or anthropogenic savannas to burning in tropical rain forests (Kauffman & Uhl 1990; Soares 1990; Walter & Breckle 1991b). The spatial pattern of simulated FRI appears to follow the main gradients, with shorter FRI in the savanna regions of the Brazilian Cerrado, the Venezuelan Orinoco plains and the high plains in Guyana, and very long FRI for the tropical and temperate forests, including the coastal rain forest, as well as the Atacama desert (Fig. 1-5, Fig. 1-6). The reasons for the establishment of savannas are manifold and appear to be critical for a correct simulation of the fire regime, furthermore complicating the definition of fire drivers in these ecosystems (Coutinho 1990; Walter & Breckle 1991b).

Boreal Eurasia

The FRI simulated for Scandinavia are close to the observed 70 and 100 years (Bradshaw & Zackrisson 1990; Parviainen 1996). The length of the fire season for central and far eastern Siberia is incorrect, however. Observations by Korovin (1996) of a decreasing length of fire season with increasing latitude and a subsequent decrease in area burnt is not captured by the model (Fig. 1-6) in the region to the east of the Yenissey river and to the north of lake Baikal. The simulated FRI are close to those reported by Furyaev (1996) of between 20 and 70 years for southern Siberia and between 70 and approximately 150 years for western Siberia (for location of the study areas see Fig. 1-5). Ecological differences in

Siberia from those found in the western part of Eurasia, e.g. permafrost soils, extensive lichen coverage and complex topography, which are not explicitly modelled in LPJ-DGVM, are important for litter hydrology and need further investigation. In many cases LPJ-DGVM estimates high soil moisture contents, which lead to almost no days with burning conditions. In the model, the onset of the snowmelt fills the top soil layer, and the short growing season is insufficient to reduce the soil moisture content to the stage, where it drops below the moisture of extinction.

Whereas the simulated FRI for the Australian and South African ecosystems compares well with those given in van Wilgen et al. (1990) and Gill (1994) those simulated for Californian chaparral were slightly overestimated (six to 50 years compared with the observed 58 to 77 years, Minnich 1998). The geographic distribution of FRI for the Iberian peninsula generally agrees with the estimates given in Vázquez & Moreno (1998).

The moist microclimate in closed tropical rain forests permits burning only in extremely dry years (e.g., during El Niño events) and in canopy openings following tree cuts (Kauffman & Uhl 1990). Since the LPJ-DGVM does not consider gap dynamics explicitly, the FRI of all the tropical rain forests are between 400 and greater than 900 years, indicating almost no fire occurrence (see Fig. 1-6). However, more detailed and spatially explicit information about the natural fire regime in seasonal rainforest is needed to understand the gradient of increasing fire occurrence from the evergreen rainforest towards the savanna region.

Africa

On the African continent the FRI vary with latitude in a bimodal manner with peaks in the Sahara and the tropical rainforest. Moving southwards into the Sahel, the FRI decrease with increasing fuel load, and are progressively lengthened further south with increasing precipitation, resulting in no fire activity in the tropical forest. Probably due to human intervention, more frequent fires than simulated are observed in the African savannas, especially in the Sahelian and Guinean savanna. In southern Africa the small coastal area of the Namib desert which has no fire is well captured in the simulation (Fig. 1-6), whereas the area of longer simulated FRI would be expected to be larger in surrounding areas of the Karoo and Kalahari ecosystems in central southern Africa where the vegetation is presumably too sparse to carry a fire (Schultz 1988; Werger 1986).

In water-limited ecosystems such as the African savannas, a sequence of years with lower annual precipitation, which kills a big part of the vegetation, leads to extreme fire events which reduce carbon pools and vegetation coverage (compare Fig. 1-7 ad, especially after simulation year 1945). Litter carbon is fluctuating close to defined limit for supporting fire spread (see section ”fire occurrence”), therefore fire does not occur in all years, when annual precipitation declined. Re-vegetation takes several years depending on climate condition and plant competition. Tropical broad-leaved rain-green woody trees strongly compete with C_4 herbs for the remaining resources. The dominance of these two PFTs, which shed their leaves during the dry season, lead to an increase in the litter pool, and thus large fractional area burnt due to the higher annual precipitation in the years after simulation year 1955 (see Fig. 1-7 b-d).

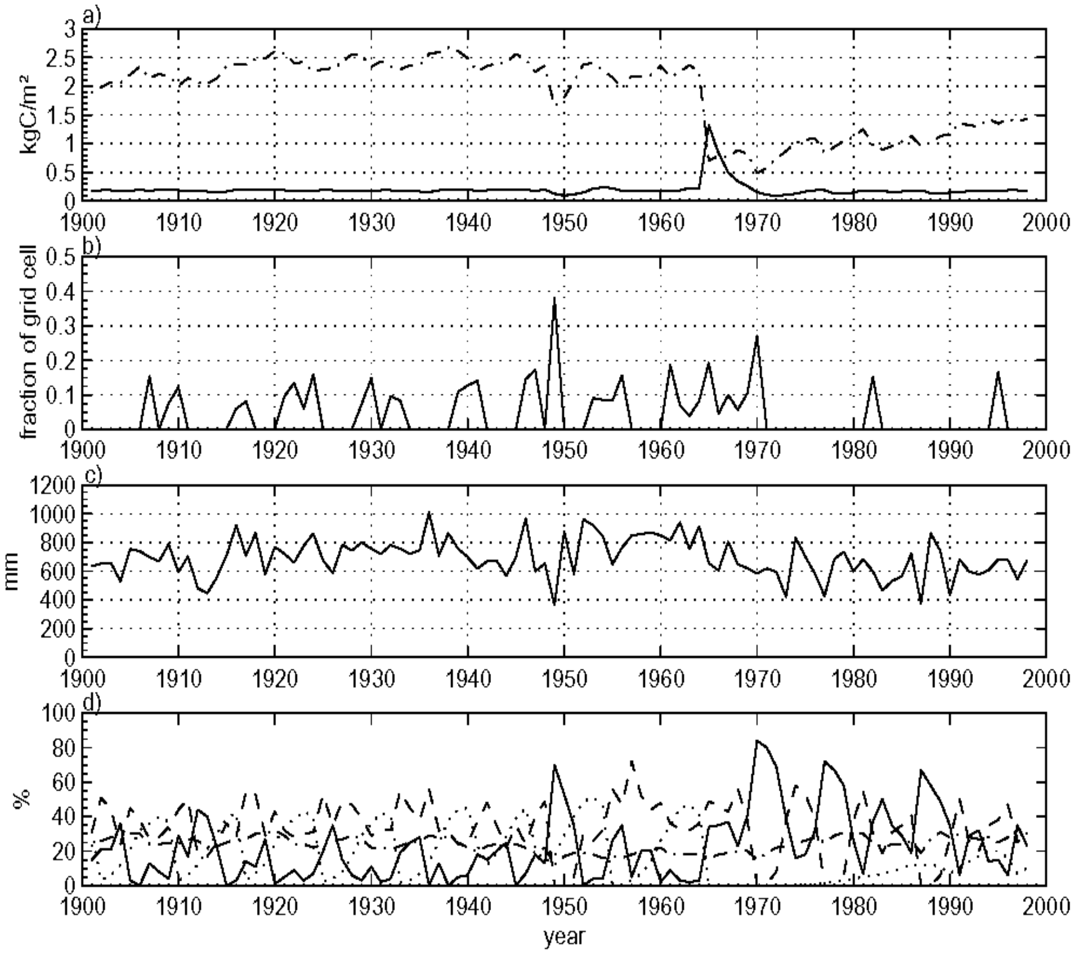

Fig. 1-7 Temporal dynamics of a) vegetation carbon (dash-dotted line) and litter carbon (solid line), b) fractional area burnt, c) annual precipitation and d) fractional PFT coverage (dashed: C_4 herbaceous, dotted: tropical broadleaved raingreen woody, dash-dotted: tropical broadleaved evergreen woody and solid: barren) for a grid cell in the northern African savanna (10.75° E, 11.75° N).

Discussion

The fire model, which is a compromise between fire history and process oriented models, successfully simulates fire regimes in many of the world’s ecosystems. In many regions it is apparently sufficient to consider only fuel load and litter moisture as the key drivers of the fire regime. In these cases, extrapolating experimental data from regional studies in order to construct globally applicable fire functions and the assumption that the accumulated length of fire season gives the annual fractional area burnt are valid.

In some regions the simulated results differed from the observations indicating that processes or environmental conditions (i.e. permafrost, lichen cover and edaphic conditions) not described in the LPJ-DGVM are important. Additional physical properties of fuel seem to be necessary for inclusion into the model in order to obtain better results especially for grassland regions. In other regions, human activities both directly and indirectly influence the fire regime through fire management and modification of land cover respectively. It was therefore not possible to fully validate the fire model against natural fire regimes in these regions. This question should be addressed in later studies, e.g. inclusion of human population as an additional fire driver. The present model functions are based on a limited amount of observed data, since data should cover a sufficient number of years in order to capture the climate variability at any location. Despite intensive literature surveys, more data showing both area burnt and length of fire season in other ecosystems is needed. Therefore, as new data becomes available the function can change.

Our fire module Glob-FIRM, coupled into a Dynamic Global Vegetation Model, provides a bi-directional feedback between fire and vegetation. PFT specific fire effects and the ability of PFTs to establish influence post-fire vegetation dynamics in the opened sites. The new assemblage of PFTs has an impact on fire occurrence through competition, plant growth and thus water demand of each PFT. An increase in fire can enhance the expansion of fire-tolerant PFTs, i.e. those with higher survivorship after fire. Their vegetation dynamics will then feed back on the fire regime. The effects of climate change on plant water demand can indirectly buffer or intensify fire occurrence and effects depending on vegetation productivity. These patterns and their consequences for vegetation composition and the global carbon cycle should be addressed in a detailed study. Glob-FIRM implemented in the LPJ-Dynamic Global Vegetation Model is a suitable tool for this, since feedbacks between vegetation composition its productivity and fire through moisture conditions and litter accumulation are considered in one combined model. Changes in either vegetation pattern or fire can be investigated with respect to their effects on other modelled processes. Under climate change conditions, it can show new aspects of the atmosphere-biosphere interaction, which could be different from existing studies, where projections of fire danger indices independent of the influence of vegetation dynamics on fire drivers were studied (Flannigan *et al.* 1998).

The modified LPJ-DGVM allows a further insight into the global role of fire for biosphere dynamics. To provide a full scope of successional dynamics, however, other disturbance agents such as insects or wind-throw, must be included in DGVMs, thus completing the picture of biotic and abiotic interactions in ecosystems.

Acknowledgements

We thank I. Colin Prentice for suggestions concerning the basic structure of the fire module, Johann G. Goldammer and Mike Fosberg and participants at several fire workshops for many comments concerning fire behaviour at various scales, and Yuri M. Svirezhev for a comment on the design of the function $A(s)$.

2. SIMULATING FIRE REGIMES IN HUMAN-DOMINATED ECOSYSTEMS: IBERIAN PENINSULA CASE STUDY

Sergey Venevsky, Kirsten Thonicke, Stephen Sitch and Wolfgang Cramer

Abstract

A new fire model is proposed which estimates areas burnt on a macro-scale (10 to 100 km). It consists of three parts: evaluation of fire danger due to climatic conditions, estimation of the number of fires, extent of the area burnt. The model can operate on three time steps, daily, monthly and yearly, and interacts with a Dynamic Global Vegetation Model (DGVM), thereby providing an important forcing for natural competition. Fire danger is related to number of dry days and amplitude of daily temperature during these days. The number of fires during fire days varies with human population density. Areas burnt are calculated based on average wind speed, available fuel and fire duration. The model has been incorporated into the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM) and has been tested for peninsular Spain. LPJ-DGVM was modified to allow bi-directional feedback between fire disturbance and vegetation dynamics. The number of fires and areas burnt were simulated for the period 1974-1994 and compared against observations. The model produced realistic results, which are well correlated, both spatially and temporally, with the fire statistics. Therefore, a relatively simple mechanistic fire model can be used to reproduce fire regime patterns in human-dominated ecosystems over a large region and a long time period.

Keywords: Fire model, number of fires, human-caused fires, areas burnt, coarse-scale modelling, Spain

Introduction

Fires have a large impact on vegetation and ecosystem structure. By killing vegetation, fires open closed canopies and facilitate seed germination for certain species, thereby influencing both the seed pools and vegetative propagules in the soil. Fire is important in the life cycle of many plant species and for their competitive abilities, thereby contributing to the structure of plant communities. Associated with fire is a release of carbon into the atmosphere. On the global scale, the total biomass burnt is about 4.5 Pg C yr^{-1}, of which more than 2.5 Pg C yr^{-1} are released in savannah and forest ecosystems (Levine 1996). This value is comparable to other major fluxes in the global carbon budget (5.5 Pg C yr^{-1} for industrial emissions, about 2 Pg C yr^{-1} ocean uptake, 1.5 Pg C yr^{-1} land use) (IPCC 1995). Therefore, fire is an integral part not only in vegetation succession, but also in global biogeochemical cycles.

A variety of models (see review in Gardner *et al.* 1999) have been developed to describe fire effects in natural ecosystems. Statistical fire models (e.g. Johnson & Gutsell 1994), using Weibull probability distributions as a function of stand age, imply a static fire regime. Application of such concepts to new environmental conditions (i.e. another study region or changed climatic conditions at the site) would require new parameterisations. An analysis of the ecosystem component responsible for changes in a particular fire regime, cannot be identified using this modelling approach. To study various facets of the vegetation-fire interaction an explicit simulation of fire dynamics (fire ignition and spread conditions) and its effects on vegetation regeneration (fire intensity, area burnt and subsequent successional dynamics) is required. Such a mechanistic modelling approach would allow studies under changing environmental conditions. Physical models (Albini 1976; Rothermel 1972), later adapted to Chaparral ecosystems by Davis and Burrows (1994), simulate fire dynamics explicitly and can be applied at small scales (hectares to square kilometres). To model fire dynamics at scales of coarse-scale vegetation dynamics (i.e. at a spatial resolution of 100 to 10000 km^2 and at daily to annual time steps, but applied over an investigation period of several decades), a generalisation of this concept as well as its parameterisations are required.

Different generalisations have been made regarding scaling and in modelling fire dynamics and its effects on vegetation dynamics. These studies differ in their interpretation of process relevance at coarser scales. Lenihan et al. (1998) uses a combination of physical models to describe fire occurrence and spread in the model MCFIRE. Keane (1996b) uses Weibull probability distributions in FIRE-BGC for fire ignitions. Fire behaviour and spread is then explicitly modelled using physical models (FARSITE Finney 1994). This approach fixes the fire weather conditions causing fire ignition to the historical pattern, and therefore ignores any changes in the fire weather system. Consideration of human and natural ignition sources, such as in CRBSUM (Keane *et al.* 1996a) in which probability functions are used, can be used as a tool for fire management and land use strategies. Thus, a fire weather index, interpreting the fire danger in certain climatic conditions, combined with a rather mechanistic formulation of human-caused ignitions would be the preferable tool to model fire occurrence. This would allow a more realistic representation, since in some ecosystems 85 to 97% of the fires are human-caused (Moreno *et al.* 1998; Shvidenko *et al.* 1998). The need for improved coarse-scale fire models, which can also be used for practical fire management has been identified by Keane & Long (1998). The successional pathway approach to model post-fire succession at the coarse-scale in CRBSUM, does not consider new successional dynamics, which could develop in a changing environment. This can be done using a vegetation model, which simulates vegetation processes dynamically.

The main problems simulating coarse-scale fires at the regional scale were described by McKenzie *et al.* (1996) as follows:

- A lack of data on the ecological effects of fire at coarse spatial resolution complicates model development and validation.
- Process-based fire models were built for fine scales (from hectares to square kilometres) and thereby assume ecosystem homogeneity, which is no longer the case at regional scales.
- Landscape patterns influence both ignition and fire spread, however these patterns are poorly (if at all) described on a coarse scale.

The importance of specific fire-relevant processes depends on the spatial and temporal scale, which needs to be recognised when their functional representation in fire models is applied at a different scale. Therefore, generalisations of the model concept, re-parameterisation and validation of the modelled processes become necessary.

In order to improve our understanding of fire regimes and their effects on vegetation dynamics, and considering the new requirements, we have developed the model Reg-FIRM (Regional FIRe Model). This model simulates general fire dynamics based on physical models, and explicitly considers different ignition sources. It is designed for fire simulation at coarse spatial scales (100 to 2500 km^2) and can be adapted for calculation on daily, monthly or yearly time steps.

The Reg-FIRM is designed as a further development of our previous fire model Glob-FIRM (Thonicke *et al.* 2001), which combines the fire history concept (Johnson & Gutsell 1994) with process-oriented fire modelling. Glob-FIRM simulates fire regimes at the global scale and coarse-scale fire-vegetation interactions.

We incorporated Reg-FIRM into the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM Sitch *et al.* 2002) to allow dynamic interactions between fire and vegetation. Reg-FIRM provides area burnt for the DGVM by estimating the number of fires, which depends on climatic fire danger, both human and natural-caused ignitions, and the average size of a fire in a grid cell.

In this study the regional fire model Reg-FIRM was run inside the LPJ-DGVM at a daily and monthly time step for the period 1974 to 1994 at a spatial resolution of 0.5° x 0.5° longitude / latitude across the Iberian peninsula, as an example of a fire-prone ecosystem with dominance of human-caused fires. Using this tool we want to study the role of the different fire drivers on the spatial and temporal pattern of fire. Here, we show that a simple non-statistical formulation can integrate the various reasons for human-caused fires. This approach, a generalisation of physical models to describe fire spread is sufficient to reproduce the observed pattern at the regional scale.

Methods

Fire effects are a product of fire ignitions, number of fires, fuel characteristics and the persistence of positive burning conditions over space and time, resulting in fire spread during a certain fire duration, and certain area burnt. Post-fire conditions then determine vegetation regeneration, which feeds back on the fire-driving conditions through new fuel development and species flammability. By separately calculating number of fires and area burnt the model allows a direct comparison to the available fire statistics. The following equation is the basis of the regional-scale fire model:

$$PB(t) = \frac{\sum_{N_{fire}(t)} S_{N_{fire}}\left(d_N\right)}{S_{grid}}, \tag{1}$$

where *PB(t)* is the fraction of a grid cell burnt up to time *t* (between zero and one), $N_{fire}(t)$ is the number of fires in the period (0, *t*), $S_{N_{fire}}$ is the area burnt by the fire number *N*, d_N is the duration of this fire and S_{grid} is the area of a grid cell. In the following section, the design of the model components will be described in detail.

This model concept implies that within one time step fires burn with the same characteristics and have an equal impact on the final area burnt. Additionally, the applicability of the equation is limited by the lowest value of S_{grid}, which should not be less than 100 km^2. Otherwise, the complete grid cell would burn and fire spread to neighbouring cells would have to be incorporated into the model. Therefore, the equation does not describe extremely large convective fires, but considers rather average fire conditions. These relatively rare events may be described using special statistical and physical methods reviewed elsewhere (Moritz 1997; Valendik *et al.* 1978).

Equation (1) can be applied in three modes, with a daily, monthly or yearly time step. Implementing the equation with a daily time step assumes stochastic ignition and duration of fire and application of fine-scale physical models of fire spread (Albini 1976; Rothermel 1972; Van Wagner 1969). By integrating fire behaviour and fire effects over a simulation year one obtains the annual area burnt, which corresponds to the fire regime of a grid cell. The fire regime for a grid cell can also be assessed annually, although less accurately, using mean annual values of number of fires and mean area burnt if we assume that all the fires during the year have an equal impact. This approach is less computationally expensive (for instance, by excluding daily weather simulation) and also enables one to obtain the response of the fire regime to climate driving forces on a monthly and yearly time step.

Implementation

We assume that at the regional scale fire disturbance in an ecosystem can be described by the following processes: occurrence of fire danger, associated with dry weather; fire ignition, possibly affected by human activity; fire spread, forced by available fuel and wind; fire termination due to weather or suppression activities; and fire-induced vegetation mortality (see e.g. Whelan 1995). With a total fuel load of less than 200gC m^{-2} no ignition and fire spread is possible (see Thonicke *et al.* 2001).

Fire danger

Since most fires are initiated in the litter layer, a series of models are required to simulate wetting and drying processes in the ground fuel layer during changing weather conditions. Several fire rating systems have been proposed which estimate the ignition potential of fuel, mainly as variations of the Canadian Forest Fire Weather Index (Van Wagner 1987), the American National Fire-Danger Rating System (Deeming *et al.* 1974) or the Nesterov Index which is widely used in Russia (Nesterov 1949).

We use the Nesterov Index (NI) for the fire danger rating, mainly because it is a relatively simple equation and does not require variables such as daily wind speed or daily humidity, for which accurate data are practically unobtainable over large regions. A comparison of Canadian and Russian fire danger systems with the satellite measurements in 1992 for Central and Eastern Siberia shows that both rating systems successfully tracked the increasingly extreme fire danger condition (Stocks *et al.* 1996). The Nesterov Index, however, is the only suitable one for estimation of ignition potential in comparison with the other two rather complex fire rating systems, because it drops to zero rapidly after a small amount of rain.

The Nesterov Index was derived as an empirical function reflecting the relationship between fire and weather based on historical data (Nesterov 1949). It is calculated using daily temperature (at 15h), dew-point temperature, and precipitation. The difference between the two temperatures is multiplied by the daily temperature and summed over the number of days since the first day, in which precipitation dropped below 3 mm:

$$NI(N_d) = \sum_{ifP(d)\le 3mm}^{N_d} T_{daily}(d) * (T_{daily}(d) - T_{dew}(d)), \tag{2}$$

where $NI(N_d)$ is the Nesterov Index (°C^2) for day N_d , and d is a positive temperature day with a precipitation of less than 3 mm. When the daily precipitation exceeds 3 mm, the Nesterov index falls to zero. *NI* values between 300 and 1000 are considered to be moderate for ignition potential. *NI* values ranging between 1000 and 4000 represent high ignition potential, while values above 4000 represents extreme potential for ignition.

The following equation is a discrete approximation of the NI:

$$NI(N_d) = \sum_{ifS\le m_e}^{N_d} \frac{(T_{\max}(d) + T_{\min}(d))}{2} * \left[\frac{(T_{\max}(d) + T_{\min}(d))}{2} - (T_{\min}(d) - 4)\right], \tag{3}$$

where the summation is made over the dry days with a positive minimum daily temperature and when the relative soil moisture $S(d)$ in the upper soil layer, expressed in relative volumetric units, is less than the moisture of extinction m_e, above which fire can not spread (Albini 1976). In the model, a value of 0.3 for woody and 0.2 for herbaceous vegetation types is taken for m_e (Thonicke *et al.* 2001). $T_{min}(d)$ and $T_{max}(d)$ are daily minimal and maximal temperatures, while ($T_{min}(d)$-4.0) represents a simple approximation of $T_{dew}(d)$, the daily dew point temperature (Running *et al.* 1987).

Expressing equation (3) as a continuous function we obtain on a daily time step:

$$NI(N_d) = \int_0^{N_d} f(S(t)) * \left[\frac{(T_{\max}(t)^2 - T_{\min}(t)^2)}{4} + 2 * (T_{\max}(t) + T_{\min}(t)) dt\right], \tag{4}$$

where $f(S(t)) = \exp\left(-\pi * \left(\frac{S(t)}{m_e}\right)^2\right)$ is a simple probability function for a dry day. Estimates of NI at monthly and yearly simulation time steps are based on monthly or seasonal averages of $T_{min}(d)$ and $T_{max}(d)$:

$$NI_m = \left(\frac{\left({T_{\max}^{m}}^2 - {T_{\min}^{m}}^2 \right)}{4} + 2 * \left(T_{\max}^{m} + T_{\min}^{m} \right) \right) * \int_0^{N_d^m} f(S(t))dt, \quad (5)$$

and

$$NI_s = \left(\frac{\left({T_{\max}^{s}}^2 - {T_{\min}^{s}}^2 \right)}{4} + 2 * \left(T_{\max}^{s} + T_{\min}^{s} \right) \right) * \int_0^{N_d^s} f(S(t))dt, \quad (6)$$

where NI_m represents the accumulated Nesterov Index for month m (with positive minimum daily temperatures), assuming minimal and maximal temperatures $T_{\max}^{m}$ and $T_{\min}^{m}$ are constant during the month and equal to their mean values. N_d^m is the total number of days in month m. The integral over $f(S(t))dt$ for the time N_d^m describes the season of critical fire conditions in the fuel bed and is named hereafter as the length of the fire season. NI_s is the accumulated Nesterov Index for an entire season with positive minimal daily temperatures, and $T_{\max}^{s}$, $T_{\min}^{s}$ are averaged over this period. N_d^s is number of days with positive minimum daily temperatures in a year.

The fire danger index *FDI*, based on the Nesterov Index, should interpret qualitative risk terms such as low, moderate, high and extreme ignition potential in quantitative terms. A smooth risk probability function for fire danger with a daily time step is

$$FDI(N_d) = 1 - \exp(-\alpha * NI(N_d)), \quad (7)$$

while for average monthly and annual values of *FDI* we receive

$$FDI(m) = \frac{1}{NI_m} \int_0^{NI_m} (1 - \exp(-\alpha * NI)) d(NI) = 1 + \frac{1}{\alpha * NI_m} * (\exp(-\alpha * NI_m) - 1), \quad (8)$$

and

$$FDI(s) = 1 + \frac{1}{\alpha * NI_s} * (\exp(-\alpha * NI_s) - 1) \quad (9),$$

respectively.

The tuning parameter α (°C^{-2}) can be used to provide different values of probability for low, moderate, high and extreme ignition risks, based on *NI* values. It is set at 0.000337, resulting in upper limits for daily FDI of approximately 0.1 for low, 0.3

for moderate, 0.75 for high and 1 for extreme ignition potentials according to (9). It can be redefined when detailed observations are available.

Number of fires

Nowadays, human-caused fires dominate over fires induced by lightning in many natural ecosystems, e.g., lightning-caused fires accounted for only 3% of the fires that occurred in Spain between 1974 and 1994 (Moreno *et al.* 1998). Even the more remote regions have a larger number of human-induced fires than lightning-caused. For example, lightning caused only 13.7% of the fires in the boreal zone of Russia between 1986 and 1995 (Shvidenko *et al.* 1998).

A simple estimate of the number of lightning-caused fires l per area and day of fire season can be derived from the total number of lightning-caused fires during twenty years in Spain, (5663 Moreno *et al.* 1998) within a total area of 505 10^3 km^2, assuming a season length with fire danger equal to 200 days (Korovin 1996; Melekhov 1978). The derived value of l, equal to 0.028 of lightning fires per day per million hectares is similar to a value of 0.015 for the boreal zone of Russia (Telitsyn 1988). We took a value of l=0.02 as a constant rate for this version of Reg-FIRM.

Ignition sources related to human activity are rather different. The majority are related either with recreation or labour activities of humans, the remaining related to intentional clearing of vegetation (Goldammer & Jenkins 1990). Generally, human-caused fires are concentrated near human settlements or transportation routes, i.e. are related to the spatial distribution of human population and the accessibility of natural ecosystems (see Pausas & Vallejo 1999, for the case of Iberian peninsula). On the other hand, human ignition potential depends on economic status and life style. With growth of urban population, people tend to spend more labour and recreation time within cities. Therefore, human-caused ignitions are hypothesised to be dependent on the population density to which various life styles are assigned as a simple approximation.

The following simple composite function describes the total daily number of possible ignition causes, human and natural, for a grid cell:

$$N_{ig}(N_d) = (k(P_D) * P_D * a(N_d) + l) * S_{grid}, \tag{10}$$

where N_{ig} is the total number of ignitions for a day N_d in a grid cell, P_D is the population density per km^2 in a grid cell, S_{grid} is the area of a grid cell in million ha. $k(P_D)$ is a scaling function representing different ignition potentials of humans in densely populated and scarcely populated regions due to differences in spatial patterns of human settlements and the exposure of humans to natural ecosystems. We took the known function of fire danger, in terms of population density representing exposure of humans to natural ecosystems, following Telitsyn (1988), and expressed in relative units:

$$k(P_D) = 6.8 * P_D^{-0.57}, \tag{11}$$

The function implies that an average person in scarcely populated regions like Australia, Canada or Russian Asia potentially produces three to four times more ignitions (because of longer activities within natural ecosystems) than, for example, an average person in densely populated regions of Europe.

Human ignition potentials

The stochastic variable $a(N_d)$ is the number of ignitions produced by one human during day N_d multiplied by a factor 10^4 (in order to scale the population density per million ha). We assume that this stochastic variable is distributed exponentially (i.e. a probability density function for $a(N_d)$ is $\lambda*\exp\langle-\lambda a\rangle$). The mathematical expectation ($1/\lambda$) of this distribution depends on life style and wealth status of humans. For example, a value $1/\lambda=0.1$ is obtained using the annual average number of fires during the period 1980-1989 for northern circumpolar countries together with their human population (Stocks 1991). The mathematical expectation has the physical meaning that almost one out of ten humans produce one ignition in a natural ecosystem during the person's active labour and/or leisure lifetime (30 years). Such a rather heuristic hypothesis results, for example, in 0.25 ignitions in a day per one million hectares for regions with a population density of 2 persons per km^2, which is in agreement with the value observed for northern regions of the Russian Far East with the same population density (Shvidenko *et al.* 1998). For peninsular Spain the value $1/\lambda=0.22$ is obtained from the number of human-caused fires (pasture burning, caused by negligence or intentional) during the period 1974-94 (Moreno *et al.* 1998, Table 6) and the average human population density over the region. The uniformly distributed random number p_d used to calculate the value of $a(N_d)$ is given by:

$$a(N_d) = -\frac{\ln p_d}{\lambda}, \tag{12}$$

where $\lambda=1/0.22$. Fig.2-1 illustrates the number of ignitions following equation (10). Therefore, the annual average number of fires $N_{fire}(s)$ for a grid cell can be estimated as:

$$N_{fire}(s) = FDI(s) \int_{when T_{\min} \geq 0} f(S(t))dt * (\gamma * P_D{}^{\Theta} + l) * S_{grid} \quad , \qquad (13)$$

where γ is a constant equal to 1.496 and Θ is equal to 0.43 for peninsular Spain (see (10), (11) and (12)). The number of fires in a month can be calculated in the same way as the annual calculation (see (11)-(13)).

Equation (13) has almost the same mathematical formulation as the equation used by the Russian forestry practice to estimate the annual number of fires per million hectares over large administrative units:

$$N_{fire}(s) = C(s) * N_d(s) * (\delta * P_D{}^{0.5} + l), \qquad (14)$$

where *C(s)* is a fire danger coefficient, based on the Nesterov Index, N_d(s) is the number of days with moderate, high and extreme fire danger, estimated using the Nesterov Index, and δ is a coefficient of population fire activity for a region (Melekhov 1978). However, estimates of *C(s)*, N_d(s) and δ can be rather complex in this method. Therefore, equation (13) with the underlying approximations is used in the model concept.

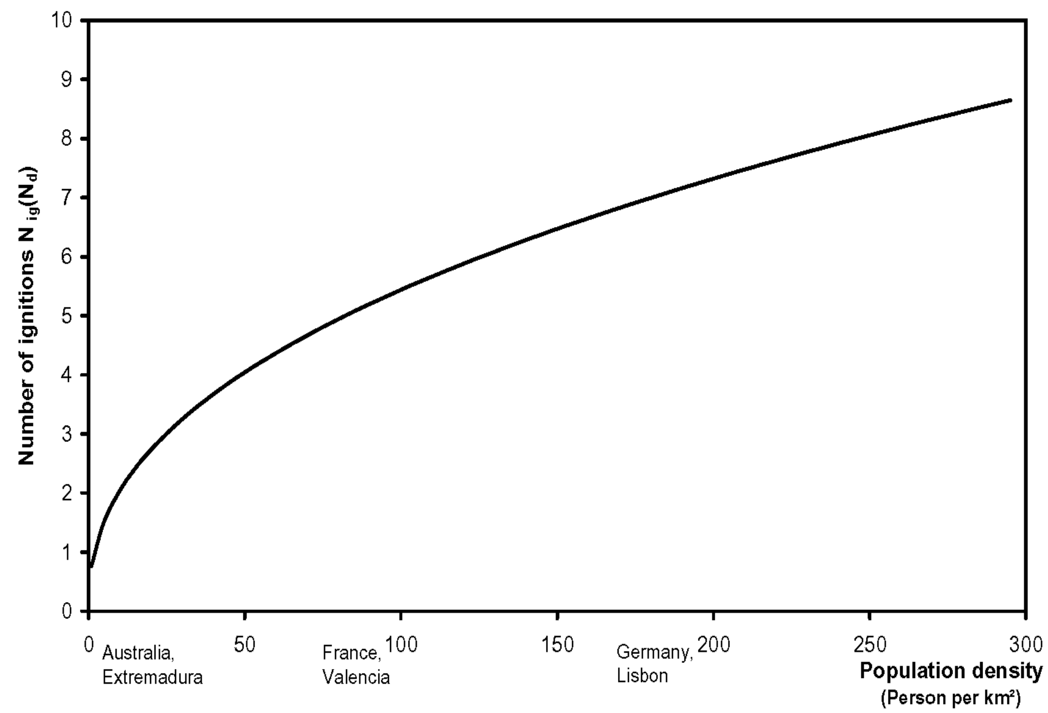

Fig.2-1 Number of possible ignitions $N_{ig}(N_d)$ related to human population density.

Humans tend to have different motivations to ignite fires, reflecting regional differences in their socioeconomic background. These need to be taken into account, when estimating human ignition potentials for a given region. Examples of regional variation are presented below. Ranges for the constants Θ and γ (and

$a(N_d)$) can be estimated for a region, given the other variables in equations (1) and (13) are known *apriori* or can be assumed, and a distribution of area burnt by population density classes is identified. The values of constants will be estimated for Africa to prove the validity of the approach for all regions. This is done using the mean percentage of area burned in each population class relative to the total extension of the class itself for the period 1985-1991 (see Table 10 in Barbosa *et al.* 1999).

Indeed, the mean annual percentage of area burned PB_{mean} by population density can be approximated as (see (1),(13)):

$$PB_{mean} = FDI_{mean} \int_{whenT_{mean} \geq 0} \widetilde{f}(S(t)dt * (\gamma * P_D{}^{\Theta} + l) * \widetilde{S}_{fire} \quad , \qquad (15)$$

where FDI_{mean} is mean value of the fire danger index, $\int_{whenT_{mean} \geq 0} \widetilde{f}(S(t)dt$ is the average length of fire season, $\widetilde{S}_{fire}$ is the average fire size in million hectare. For Africa FDI_{mean} is set to 1, the average length of fire season $\int_{whenT_{mean} \geq 0} \widetilde{f}(S(t)dt$ to 182 days (half a year, seeBarbosa *et al.* 1999), and lightning fires (l=0) are neglected. From these variables, the average fire size for Africa is the most uncertain, since long-term observations are missing. For example, for Kenya the average fire size varies from 347 ha in 1990 to 40 ha in 1999 (Ndambiri & Kahuki 2001). Therefore, average fire size $\widetilde{S}_{fire}$ is set to $150*10^{-6}$ million ha, which is the average value for shrublands according to Keane and Long (1998).

The values for Θ and γ (and $a(N_d)$) are calculated through logarithmic regression of PB_{mean} by P_D. PB_{mean} was obtained from satellite observations, where two methods, Sc1 and Sc2, were used to detect area burnt (Barbosa *et al.* 1999). The first method (Sc1) is believed to overestimate the total area burned, while the second (Sc2) is probably near to the minimum value for burned area estimations. The values of Θ and γ (and $a(N_d)$), obtained for the Sc2 method, are close to those used for peninsular Spain: $\Theta = 0.4$, γ=1.93 and $a(N_d)$ =0.28. For the Sc1 method these values are equal to $\Theta = 0.4$, γ=3.43 and $a(N_d)$ =0.51. Therefore, the exponent Θ, describing the spatial distribution of human settlements in relation to fires, does not vary much by geographical region (see also equation (14)). On the other hand, the number of ignitions produced by one human during the day, $a(N_d)$, can differ several times, depending on life style (rural versus urban, traditional versus modernized) and land use practices among regions.

Fire spread

Active fires increase in size depending on their rate of spread, as the flame front moves through the fuel bed and favourable burning conditions continue over time (defined as the duration of a fire). The rate of spread is driven by wind and fuel characteristics. The more energy the flame front can build up, the faster the fire can spread. The combined influence of these factors results in a specific geometric shape, the fire perimeter, from which the area burnt can be deduced. The impact of topography, another important driving factor, is not considered at this stage.

Here, the fire spread model on a daily time step is based on the hypothesis that we can not distinguish between different burning conditions of each fire in a grid cell and the assumption, that all fires, started in one day, have the same (stochastic) duration and (constant) rate of spread:

$$PF(N_d) = \pi * U(N_d) * d(N_d). \quad (16)$$

$PF(N_d)$ is the perimeter of the fire, $U(N_d)$ is the frontal fire rate of spread, $d(N_d)$ is a stochastic duration of fire initiated in a day N_d. This method generalises fire-spread functions for single fire events to the application at the regional scale. Within one day the probability that within one grid cell (note the cell size chosen at the regional scale) burning conditions are notably different between each fire, is very low. We used an exponential distribution for the duration of fire (the probability density function being μ*exp(-μd)), with a mathematical expectation approximately equal to one day (1.0). This distribution function provides a good fit to the histogram for duration of fire incidents, based on 50-year fire statistics for Russia (Korovin 1996). Values for $d(N_d)$ are obtained from equation (12) with λ=μ=1/1.0

Rate of spread $U(N_d)$ is estimated using the Rothermel (1972) approach to fire spread modelling, but in a rather simplified form, as proposed by Telitsyn (1988; 1996):

$$U(N_d) = U_0(N_d) * \left(1 + A * W(N_d)^B\right), \quad (17)$$

where $U_0(N_d)$ is the rate of spread for the backing fire (i.e. without wind) in m s^{-1}, $W(N_d)$ is the wind speed in m s^{-1}, A and B are constants (Rothermel 1972). We fixed both the constants A and B to 1.0, according to Telitsyn's (1996) approximation, and do not simulate wind speed, assuming an average value of 3.7 m s^{-1} according to Vázquez and Moreno (1998). The constant value of wind speed was taken because of the general absence of a coarse-scale wind model with daily resolution.

The rate of spread for a backing fire was approximated by Telitsyn (1988; 1996) from heat balance equations for inward burning within the fuel bed:

$$\sigma * E_i * \left(T_f^4 - T_0^4\right) = \rho(N_d) * \left[c(t_i - t_0) + L * \omega(N_d)\right] * U_0(N_d), \quad (18)$$

where σ is the Stefan-Boltzman constant 5.7*10^{-8} J m^{-2} K^{-4} s^{-1}, E_i is the inward emissivity of the "flame-fuel" system. T_f and T_0 are the absolute temperatures of the flames and fuel surface in K, respectively. $\rho(N_d)$ is the bulk density of the fuel bed in kg m^{-3}, c is specific heat of fuel in J kg^{-1} C^{-1} and t_i is the ignition temperature of fuel in °C. t_0 is the temperature of fuel surface in °C, L is the latent heat of evaporation 2.6 10^{6} J kg^{-1} and $\omega(N_d)$ the fuel moisture content expressed as a mass fraction.

Assuming that some of the parameters are constant, as given in published data (Davis *et al.* 1959; Demidov & Saushev 1975; Rothermel 1972), we obtain the following approximation for $U_0(N_d)$:

$$U_0(N_d) = \frac{3 * E_i}{\rho(N_d) * (16 + \omega(N_d) * 100)}, \quad (19)$$

when T_f = 1200 K, T_0 =293 K, c = 1400 J kg^{-1} C^{-1}, t_i = 300°C, t_0 = 20 °C (Telitsyn 1996).

We take the soil moisture *S(t)* in the upper 50 cm of the soil as a surrogate for the fuel moisture content and $\rho(N_d)$ as the fuel bulk density, weighted by local vegetation types:

$$U_0(N_d) = \frac{3 * E_i}{\sum_j w(j)\rho_j(N_d) * (16 + S(N_d) * 100)}, \quad (20)$$

where $w(j)$ is the fractional ratio of a given vegetation type *j* at the location. Since fuel bulk densities differ between vegetation types *j*, fuel flammability will vary according to vegetation composition.

The inward emissivity of the "flame-fuel" system E_i varies for low-intensity fires between 0.1 and 0.3, and between 0.3 and 0.7 for fires of moderate intensity (Demidov & Saushev 1975; Telitsyn 1996; Thomas 1965).

The perimeter shape of a wind-driven fire can often be approximated by an ellipse (Van Wagner 1969). According to this approximation the ellipse will have an elongated semi-major axis in the wind direction and the smaller axis represents the progress of the backing fire (Albini 1976). Complex exponential regression equations for parameters of elliptical models for areas burnt by wind speed were elaborated at the Northern Forest Fire Laboratory, Missoula, USA, (see Albini 1976). An application of these equations within a range of wind speeds between 0 and 4 m s^{-1} results in a ratio of semi-major to semi-minor axes for an elongated ellipse of between 0.45 and 0.6; a value of 0.5 for a wind speed of 3.7 m s^{-1}. This two-to-one relation between the length and width of area burnt is often observed for fires in the boreal zone (Melekhov 1978). It allows us to make simple estimates of area burnt assuming that it is approximately one half the area of a circle with a diameter equal to the elongated axis of the downwind driven ellipse:

$$S_{fire} = 0.5 * \frac{PF^2(N_d)}{4*\pi} , \qquad (21)$$

A simplified version of the last equation ($S_{fire} = 0.04 * PF^2$) is applied for practical purposes by the Russian Forest Service to obtain a rough estimate of fire spread (Far Eastern Forest Institute 1987).

The average annual fraction burnt is relatively straightforward to derive, if one assumes that fuel physical parameters are constant (i.e. average rate of spread $\overline{U}$ is constant) during a year (see (1), (13), (16) and (21)):

$$PB(s) = \frac{N_{fire}(s) * \pi * \overline{U}^2}{8 * S_{grid} * \mu^2} , \qquad (22)$$

The monthly fraction burnt can be calculated in the same way.

The Lund-Potsdam-Jena Dynamic Global Vegetation Model

Reg-FIRM was incorporated into the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM Sitch *et al.* 2002) and tested over peninsular Spain for the period 1974-1994. The vegetation processes relevant to fire, in terms of driving fire and fire effects, will be described in the following section.

The LPJ-DGVM was constructed in a modular framework. Individual modules describe key ecosystem processes, including vegetation establishment, resource competition, growth and mortality. Vegetation structure and composition is described by nine plant functional types (PFTs), which are distinguished according to their plant physiological (C_3, C_4 photosynthesis), phenological (deciduous, evergreen) and physiognomic (tree, grass) attributes. Other land cover types, including agriculture and pastures, and the historical impacts of land-use on vegetation dynamics are not considered here. The model is run on a grid cell basis with input of soil texture, monthly fields of temperature, precipitation and percentage sunshine hours. Each grid cell is divided into fractions covered by the PFTs and bare ground. The presence and fractional coverage of an individual PFT depends on its specific environmental limits, and on the outcome of resource competition with the other PFTs. The annual values of foliar projective cover of PFTs in a grid cell were used as the fractional ratios of different vegetation types $w(j)$ in order to estimate average rate of fire spread in the fire model (see equation (20)).

The two-layer soil water balance model is based on (Haxeltine & Prentice 1996b). Moisture in each layer, expressed as a fraction of water holding capacity $S(N_d)$, is updated daily. Percolation from the upper to the lower layer, and absolute water holding capacity are soil texture dependent. The value of $S(N_d)$, used in the fire model (see equations 13, 19), is also affected by the water use efficiency of the vegetation in the LPJ-DGVM. Extensive water use by a PFT can increase the fire risk by lowering soil moisture content.

Establishment and mortality are modelled on an annual basis. Plant establishment, in terms of additional PFT individuals, depends on the fraction of bare ground available for seedlings to successfully establish. Natural mortality is taken as a function of PFT vigour, and corresponds to an annual reduction in the number of PFT individuals. Dead biomass enters one litter pool and two soil pools. Mortality also occurs due to fire since the areas occupied in a grid cell by each PFT decreases proportionally to the total area burnt. The number of individuals surviving a fire is defined for each PFT (expressed in terms of a fraction, see Thonicke *et al.* 2001). Vegetation types are therefore differentially affected by fire. The bare ground opened by fire is used for establishment of seedlings, providing a feedback between fire and vegetation in the LPJ-DGVM.

Data

Parameters of the fire model

The number of ignitions produced by one human $a(N_d)$ was set to 0.22 and fire duration $d(N_d)$ to 1.0. The inward emissivity of the ”flame-fuel” system E_i was fixed at 0.3, assuming that fires had moderate intensity in peninsular Spain during the simulation period. Fuel bulk densities of the four PFTs occurring in the Iberian Peninsula as defined by the LPJ-DGVM, are listed in Table 2-1.

Plant Functional Types	**Fuel bulk density ($kg\ m^{-3}$)**
Temperate needle-leaved evergreen tree (e.g. Abies alba, Pinus sylvestris, P. nigra, P. halepensis)	16
Temperate broadleaved evergreen tree (e.g. Quercus ilex, Qu. suber, Qu. coccifera)	10
Temperate broadleaved summergreen tree (e.g. Fagus silvatica, Quercus robur,Castania spp., Betula spp.)	10
C_3 perennial grass (graminoids)	2
C_4 perennial grass	2

Table 2-1. Fuel bulk density by Plant Functional Type (PFT) used in LPJ-DGVM. The values were obtained from the table of fuel parameters for the Mediterranean region used by the Spanish Environmental Ministry (Ministerio de Medio Ambiente, España) for fire management (Merida 1999). The thirteen fuel types in this table were combined into the five appropriate PFTs of the model. PFT specific fuel bulk density was calculated dividing the average fuel load by the average fuel depth.

Model input

All input data sets were provided at a 0.5° x 0.5° longitude / latitude spatial resolution. The monthly climate data (precipitation and temperature) were provided by the CCMLP project (Carbon Cycle Model Linkage Project) over the historical period 1860-1995 and derived from the data of Hulme (1995) and Jones (1994). Historical CO_2 concentrations were derived from ice core and atmospheric measurements (Enting *et al.* 1994). Soil texture information was obtained from the FAO soil data set (FAO 1991). Population density data from 1990 onwards were extracted from the global population density at a 5' resolution and upscaled to a 0.5° grid (FAO 1991; Zobler 1986). The population growth rate for the period 1974-1994 was obtained from the Spanish National Institute of Statistics (Instituto Nacional de Estadística (1999).

The model was validated by comparing simulation output with published data of number of fires and area burnt in peninsular Spain during the period 1974-1994 (Moreno *et al.* 1998). These data include the geographical distributions of number of fires and areas burnt for the entire period and their yearly course.

Results

Geographic distribution

The simulated geographic distribution of number of fires and area burnt over peninsular Spain during the period 1974-1994 generally reproduce the observations (compare Fig. 2-2 and 2-3 with Fig.4 in Moreno *et al.* 1998). A large area of peninsular Spain has between 2 and 20 fires per 10000 ha and an area burnt between 10-1000 ha for the two decades, which agree well with the fire statistics for the entire period (see Fig. 2-2).

The spatial distribution of number of fires fits observation better than the distribution of areas burnt, reflecting considerable differences in climate conditions and population concentration in different regions of the country. The largest number of fires over the twenty year period (50 to 100 fires per 10000 ha) were both simulated and observed in the coastal zone of southern Spain in provinces of Andalusia, Valencia and Catalonia, as well as near big cities like Madrid and San Sebastian. The lowest number of fires (two to nine fires per 10000 ha for 1974-1994) were both simulated and observed for the Iberian mountain system, the Zaragoza province and the Pyrenees. The model underestimates the number of fires in the Galician province and in the Cantabrian Mountains (2 to 50 fires per 10000 ha in 1974-1994 against 50 to 100 observed fires). This underestimation is most likely related to the high number of intentional ignitions in the region initiated mainly in the early nineties (see Fig. 5 and Fig.12 in Moreno *et al.* 1998).

Both observed and simulated geographic distributions of areas burnt for 1974-1994 are generally more smooth than for number of fires, e.g. in mountainous regions of Catalonia, Valencia and Andalusia (see Fig. 2-3). In the model vegetation pattern influences fuel composition in these regions damping a rapid spread of fires. Again the largest areas burnt both observed and simulated (100 to 10000 ha in 1974-1994) occur in the coastal Mediterranean zone, while areas burnt are considerably less in the North-Eastern part of Spain (10-500 ha in 1974-1994). The model does not reproduce large areas burnt in Galicia and the Cantabrian Mountains because the number of fires in these regions is underestimated.

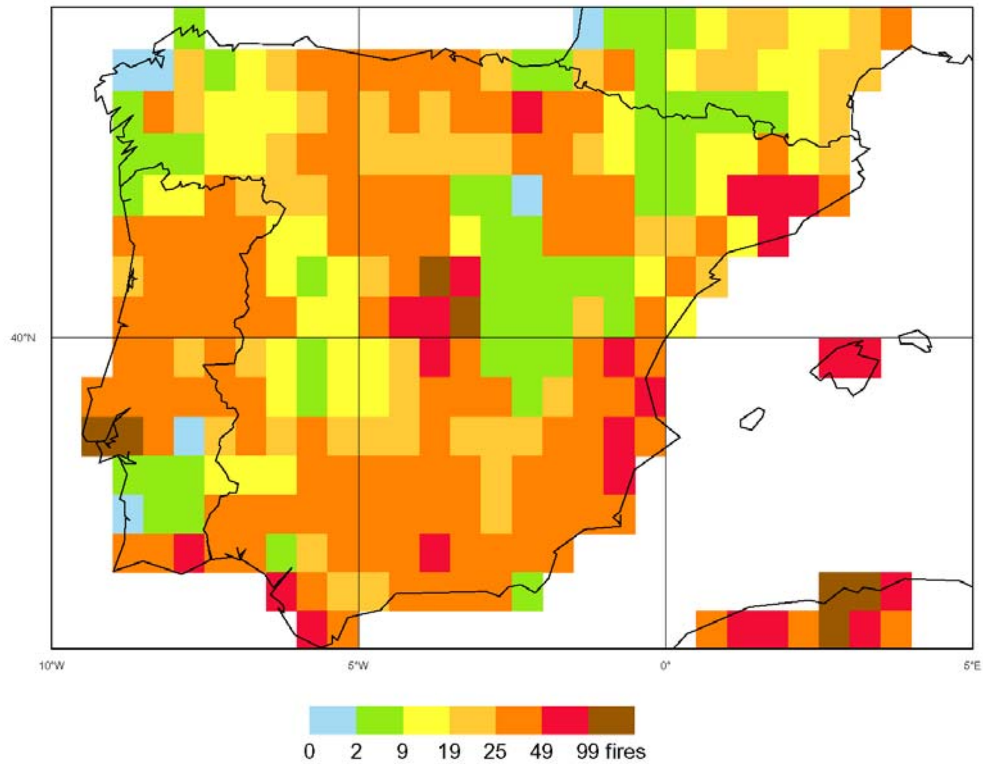

Fig.2-2 Simulated number of fires per 100 km² on the Iberian peninsula during 1974-1994.

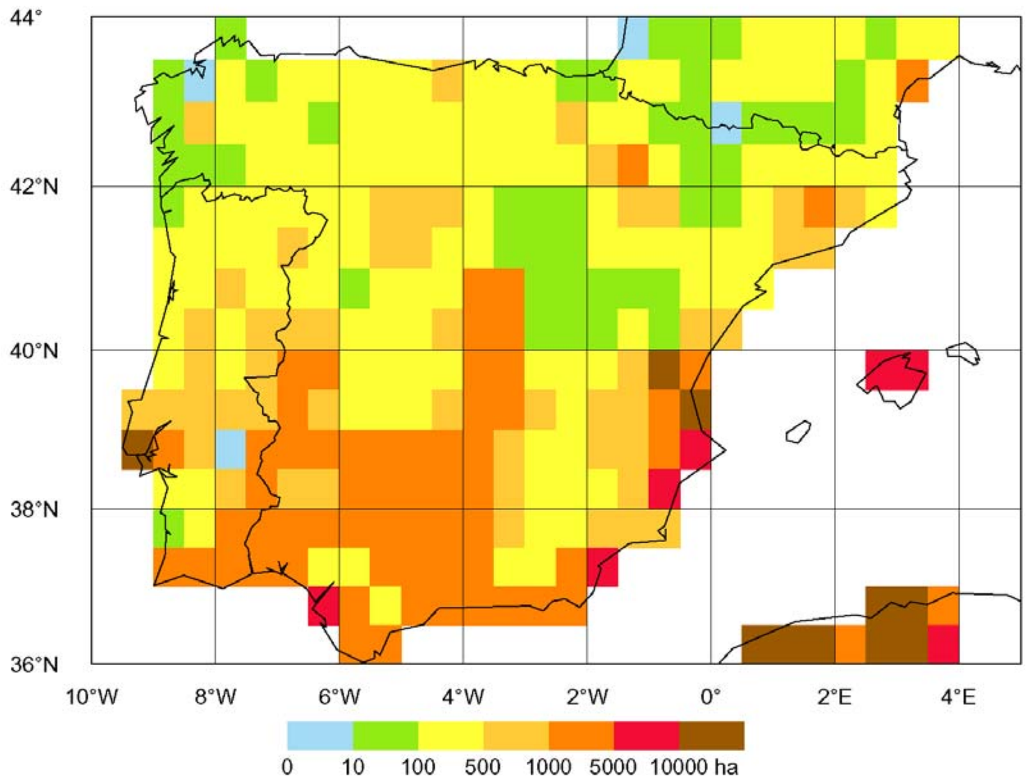

Fig.2-3 Simulated areas burnt (ha) on the Iberian peninsula during 1974-1994.

Annual dynamics of fires in Spain for 1974-1994

The observed total number of fires and areas burnt over peninsular Spain between 1974 and 1994 were extracted from Table 1 in Moreno et al. (Moreno *et al.* 1998) and compared against the simulated totals (see Fig. 2-4 to 2-5).

It can be seen that observed and simulated number of fires and areas burnt have similar annual dynamics. The annual mean number of fires and areas burnt are well reproduced. The simulated number of fires has the same maximum-minimum sequence until 1988, except for 1983 (see Fig. 2-4). In 1983 the number of fires and areas burnt are simulated to increase (which is opposite to fire statistics data) as a consequence of a considerable drop in monthly precipitation seen in the input climate data. The simulated and observed areas burnt generally have the same temporal pattern, except in 1983 and 1989-1990 (see Fig. 2-5). In some years, the simulated number of fires exactly matches observation, while the corresponding annual area burnt is lower (compare year 1979 and 1980 in Fig.2-4 and 2-5). Here, simulated conditions for fire spread differ from those observed. Discrepancies in vegetation composition, indirectly influencing fire spread through specific fuel characteristics, and in simulated moisture conditions might have reduced fire spread and thus area burnt. A good correlation in temporal patterns between observed number of fires and areas burnt is obtained for peninsular Spain before 1989 (compare with Fig.2 in Moreno *et al.* 1998). The discrepancy thereafter may be an indication of possible problems in the fire history records for these five years. Possible socio-economic changes in land use practices in this time accompanied with improvements in fire management and fire detection in Spain may be one of the explanations for the dramatic increase in number of fires while the increase in area burnt is lower (compare Fig.2-4 and 2-5). However, the formulation of human-caused ignitions in Reg-FIRM may not be sufficient to capture changes in fire causes as well as fire management after 1989.

The observed and simulated annual trend and amplitude of total areas burnt in Spain coincides well (see Fig. 2-5), while the simulated number of fires has both a smaller trend and amplitude than observed (Fig. 2-4). There are a few possible explanations for such model behaviour, e.g. soil moisture dynamics may not be simulated correctly by LPJ-DGVM, or the monthly averaging of fire danger index can produce a too rough approximation, compared with the daily fire danger index. However, the most likely reason is absence of dynamic land use changes in LPJ-DGVM for the simulated period. The socio-economic changes, which have taken place in the last decades in the southern EU countries, e.g. rural exodus and abandonment of agricultural lands, reduced grazing pressure, urbanisation of some rural areas, are mentioned by many authors (Moreno *et al.* 1998; Rego 1992; Vélez

1997) as some of the main reasons of notable changes in the number of fires and areas burnt in these countries.

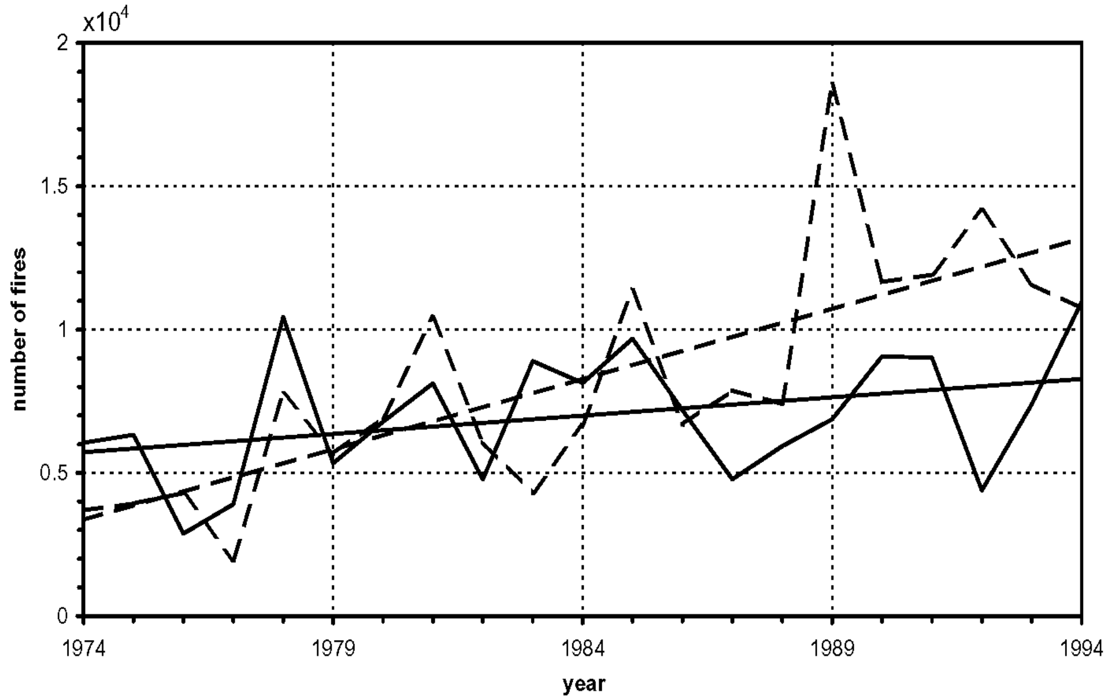

Fig.2-4 Time series of the total number of fires in peninsular Spain during 1974-1994. The dashed and solid lines represent the observed and simulated number of fires, respectively.

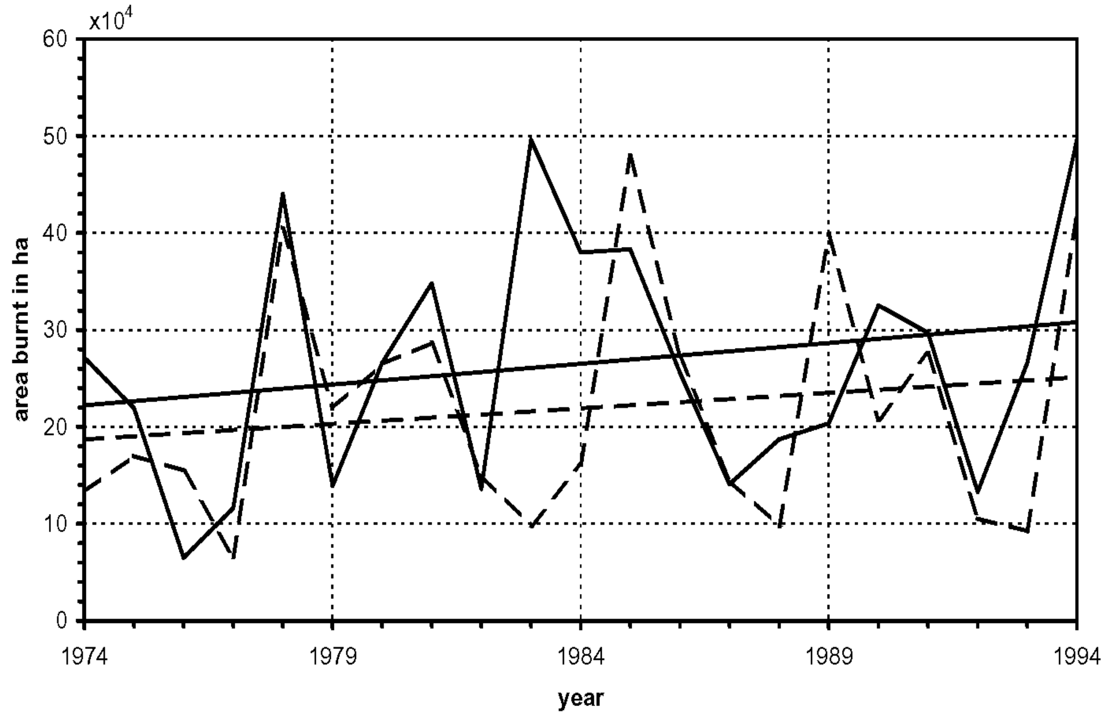

Fig.2-5 Time series of the total area burnt (ha) in peninsular Spain during 1974-1994. The dashed and solid lines represent the observed and simulated area burnt, respectively.

Dynamics of fires by size for the period 1974-1994

The annual area distributions for peninsular Spain in terms of numbers of fires and the surface burned were calculated from Table 3 in Moreno et al. (1998) and compared with the simulated distributions for the period 1974-1994. This table contains the number of 10 x 10 km grid cells for each case subdivided into five classes. For the comparison exercise we reclassified the data for both cases into two classes and summed areas, where no fires were observed. There were a total of 220 grid cells at 0.5° resolution for peninsular Spain, and therefore 5500 grid cells at a 10 km resolution (assuming each large cell contains 25 smaller ones). The two classes, from 0 to 3 fires per 100 km^2 (S - area with few ignitions) and more than 3 fires per 100 km^2 (F – area with frequent ignitions) were defined for the numbers of fires comparison. Similarly for the surface burned we defined two classes: grid cells having the surface burned less than 100 ha per 100 km^2 (A is the area with fires of average or small size) and more than 100 ha per 100 km^2 (L is the area with large fires). The observed distributions in 1974-1994 for the four classes at the 10 km resolution (in fractions) are presented in Table 2-2.

For each year the 220 grid cells at 0.5° resolution were allocated into the four classes, according to the simulated numbers of fires and the simulated surface burned. The total number of cells in the two larger classes (S – grid cells with less than 3 ignitions per 100 km^2 and A – grid cells, having less than 100 ha burned per 100 km^2) was compared against observation, obtained by multiplying 220 by the ratios from Table 2-2. The two larger classes (S and A) were taken for comparison with observations in order to investigate the model's ability to capture major features of fire patterns in Spain. Besides, the two smaller classes (F-area with frequent ignitions and L – area with large fires) have a complementary dynamics.

The observed and simulated total numbers of grid cells in the two classes have similar temporal dynamics, although the magnitude differs slightly. The calculated mean number of grid cells for the class S in the period 1974-1994 is 207 compared with 191 observed, while the calculated mean number of grid cells for the class A in the same period is 192 compared with 206 observed. Such discrepancy follows from the comparison method of two samples with significantly different sizes (5500 and 220) due to loss of information during sample aggregation. Indeed, the inner heterogeneity of the grid cells at 10 km resolution within the larger 0.5° cells can considerably affect the final distributions in the four classes. Spatial dependency in statistical geo-referenced data analysis (so-called *modifiable area unit problem*) is well recognised, but the problem has still not been resolved (Wong 1996). In our case, to avoid the systematic bias of the method we subtracted the means of the two classes from observed and simulated time series and compared the dynamics of anomalies (see Fig. 2-6 to 2-7)

year	0 to 3 fires per 100 km^2 (S)	> 3 fires per 100 km^2 (F)	surface burned < 100 ha per 100 km^2 (A)	surface burned > 100 ha per 100 km^2 (L)
1974	0.925	0.075	0.948	0.052
1975	0.937	0.063	0.952	0.048
1976	0.920	0.080	0.946	0.054
1977	0.959	0.041	0.976	0.024
1978	0.865	0.135	0.896	0.104
1979	0.909	0.091	0.935	0.065
1980	0.862	0.138	0.907	0.093
1981	0.838	0.162	0.902	0.098
1982	0.884	0.116	0.940	0.060
1983	0.907	0.093	0.968	0.032
1984	0.883	0.117	0.939	0.061
1985	0.815	0.185	0.866	0.134
1986	0.872	0.128	0.922	0.078
1987	0.873	0.127	0.942	0.058
1988	0.866	0.134	0.952	0.048
1989	0.779	0.221	0.881	0.119
1990	0.813	0.187	0.927	0.073
1991	0.815	0.185	0.928	0.072
1992	0.831	0.169	0.958	0.042
1993	0.852	0.148	0.975	0.025
1994	0.841	0.159	0.956	0.044

Table 2-2. Distributions of 10 km resolution grid cells for four classes in peninsular Spain during the period 1974-1994 (fractions to the total).

The dynamics of anomalies for areas with few ignitions and for areas with fires of small or average size are simulated rather realistically. We have inversions for simulated dynamics of anomalies for the number of grid cells in the class S in only four years (1976, 1983, 1989, 1993) and a large difference in the magnitudes in 1994. The additional inversion in 1988 appears for simulated dynamics of anomalies for the number of grid cells in the class A, thus the application of the fire spread model does not result in considerable, additional error. The temporal patterns of areas with few ignitions and for areas with fires of small or average size are well reproduced in all other years. Therefore the model is able to reproduce major features of annual

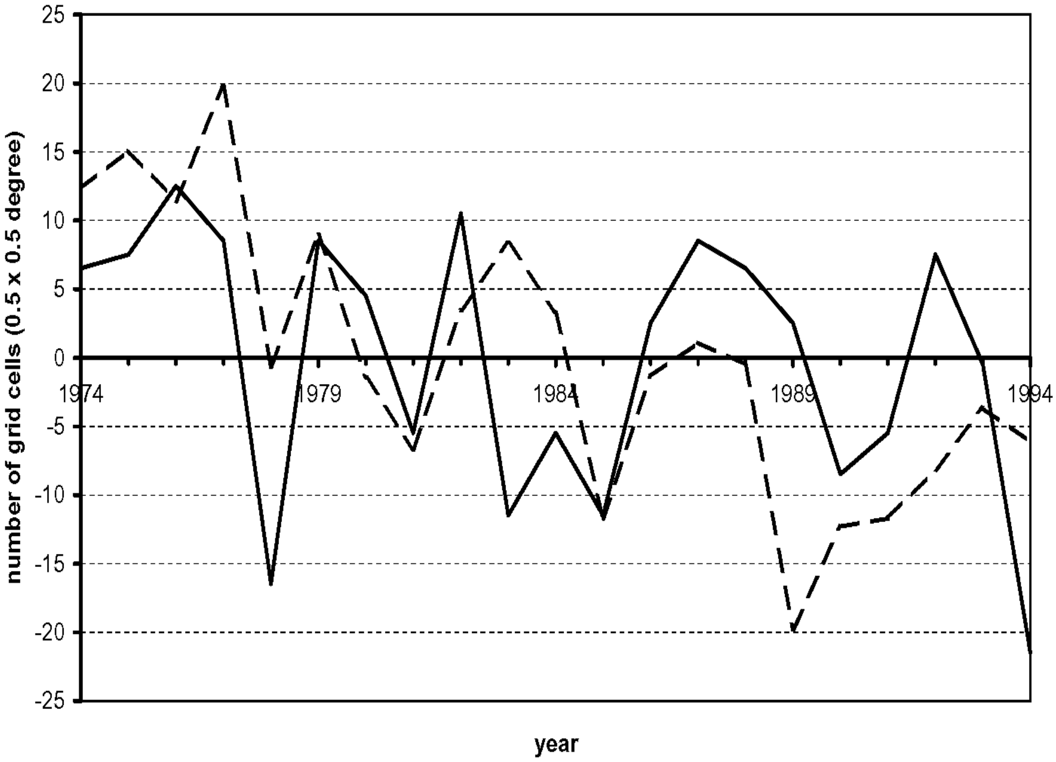

Fig.2-6 Area anomalies for fires less than 3 per 100 km^2 in 1974-1994 (units are number of grid cells 0.5 x 0.5 degree). The dashed and solid lines represent the observed and simulated area anomalies, respectively.

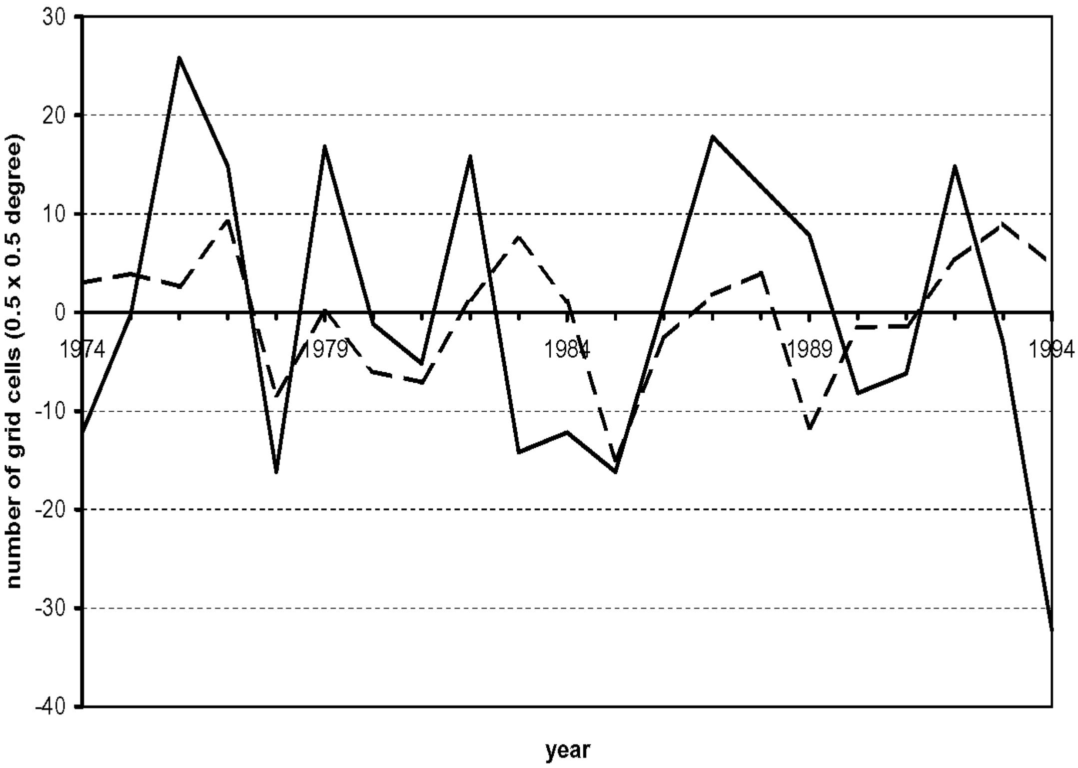

Fig.2-7 Area anomalies for less than 100 ha burned per 100 km^2 in 1974-1994 (units are number of grid cells 0.5 x 0.5 degree). The dashed and solid lines represent the observed and simulated area anomalies, respectively.

dynamics for geographical patterns of ignitions and fire regimes, although more accurate spatial statistical analysis is needed if observed geo-referenced data is available. However, the simulated time series have larger amplitude than those observed (especially in the case for anomalies of areas with fires of small or average size). Most likely only a simulation on a daily (not on monthly) time step can produce a closer fit for these time series, because the fire danger index and the spread of fire are sensitive to precipitation and the wind speed.

Discussion

This study can be considered a successful application of a relatively simple mechanistic fire model in reproducing both spatial and temporal dynamics of fire over a large region and a long time period. By making simple assumptions about climate and human-induced changes, the fire model allows investigation of fire regimes on different time scales over large geographical regions. Processes selected to describe fire at the regional scale are at the right level of explicitness as the agreement in the temporal patterns demonstrates. Simulation results at a monthly resolution show that it is possible to study fire-vegetation feedbacks using a mechanistic approach instead of stochastic modelling of fires that require more computer resources and parameter values.

The relatively good agreement between observed and simulated number of fires demonstrates the importance of both the climatic fire risk and the ignition sources, on fire occurrence (see Fig. 2-4). The reproduced pattern shows the impact of inter-annual climate variability on fire, since in this simulation study we assumed a constant number of ignitions per human, expressed in a constant $a(N_d)$ (see equation (10)). By keeping both wind speed and fire duration constant one can isolate the individual impact of vegetation (see equations (16), (17) and (20)). Since a good agreement between simulated and observed area burnt was obtained this implies that vegetation has a dominant influence on fire. Fires cannot spread if the fuel bed is too moist or the amount of fuel is too low, despite multiple ignitions. However, in the case of large, convective fires this influence is neglected due to extremely high fire intensities. Consideration of convective fires, as an extreme fire event, lies beyond the focus of Reg-FIRM. Therefore, the generalisations made here to describe fire spread, and thus area burnt, at the regional scale are sufficient to reproduce the observed pattern.

The constant value for human-caused ignitions taken here seems to adequately represent a country-wide aggregate of all different regional motivations for humans to set fire in peninsular Spain (see Fig. 2-4). There are different factors, which could contribute to improve the simulated spatial patterns of both, number of fires and area burnt, since inconsistencies in simulated number of fires drive the final area burnt per grid cell (see equation (1)). The assumption that different life styles in rural and urban areas can be assigned to a certain population density, is very likely valid in only a few regions of peninsular Spain. A regionally specific formulation of *a(Nd)* (see equation (10)), describing regionally different land management systems in terms of their motivations to set fire especially in rural areas, could improve the simulation results. Another potential improvement would be the explicit consideration of land use and its changes in Reg-FIRM, or LPJ-DGVM as a whole.

This would reduce the potential area burnt in grid cells, where land use types such as settlement areas and permanent agriculture reduce the size of unmanaged land. A sharpening of the hitherto smooth spatial distributions of simulated area burnt would result (see Fig. 2-5), whilst not affecting the validity of the model concept.

In addition, inclusion of land cover and land use changes would lead to an increase in simulated number of fires and area burnt in regions, where potential area burnt increased in recent years due to the effects of land abandonment (re-vegetation and fuel build-up and therefore higher connectivity of the fuel bed), and where fire causes changed concurrently, e.g. shift from setting fire for pasture improvement to negligence due to tourist activities, (see (Pausas & Vallejo 1999). This might contribute to improve the model performance in the North-west of Spain.

In-depth studies should investigate the impact of these factors on the Iberian fire regime, to get a further insight into the interactions and feedbacks between fire regime, land use and vegetation composition.

Another possible improvement would be to implement an explicit lightning-ignition model, recognising lightning-caused ignition as a function of elevation, thus improving the results in mountainous regions. A re-parameterisation is required if Reg-FIRM were to be applied to new study areas, e.g. the values of lightning and human-caused ignitions, wind speed and fire duration must be changed. Implementation of a lightning-ignition model, however, would enhance the applicability of Reg-FIRM to other study areas, where for example the ratio between lightning and human-caused ignitions is higher.

Reg-FIRM can be incorporated in any dynamic vegetation model (DVM), given the DVM can provide the necessary input to drive the fire model. It can be used to investigate both the causes and effects of fire on vegetation and the long-term feedbacks of vegetation on the fire regime. One example is the simulation study presented here. The advantage of this model concept is that the model can be calibrated and the results validated using the type of data, which are usually collected in official fire statistics, such as fire danger, number of fires and area burnt. As was the aim of this study, it has been shown that human-caused fires can be included in regional-scale fire modelling using a mechanistic approach, and both climatic and human ignition potentials are needed to describe observed pattern sufficiently. The relative impacts of climate variability and human population development on fire dynamics can be assessed by sensitivity analysis within Reg-FIRM for representative regions, providing insight into socio-economic and biophysical components of global change. Therefore, Reg-FIRM can be seen as a useful tool to study the development of fire regime and vegetation under future global change conditions.

3. VEGETATION AND FIRE INTERACTIONS IN HUMAN-DOMINATED FIRE REGIMES

Kirsten Thonicke, Sergey Venevsky and Wolfgang Cramer

Abstract

The human influence on environmental processes has been described for many types of land use. One of the oldest tools to modify people's environment is fire, which has dominated fire regimes in many regions over long time scales. This paper focuses on two European case study regions, a Mediterranean-type and a temperate one, where 80 to 90% of the fires are human-caused and the ecological consequences are expected to be different. The objective in this study is to test the flexibility of Reg-FIRM, when applied to other regions, which have different levels of human-dominated fire regimes.

A sensitivity analysis, using historic climate input, is done to compare simulation results between the global fire model Glob-FIRM (Thonicke *et al.* 2001) and Reg-FIRM to check the importance of fire processes. Simulation experiments are then designed to explore influences of vegetation on fire. Vegetation composition influenced fire spread conditions in the temperate forest and had little impact on fire ignition potentials, except when only broad-leaved deciduous forests were assumed. Comparisons with observations allow interpretations of the factors governing a specific fire regime. The developed simulation tool can be used to assess implications for land management and effects of climate change on vegetation-fire interactions, which opens the tool to a wide range of potential users.

Keywords: regional fire model, human-caused fires, Spain, Brandenburg, vegetation-fire interaction

Introduction

With the Neolithic revolution, when humans started agriculture and pastoralism and first settled in small villages, use of fire in landscapes changed. Anthropogenic fires became dominant over naturally caused fires. Previously, humans had indirectly affected fire regimes as grazing pressure due to hunting changed the type and amount of fuel. The onset of controlled burning to modify natural fire regimes was new (Pyne & Goldammer 1997). Fires were used to restructure the landscape to human-specific purposes, such as open fields for agriculture and to maintain open grasslands for grazing. Pastures, agricultural fields and the forest understory were burnt to purify and fertilise the land with nutrient-rich ash before the next seed. Slash-and-burn agriculture has been practised in tropical, temperate and boreal forest areas for centuries to millennia. For this land use type humans used fire to deforest, prevent shrub encroachment and fertilise the soil. These were quite often only marginal soils with the consequence of dramatic harvest decline after only a few years forcing people to shift their cultivation area to another forested area (Goldammer *et al.* 1997; Jordan 1989; Saldarriaga 1989).

Examples have been found, where in historic and present time periods not only the frequency of fire changed influencing vegetation composition, but also the timing in the season to set fire (in order to achieve a specific effect on fuel consumption), fire intensity and post-fire effects changed (Keeley 2002; Pyne & Goldammer 1997; Vázquez & Moreno 1998; Yibarbuk *et al.* 2001). The effects human use of fire and their land use practices had on ecosystems, did not only depend on how intense they exploited their natural resources, but were also driven by climatic conditions and vegetation status. Ecosystems with slow litter decomposition, which are usually moisture limited, are very sensitive to modified fire regimes due to changes in ignition sources, fuel dynamics, moisture content, whereas changes in fuel dynamics would have less impact on fire regimes, where litter decomposition is faster. The co-evolutionary effects of vegetation-fire interactions are observed in a number of plant adaptive strategies affecting flowering, germination, re-sprouting to production of volatiles to enhance fire ignition (e.g. Bond & Midgley 1995; Davis 1998; Davis & Richardson 1995; Gignoux *et al.* 1997; Gill 1981a; Valette 1997; Whelan 1995).

Fire regimes have again been altered since the beginning of industrialisation as changes in labour facilities led to increased urbanisation and a reduction in land use pressure. Technological development improved agricultural techniques, thereby leading to abandonment of marginal agricultural fields, and the use of chemicals to replace fire to fertilise plots. The increasing use of fossil fuels changed the energy sector dramatically, e.g. reduced the pressure to use firewood for energy supply, thereby steadily decreasing the amount of biomass burnt (Pyne & Goldammer 1997). Industrialisation also changed the function of forests and their service to human societies. Commercial timber harvest at high efficiency now required the exclusion of fire to optimise economic gain. After heavy exploitation of forests in the middle Ages, harvests improved. With this change in policy fire suppression became a new tool. Exported from Central Europe to the rest of the world, it has dominated forest and fire policy in the Northern Hemisphere over the last two centuries (Pyne & Goldammer 1997; Pyne 1995).

Urbanisation and the expansion of transportation routes have also affected people's relationship to their environment with respect to fire. Fires, intentionally ignited to manage the landscape, became less important, while fires ignited by negligence started to increase in number. Modern, industrial societies are largely economically independent of local environmental resources and are characterised by more labour and leisure mobility. These socio-economic changes in human society should also have a consequence in their understanding of ecosystem function and service. These factors might explain geographical distributions of fires caused by negligence, which have local maxima along transport routes, in regions attractive for tourists, and around big cities, where people tend to spend their leisure time in nature (e.g. Korovin 1996; Moreno *et al.* 1998).

Peninsular Spain and the state of Brandenburg, Germany, were chosen as case study regions to investigate fire regimes and their interaction with vegetation. Both fire regimes are characterised by 80 to 95% of all fires being human-caused. Spain has a long history of pastoralism and agriculture and their associated use of fire (Pyne 1995). Fires are ignited intentionally to renew pastures and prevent shrub encroachment, to burn agricultural debris, to clear forest understory for improved gathering and hunting (Naturaleza 1996), but also caused to a large extent by negligence, reflecting an increasing influence of tourism on fire (Pausas & Vallejo 1999), patterns which are reflected in the national fire statistics (Moreno *et al.* 1998). Fire was used in Germany mainly until the 19th century to fertilise agricultural fields, renew heathlands, clear peats for cultivation and to clear forest understory to maintain timber and allow browsing and gathering (Goldammer *et al.* 1997). Technological development and establishment of forestry laws stopped the intentional use of fire. Today, the majority of fires in Brandenburg are ignited due to negligence (Landesforstanstalt Eberswalde 2000) and immediately suppressed to reduce the hazard to timber harvest.

These characteristics of both fire regimes, and their impact on vegetation dynamics require a socio-economic perspective to fire modelling inside vegetation models. The regional fire scale model Reg-FIRM (Venevsky *et al.* 2002) simulates human-ignition potentials and other fire processes using a mechanistic, process-based approach. This flexible model approach should enhance its application to other regions, because of the small amount of re-parameterisations required.

A sensitivity analysis will emphasise the role various fire processes have for the successful simulation of human-dominated fire regimes in their spatial distribution as well as inter-annual variability. Therefore, two fire modules Glob-FIRM (Thonicke *et al.* 2001) and Reg-FIRM, which model fire processes at different level of complexity and temporal resolution, are tested against each other. Both models are incorporated into LPJ-DGVM (Sitch *et al.* 2002) and were selectively used to simulate regional fire pattern for the forested areas in the state of Brandenburg, Germany, and for areas of peninsular Spain.

In simulation experiments historic fire regimes, given a specific human impact, and their interactions with vegetation dynamics will be investigated for Brandenburg. Differences between simulated and observed number of fires and area burnt, respectively, will be analysed. Implications of specific fire regimes for their associated vegetation composition and dynamics will be discussed for the two study regions. The following questions are of special interest in this study. Which fire processes become important in regional simulation studies? What can be learnt from both examples to improve our understanding of vegetation and fire interactions as well as fire regimes per se?

Methods

Sensitivity analysis

The two fire models, Glob-FIRM and Reg-FIRM, selectively running inside the LPJ-DGVM framework, are tested to prove their ability to reproduce mean values and inter-annual variability of observed data. In the case of Glob-FIRM, annual areas burnt (in ha) are calculated from fractional area burnt depending on the area covered by one grid cell. Annual areas burnt in Reg-FIRM are already area-specific. Results of both models are compared against observations using multiple regression methods and ANOVA statistics. Both models are applied to the two case study regions on a 0.5°x0.5° grid resolution over the time period of observation. Since observational data were available for forested areas of Brandenburg, simulation results were masked accordingly to allow comparison between observed and simulated number of fires and area burnt.

Vegetation cover in Brandenburg

In the LPJ-DGVM, spatial distribution of Plant functional types (PFTs) is simulated according to climatic, edaphic conditions, inter-specific plant competition and fire regime without any human intervention, i.e. potential natural vegetation. The onset of agricultural practices in Brandenburg goes back to between the 9^{th} and 14^{th} centuries, which were combined with intensive logging of natural forests. Forestry management started in the 19^{th} century. For reforestation fast-growing pines were preferred to the naturally occurring broad-leaved, deciduous trees. To reconstruct potential natural vegetation (PNV) for Brandenburg two methods were used. The vegetation composition in “islands of natural vegetation” was either extrapolated to sites of similar growing conditions, or secondary plant communities were replaced by their associated, potential plant communities (Krausch 1992). The resulting map of potential natural vegetation of Brandenburg was used to validate the simulated potential vegetation composition.

Actual vegetation composition as described in MUNR (1998), shows a majority of planted pine forests and the reduction of deciduous, broad-leaved forest to smaller islands. This digital map was aggregated and re-classified into a Plant functional type classification to allow comparisons with simulated vegetation. Pure stands were considered, when more than 40 % of the pixel was covered by a single PFT, mixed forest was defined, when both broad-leaved deciduous and needle-leaved evergreen forests covered more than 40%. The map of actual vegetation composition was used to prescribe the vegetation distribution for simulating fire regimes with actual, i.e. realistic, vegetation cover.

Fire simulations were run with potential natural vegetation, actual vegetation and broad-leaved deciduous forest. In the first case, the vegetation distribution was simulated by the LPJ-DGVM, in the other the LPJ-DGVM was run with prescribed vegetation types.

Data

Model parameters

Parameters used in the simulation experiments for both case study regions are summarised in Table 3-2. The parameters used in the application to Peninsular Spain are similar to those used in Venevsky et al. (2002).

Region of application	**Wind speed ($m*s^{-1}$)**	**Human-ignition Potential**	**Lightning - ignition Potential**	**Fire duration**	**Inward-flame emissivity**	**Fuel bulk density of temperate broadleaved summergreen woody[3]**
Brandenburg	1.28[1]	0.0825[1]	0.16[1]	1.0	0.3	22
Peninsular Spain	3.7[2]	0.22[2]	0.02[2]	1.0	0.3	22

Table 3-2 Model parameters used in Reg-FIRM. [1] (Landesforstanstalt Eberswalde 2000); [2] (Vázquez & Moreno 1998); [3] Modified according to measurements in Hély (2000)

Model input

All input data sets, used in the sensitivity analysis, were provided at a 0.5° x 0.5° longitude / latitude spatial resolution. The monthly climate data (precipitation and temperature) used for peninsular Spain were provided by the CCMLP project (Carbon Cycle Model Linkage Project) over the historical period 1860-1995 and derived from the data of Hulme (1995) and Jones (1994). For the historical Brandenburg simulation the LPJ-DGVM was run using CRU05 1901-1998 0.5° x 0.5° longitude / latitude monthly climate data, provided by the Climate Research Unit, University of East Anglia, UK. Historical CO_2 concentrations were derived from ice core and atmospheric measurements (Enting *et al.* 1994).

Soil texture information was obtained from the FAO soil data set (FAO 1991). Population density data from 1990 were extracted from the global population density on a 5' resolution and modified to a 0.5° x 0.5° spatial resolution (Tobler *et al.* 1995). The population growth rate for peninsular Spain over the period 1974-1994 was obtained from the Spanish National Institute of Statistics (Instituto Nacional de Estadística 1999). Similar data for population growth rate have been found for Brandenburg in (Statistik 2002)

For simulations of the Brandenburg fire regime under different vegetation conditions, all input data were provided at a 10' grid cell resolution. The historical CRU climatology at 0.5° x 0.5° was converted to a 10'x10' historical climatology using the 10' x 10' long-term mean climatology available at PIK for Europe for the 30 years period 1931-1960 (Cramer, unplublished data). This 10'x10' long-term mean climatology is considered the most comprehensive climatology for Europe. The 0.5°x0.5° long term mean CRU climatology was computed for the same period, then the annual 0.5°x0.5° CRU anomalies for all monthly climatic variables. The 0.5°x0.5° CRU anomalies were then spatially interpolated to 10'x10' (it is less risky to interpolate climate anomalies than absolute climate values). These 10'x10' anomalies were then added to the long-term mean climatology to produce a 10'x10' historical climatology for Europe (Bondeau, personal communication).

Results

Sensitivity analysis

Glob-FIRM (Thonicke *et al.* 2001) and Reg-FIRM (Venevsky *et al.* 2002) operate at different temporal and spatial scales and have a different level of complexity in their modelling approach. Glob-FIRM considers moisture content of litter in relation to vegetation-type dependent moisture of extinction, length of fire season and fire-resistance of vegetation types to survive fire. In Reg-FIRM fire ignition depends on the climatic fire danger, litter moisture content and the ignition potentials for human and lightning caused fires. Fire spread is a function of litter moisture, vegetation-type dependent bulk density and wind speed.

Multiple regressions were performed and ANOVA statistics obtained, which relate simulated time series of annual area burnt against observed data for forested areas of Brandenburg and for peninsular Spain. Results for the multiple regression coefficient, slope factor and F distribution for each model application were related to each other in a 3-dimensional graph to obtain the best fit in terms of correlation and variability with observed data (see Fig. 3-1). The smaller the bars and the closer they get to 1, the better they coincide with observed data.

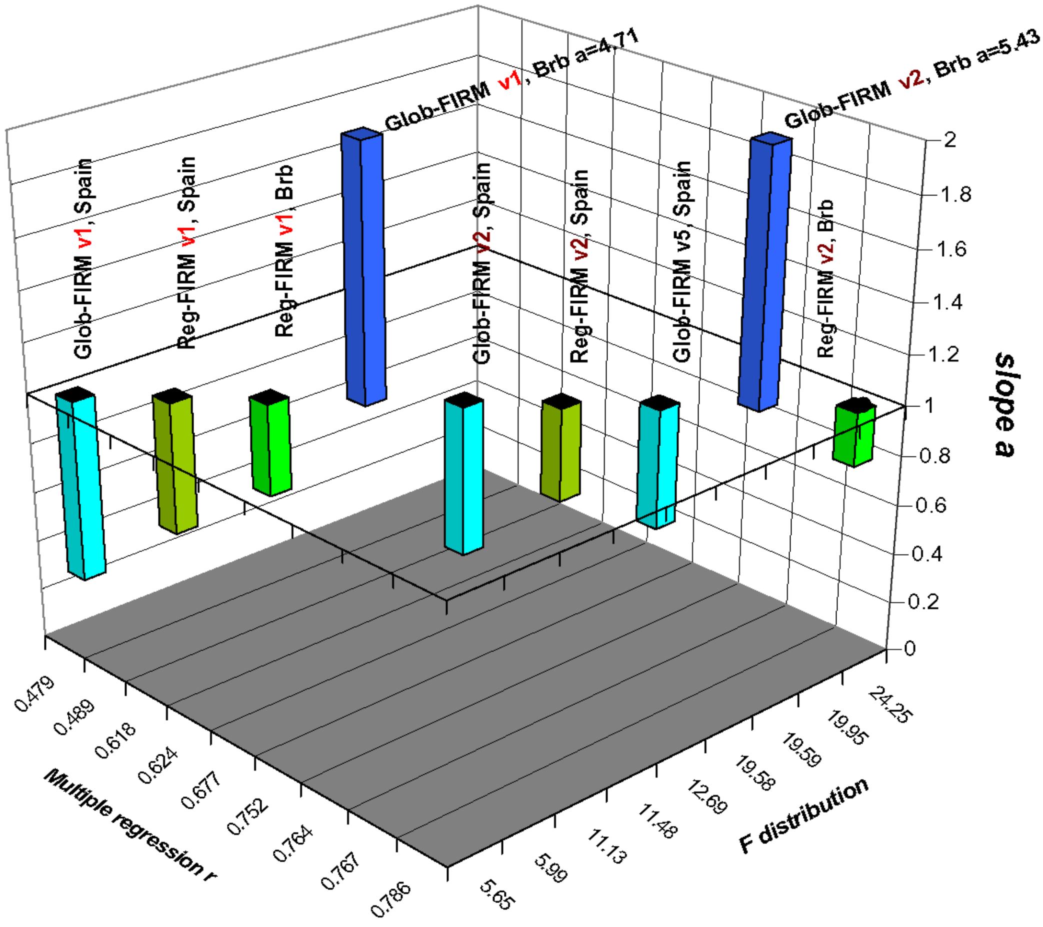

Figure 3-1 Relation of multiple regression coefficient, slope factor and F distribution for regression of simulated area burnt against observed area burnt. Blue colors Glob-FIRM application, green color Reg-FIRM application; v1- 20 observations, v2- 17 observations, i.e. 4 years with bad fit excluded. Note that the axes are not decimal, but ranking of the sample is categorical.

Glob-FIRM was the least capable to simulate the observed inter-annual variability and significantly overestimated annual area burnt in Brandenburg. Results did not improve notably, when 4 years of bad fit between modelled and observed area burnt were excluded. Reg-FIRM results coincided better with the observed inter-annual variability as well as correctly estimating the absolute values. Interactive regression analysis showed that by excluding 4 years of misfit, the regression results improved notably (r_{v1}= 0.489, a_{v1}=0.473 as compared to r_{v2}=0.752, a_{v2}= 0.629). In order to quantify the impact of human ignition potentials on area burnt, another simulation experiment using Reg-FIRM was made, where human population density was set to 1 person per km². The multiple regression coefficient was similar to that with spatially variable human population density, but the underestimation of area burnt was higher and the F distribution slightly smaller. The fact that the Brandenburg simulations of area burnt were better than those simulated for peninsular Spain, can be an artefact influenced by the size of the investigated regions. The state of Brandenburg is 14 times smaller than peninsular Spain.

Influence of vegetation composition on fire in Brandenburg

Broad-leaved deciduous forests Potential dominate the natural vegetation in Brandenburg. Only in the south on sandy soils, are natural forests dominated by pines, although stands are always mixed with broad-leaved deciduous trees. LPJ-DGVM however simulates mixed broad-leaved/needle-leaved forest for the entire region with needle-leaved increasing in their dominance southwards. Edaphic conditions, not well described in the vegetation model might be responsible for this difference. The lack of simulated dominant broad-leaved summergreen woody PFT might have an influence on the overall results leading to a less pronounced difference in simulated fire pattern between actual and potential natural vegetation (PNV).

Comparing the temporal dynamics between simulated and observed number of fires, one can see that the numbers of fires follow the sequence of minima and maxima distribution in time for all simulation experiments, except for three years (1979, 1985 and 1991; see Fig. 3-2). The model could not capture the magnitude of extreme fire years, where observed number of fires increased dramatically. The declining trend in number of fires, however, coincides with that observed. Under- and/or overestimations of number of fires are “transferred onto” the estimation of area burnt, following the hypothesis of Reg-FIRM, that all fires ignited in one time step have the same burning conditions.

Between 1987-1990 simulated and observed number of fires shows a good agreement, a feature, which cannot be found for the area burnt in the same years. In 1988 an number amount of human-caused fires resulted in the second largest amount of area burnt, whereas in subsequent years the number of fires, only slightly different from the year before, led to a relatively small amount of area burnt. The time series of observed area burnt is characterised by extreme inter-annual variability. A few peaks of extreme years are followed by a number of years with very little area burnt. Furthermore, considering the lower variability in number of fires, this pattern can be explained by fire suppression. Since all fires get extinguished by fire fighters and considering the fact that Brandenburg is rather densely populated and forests are easily accessible, no natural fire duration is possible as the climate conditions and other natural conditions would allow. Therefore, simulated area burnt can be understood to a certain extent, how large the area burnt could have been with a let-it-burn policy.

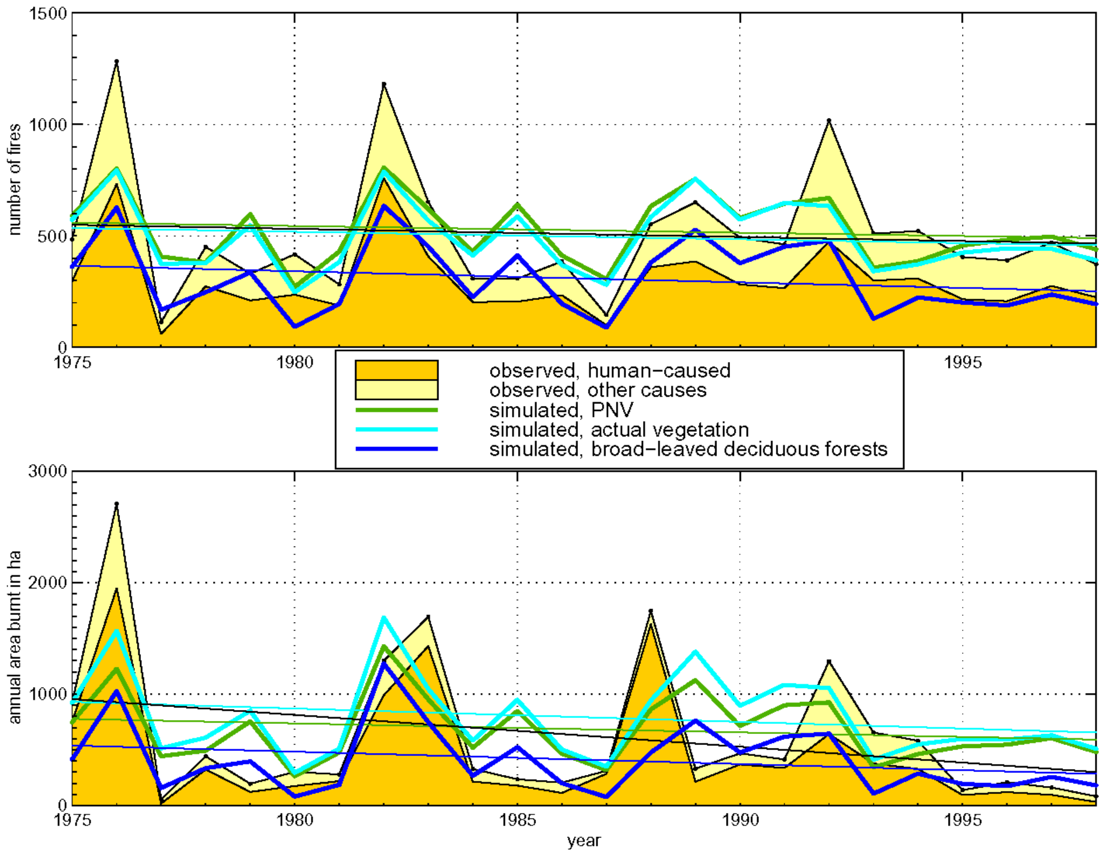

Figure 3-2 Annual numbers of fires (upper panel) and area burnt (lower panel) in forested areas of Brandenburg 1975-1998. Reg-FIRM was run on 10′ resolution. Filled areas are observed data: human-caused fires (orange), other fire causes (yellow). Simulation results with potential natural vegetation (green line), simulation results with actual vegetation (cyan line), simulation results with broad-leaved deciduous forests (blue line).

Comparing the simulation results for PNV with actual vegetation, it can be seen that vegetation composition seems to have little influence on fire ignition. Vegetation impact here is indirectly through litter moisture content and amount of fuel. The magnitude and inter-annual variability of number of fires for both experiments are almost identical. This is not the case for area burnt, where the range of simulated area burnt under actual vegetation is larger than under potential natural vegetation. This special influence becomes evident with prescribing exclusively broad-leaved deciduous forests. This is the only experiment, where vegetation composition influences fire occurrence a lot due to transpiration capacity. This is then transferred onto area burnt, where the high bulk density of the fuel additionally reduces fire spread and thus area burnt. Vegetation-type dependent fuel bulk densities have a direct impact on fire spread. Vegetation composition is therefore important for average burning conditions in forests. This behaviour can be of importance to forest managers, when reforestation and restructuring of species composition has to be decided, especially with respect to potential impacts of climate change.

Understanding fire regimes: Brandenburg vs. Peninsular Spain

For peninsular Spain one experiment was set under historic climate conditions, following the analysis that simulated vegetation distribution agrees with observations mapped by forest inventory (compare Naturaleza 1998). Simulated inter-annual variability for number of fires is in good agreement with observed data in the first two thirds of the simulation period. After 1989 discrepancies between observed and simulated number of fires increase dramatically. This is not transformed onto simulated area burnt. Here simulated area burnt coincide relatively well with observed data, except 1974, 1983 and 1989. Observed number of fires and area burnt are steadily increasing. Exceptional fire years, as seen in number of fires and area burnt, are depicted by local maxima in both number of fires and area burnt. Here, the model hypotheses, that all fires ignited in one time step, have the same burning conditions can be confirmed (see Fig. 3-3 upper left and lower left).

Study region	Number of fires	Area burnt	Number of observations
Brandenburg	0.738	0.169	25
Peninsular Spain	0.348	0.681	21

Table 3-2. Multiple regression coefficients

In areas with successful fire suppression, as is the case in forested areas of Brandenburg, discrepancies between simulated and observed area burnt are larger than for number of fires. Successful simulation of time sequences of number of fires does not result in a similar agreement between simulated and observed area burnt. Simulations using realistic, i.e. actual vegetation, however, lead to increased area burnt and greater range, but do not change minima-maxima sequences. Table 3-2 shows the multiple regression coefficients for simulation results of both study regions.

There have been a variety of hypotheses attempting to explain the sharp increase in number of fires, and also area burnt in peninsular Spain. Dramatic changes in the agricultural sector and further industrialisation reduced land use pressure that led to land abandonment, a fuel bed growing in size due to regenerating woodlands and shrublands, thus changing landscape structure. Changes in the energy sector reduced the pressure to collect firewood, together with new forest plantations of pines and eucalyptus forests started to re-grow and at the same time to produce more fuel, which is only slowly decomposing (Pausas & Vallejo 1999; Pyne 1995). These factors might have contributed to the increase of fire in peninsular Spain. Another possible explanation can be the improved fire detection and fire fighting, trying to reduce fire hazard since the late 80's. If fire is still used as a management tool in some parts of peninsular Spain, then fire suppression would have been less successful in these regions.

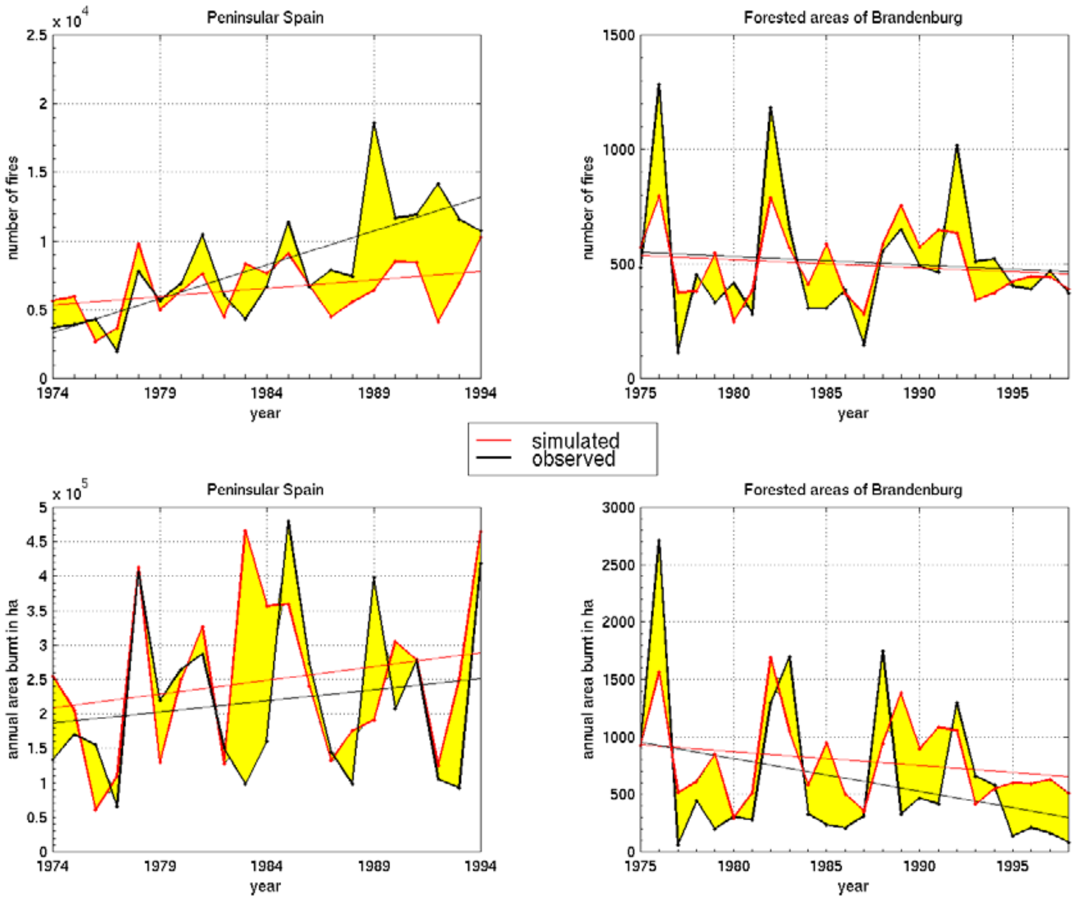

Figure 3-3 Numbers of fires and area burnt, simulated (red) and observed (black), for peninsular Spain 1974-1994 (upper and lower left) as published in (Venevsky *et al.* 2002), and forested areas of Brandenburg under actual vegetation simulated at 10'x10' spatial resolution (upper and lower right).

When people continue to burn the same area as they used to do in areas with fire suppression, the plots would have to be ignited several times (i.e. increase in number of fires) to achieve the same area burnt. The application of Reg-FIRM to peninsular Spain underestimates the number of fires in the time period of discussion. Reg-FIRM considers the same fire duration for all fires, as well as the same burning conditions for all fires ignited in one time step. Following this hypothesis, an underestimation in number of fires should result in underestimation of area burnt as well. But simulated areas burnt coincide rather well with the observation in these years. This fact can lead to the conclusion that in a fire regime, dominated by intentional fires, fire suppression has a limited success, when people still use fire as a tool to manage the landscape for their purposes. They try to optimise the area burnt, no matter how many ignitions would be necessary.

The opposite is true in the forested areas of Brandenburg. Here, the discrepancy between simulated and observed number of fires is smaller than for area burnt. The majority of fires are ignited by negligence; people do not have a specific interest to burn the forest. Fire suppression reduces fire duration and thus fire size, resulting in numerous small-scale fires. Except for a few extreme fire years, where it seems to have been hard to fight fires, they get extinguished successfully.

Conclusions

These results of the sensitivity analysis show clearly, that in order to successfully simulate the temporal patterns of area burnt at the regional scale both vegetation impact and human ignition potentials need to be considered in a simulation model. Glob-FIRM is an adequate model to reproduce the average fire pattern at the global scale, but the fixed relationship between length of fire season and fractional area burnt is not robust enough to capture the temporal pattern in specific regions. Formulations found in Reg-FIRM to represent climatic fire danger and impact of vegetation on fire ignition and spread are at the right level of complexity to simulate human-dominated fire regimes. Furthermore these results also allow us to conclude that the LPJ-DGVM is capable to reproduce litter moisture content, fuel dynamics, vegetation composition and their dynamics to give the correct preconditions for the fire model.

Vegetation composition has not obviously a crucial effect on fire ignition in temperate climates, but on area burnt as the comparison between potential natural vegetation and actual vegetation in Brandenburg has shown, with the exception of broad-leaved summergreen forests. Its importance will possibly increase, when under climate change conditions, fire risk and spread conditions change and trees can differently adapt. In fact, this can be an important issue for forest management, when new reforestation plots are planned.

These experiments also show that applying a simulation model like Reg-FIRM to regions, which are characterised by human-dominated fire regimes, can help in understanding the underlying mechanisms. Reg-FIRM can be applied to regions, not only different in climate and vegetation, but also having different fire management. The interpretation of fire regimes shown in this study shows the influence of land use change on vegetation and fire dynamics. Further investigations that identify, which land use changes determine the changes found in fire regimes, are required. Dynamic links to these factors are crucial, because they can help to understand their implications under changing environmental conditions that are not only climate change, but also land use change.

Acknowledgements

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4. FIRE AND VEGETATION DYNAMICS IN A CHANGING CLIMATE

Kirsten Thonicke and Wolfgang Cramer

Abstract

Fire is likely to change under climate change conditions. The complex interactions between vegetation productivity and climate conditions in temperature vs. moisture-limited ecosystems make projections about shifts in fire regimes and vegetation dynamics difficult, unless the full range of fire processes and interactions with vegetation dynamics is captured in a simulation model. Therefore, to explain the links between climate drivers and environmental processes under climate change and rising CO_2, simulation experiments were conducted, in which one GCM output was applied.

The LPJ-DGVM including the global-scale fire model Glob-FIRM was run using the ECHAM4 climate change scenario based on the IS92a emission scenario to investigate its effects on fire and vegetation dynamics. Patterns of changes in fire season versus biomass burnt were analysed. Changes in length of fire season do not always imply the same rate of change in biomass burning, but can be modified by either the non-linear relationship between length of fire season and fractional area burnt or changes in vegetation productivity and thus fuel limitation. Latitudinal averages of biomass burnt and net ecosystem production show how fire can not only influence the magnitude of net carbon exchange, but also its sign. Results outline the importance of a dynamic link between vegetation and fire processes in simulation models. This is also true for the regional climate change application, where the regional-scale fire model Reg-FIRM was applied to two human-dominated fire regimes for the HadCM3 climate change scenario based on the A1FI SRES emission scenario. Changes in climatic fire danger seem to have a delay in their response to climate change. When considering vegetation impact on fire spread, changes in area burnt occur earlier and are more pronounced. Fire changes had no effect on vegetation composition in Brandenburg. Increases in grass cover in Peninsular Spain led to large increases in fire.

Introduction

Fire as a natural disturbance is driven by abiotic as well as biotic environmental factors and influences nutrient cycling, vegetation dynamics, and especially the associated regeneration pattern. Fire releases carbon, aerosols and other trace gases to the atmosphere that influence atmospheric chemistry and climate. Varying in space, time and magnitude fire changes according to its climatic and biotic drivers. These relationships are hypothesised to become critical environmental factors in a changing climate that are likely to intensify climate change effects on ecosystem functions.

Disturbances have been identified to increase the seasonal carbon cycle in high-northern latitudes, an effect that could not be explained by climate warming alone. After disturbance in these ecosystems, evergreen woody vegetation and mosses are replaced by deciduous woody vegetation and grasses. The latter have shorter photosynthetic active season length and are, due to their young age, more productive. Decomposition accelerates after the removal of the insolating moss layer, which leads to enhanced heat transfer into the soil and deepened soil thaw depth. Given the fact that disturbance rates increased during the last decades, the observed increases in seasonality of atmospheric carbon was attributed to disturbances (Zimov *et al.* 1999).

A temperature increase in ecosystems, where temperature limits the length of the fire season, would substantially increase fire risk and thus carbon release from biomass burning given the large carbon storage in those forest ecosystems (see estimates given by Kasischke 2000; Stocks *et al.* 1998). More complicated would be the effects of climate change on fire, where precipitation limits vegetation growth and thus fuel production that despite fire-critical climate conditions, fires occur only periodically or episodically. Any climate change that enhances vegetation productivity, would then sustain fire in the following dry season. Kitzberger (2001) investigated similar relationships between climate variability, vegetation productivity and fire in southwestern United States and northern Patagonia, Argentina, which are influenced by El-Niño-Southern-Oscillation (ENSO). Here, the sequence of El-Niño and La Niña events give an example, how associated changes in fuel moisture conditions, in vegetation growth and in length of dry seasons influence fire occurrence. Increased moisture enhanced vegetation growth that increased fuel availability under El-Niño conditions and led to higher fire occurrence and widespread fires under La Niña climate conditions of reduced precipitation and increased temperature.

Furthermore, many fire regimes are human-dominated. They are characterised by active fire management, i.e. the use of fire to shape vegetation structure and composition, and/or by accidentally caused fires (see also chapter 3). Changes in land use that are partly associated with urbanisation processes increase the amount of accidentally caused fires as observed in Peninsular Spain. An associated reduction in landscape heterogeneity is assumed to additionally increase the total area burnt (Moreno *et al.* 1998; Rego 1992; Vélez 1997). Environmental conditions, which lead to fire disturbance and effects of changed human impact on fire regimes need to be investigated in the context of vegetation-fire interaction, especially under climate change conditions.

Previous climate change studies investigated effects of carbon and of nutrient cycles on high-latitudinal ecosystem productivity (using the hybrid v4.1 DGVM, White *et al.* 2000), effects of changing fire disturbance on NPP (using CENTURY 4.0 Peng & Apps 1999), or changes in forest fire potentials (Stocks *et al.* 1998). White (2000), Peng (1999) and (Kasischke 2000) modelled fire as a stochastic event, based on historic fire return intervals, which were also applied under climate change conditions. These estimated fire impacts are non-mechanistic and therefore independent of environmental conditions that are likely to change under a changing climate. Other studies concentrated only on one aspect of the fire regime (lightning (Price & Rind 1994), fire season (Flannigan & van Wagner 1991), fuel loads (e.g. Bessie & Johnson 1995; Keane *et al.* 1996b). Climatic fire danger indices, such as the Canadian Fire Weather System (FWI) applied to 4 different General Circulation Models (GCMs) in Stocks (1998) for the circum-boreal zone, allow investigations of climate driven impacts on fire occurrence, but neglect vegetation-fire interactions. Mouillot (2002) studied changes in fire effects and the regeneration pattern for a Mediterranean-type ecosystem, but did not include effects of rising CO_2. All these studies represent only one part of the ecosystem interactions under climate change conditions.

In this context, process-based fire modules that cover important fire processes and are dynamically linked to dynamic vegetation models, become important, especially when effects of climate change are to be investigated. These aspects, which have not been considered completely in one simulation study, will therefore be the focus of this study. The fire model Glob-FIRM (Thonicke *et al.* 2001), which takes litter moisture and fuel load into account to estimate fire occurrence and length of fire season to calculate area burnt, will be applied to climate change conditions. For two sample regions with human-dominated fire regimes the regional-scale fire model Reg-FIRM (Venevsky *et al.* 2002), which explicitly considers human- and lightning caused fires and has more process-based simulation approaches for fire occurrence and spread, is applied.

The Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM), in which two fire models can selectively run, allows investigation of climate change impacts on the terrestrial biosphere, such as the effect of CO_2 fertilisation on water use efficiency and vegetation productivity, and thus fire occurrence and heterotrophic respiration. In this study we want investigate the interactions between fire and vegetation would face under climate change conditions at the global and regional scale. The main focus is to explain the links between climate drivers and environmental processes, when climate is changing, therefore one GCM scenario is used, respectively. The following questions will be addressed to investigate possible dynamics.

What will be the effects of climate change on fire in temperature vs. moisture-limited ecosystems?

How will biomass burning emissions change in relation to length of fire season, when CO_2 fertilisation affects increases water use efficiency and is likely to reduce fire occurrence?

How will global change, as expressed as the effects of economic and human population growth on GHG emissions, affect human-dominated fire regimes? What will be the effects on vegetation?

Methods and Data

To address the above questions simulation experiments were designed using the LPJ-DGVM (Sitch *et al.* 2002). Fire models were only recently incorporated allowing simulations of fire and its interactions with vegetation. The global fire model Glob-FIRM will be used to investigate climate change effects on average fire conditions and related vegetation changes at the global scale. It takes the daily fire probability into account that is determined by litter moisture status. Following the hypotheses that with longer persistence of burning condition fire sizes increase, given a minimal fuel amount, the annual length of the fire season determines the fractional area burnt. Vegetation composition influences fire occurrence due to plant-specific moisture of extinction; vegetation productivity and litter fall are responsible for dead fuel load. The latter is a critical variable in moisture-limited ecosystems, where fuel load limits fire spread and thus area burnt. Fire effects depend on the plant survivorship defined on a per Plant Functional Type basis. It opens bare ground for establishment of new seedlings and initiates regeneration.

Compared to Glob-FIRM the regional-scale fire model Reg-FIRM considers more fire processes covering fire occurrence and fire spread, such as human and natural ignitions sources, climatic fire danger and explicitly calculates rate of spread. Furthermore, vegetation composition and status has a larger influence on fire spread through litter moisture and fuel bulk density, weighted per PFT present. This fire model is therefore applied to study impacts of regional climate change on human-dominated fire regimes and their implications for vegetation dynamics. For this in-depth analysis two study regions have been chosen, the forested areas of Brandenburg, Germany, as an example of a temperate ecosystem and Peninsular Spain as a region covering ecosystems of warm-temperate to Mediterranean-type climates. The successful simulation of historic fire pattern and vegetation dynamics has been proven for both regions and was shown in chapter 3.

General Circulation Models provide future climate projections in combination with certain emission scenarios that imply assumptions about future human population and economic growth (Houghton *et al.* 2001). An analysis of GCM scenarios by the IPCC (Cubasch *et al.* 2001) outlined a variety of possible climate changes as consequences of different emission scenarios that differ in their magnitude as well as their regional differences. The global simulation experiment was run using anomalies of ECHAM4 GCM climate change scenario that used the IS92a CO_2-Scenario including sulphate emissions (Roeckner *et al.* 1996). These climate anomalies were calculated from historic climate data (1968-1998), provided by the Climate Research Unit in East Anglia, UK, which were subsequently linearly interpolated to 0.5°x0.5° spatial resolution (Schaphoff, unpublished data). In the case of regional climate change applications, the HadCM3 GCM scenario for the A1Fi SRES emission scenario, provided at 10'x10' grid cell resolution by Mitchell (2002), were used in the simulation experiments to show the effect of an extreme future climate projection on ecosystem processes.

Soil texture information was obtained from the FAO soil data set (FAO 1991). Population density data from 1990 were extracted from the global population density on a 5' resolution and modified to a 10' x 10' spatial resolution (Tobler *et al.* 1995). The population growth rate for peninsular Spain over the period 1974-1994 was obtained from the Spanish National Institute of Statistics (Instituto Nacional de Estadística 1999). Similar data for population growth rate have been found for Brandenburg in (Statistik 2002). Under climate change conditions, human population growth was kept constant at the 1999 level as a first approximation.

A qualitative analysis was conducted to estimate feedbacks between vegetation and fire under climate change conditions by applying Glob-FIRM to the global scale. Relative changes in the length of fire season and biomass burnt, both averaged over 2070-2100, were compared against their historic averages (1970-2000). Both were additionally compared against each other to distinguish the influence of climate and that of vegetation dynamics on fire emissions. For each, combination categories were defined, which can be explained by the specific causes of climate or vegetation change (see Table 4-1).

Effects of changes in temperature and precipitation under rising CO_2 emissions are likely to have effects on fire that are different between temperature-limited and moisture-limited ecosystems, and thus their associated fire regimes. Drastic temperature increases could cause an increase in the length of fire season and lead to dramatic increases in biomass burnt given the large amount of biomass in forested ecosystems. In moisture-limited ecosystems, however, precipitation changes could lead to qualitatively different effects on fire depending on vegetation productivity and length of fire season. Here, increases in precipitation could enhance vegetation productivity that - given sufficient length of fire season - increase fire-related emissions, because fuel build-up would no longer limit fire spread. Only drastic precipitation increases or CO_2 fertilisation effects could lead to decreasing length of fire season and therefore reduce biomass burning. Changes in litter carbon as simulated by the LPJ-DGVM and changes in climate given by the scenario are different in their magnitude and spatial distribution.

Changes in fuel production affect fire spread and thus biomass burning and explain qualitative changes in biomass burning, which are not (only) affected by changes in length of fire season. Enhanced vegetation growth can increase dead fuel load and shift the amount just above the minimal threshold (see Thonicke *et al.* 2001) and thus increase fire given a regular recurrence of dry seasons in a particular ecosystem. To assess its influence, changes in the number of years that have fuel limitation, were compared between historic (1970– 2000) and transient conditions (2070 and 2100). Five classes explain changes in occurrence of fuel limitation per se and, when occurring in both time periods, magnitude changes were assessed.

No. of category	**Relative change in length of fire season *L***	**Relative change in biomass burnt *B***	***B - L***	**Influence**	
				Climate	**Vegetation**
1	>1	>1	>0	Increase in fire-critical moisture conditions	Increase in biomass burning more due to enhanced vegetation growth than due to increase in length of fire season
2	>1	>1	<0	Increase in biomass burnt only driven by increase in length of fire season	
3	>1	<1	<0		Decrease in vegetation productivity limits amount of biomass burnt
4	<1	>1	>0		Decrease in length of fire season compensated by enhanced vegetation productivity that results in higher emissions
5	<1	<1	>0	Decrease in length of fire season larger than decrease in biomass burnt	
6	<1	<1	<0		Decrease in length of fire season smaller than decrease in biomass burnt, influence of vegetation productivity

Table 4-1. Interpretation of relative changes in length of fire season versus relative changes in biomass burnt

The influence of biomass burnt on net ecosystem production (NEP) was investigated by analysing the changes in different components, which constitute NEP with

$$NEP = (NPP - R_h) - BB,$$

where NPP is the net primary production, R_h the heterotrophic respiration and BB the biomass burnt. To emphasize changes in the influence of biomass burnt on NEP anomalies were calculated from historic conditions, averaged over 1970 to 2000. Latitudinal averages were obtained from areas that are characterised by relatively similar ecological conditions, and were produced for NEP and $(NPP- R_h)$, respectively. Changes in biomass burning were obtained by the same methods and complement each plot of a specific latitude band. Anomalies of NEP lower than anomalies from *(NPP- R_h)* alone would occur in regions with increasing fire, showing its growing influence. Anomalies of NEP larger than those of NPP-r_h alone would imply decreasing influence of fire on NEP as compared to historic conditions, and thus a gain in carbon sequestration due to the reduced carbon loss.

Results

Changes in fire at the global scale

Increases, compared to the historic average, in both length of fire season and biomass burnt (category 1 and 2 in Fig. 4-1, upper right panel) are observed in many parts of the world, especially in tropical and sub-tropical dry ecosystems, but also in the high northern latitudes. The causes, however, are manifold. Whereas in the high-northern ecosystems, despite increasing precipitation, a dramatic warming leads to increases in fire (which is also accompanied by increases in vegetation productivity) in some tropical and subtropical regions reduced precipitation seems to drive increases in biomass burnt (compare areas of category 1 with precipitation anomalies). The majority of forest ecosystems is characterised with an increase in length of fire season, but a smaller increase in biomass burnt (category 2). Despite of dramatic warming at the end of the 21st century, which has been limiting ecosystem processes in boreal regions, these are still small enough to limit corresponding increases in biomass burnt (see shape of function A(s) relating length of fire season and fractional area burnt in Thonicke *et al.* 2001). Furthermore, in these regions, vegetation carbon is increasing by only a small amount, in some parts even decreasing, so that the potential storage of biomass that could be burnt per unit area is limited. Given the reductions in precipitation in the central and southern Amazon, Central Africa and partly in South-East Asia, increase in biomass burnt, larger than the increase in length of fire season, could be expected. The amount of change relative to its historic values, seems to be still tolerable by the affected tropical rainforests and can possibly explain, why the increase in biomass burnt is smaller than the increase in length of fire season (category 2). Apart from Amazonia, the temperature increase is relatively small in the other affected tropical regions.

A reduction in precipitation and therefore fuel load is responsible for reduced fire-related emissions despite increased length of fire season (category 3) in dry tropical and subtropical ecosystems, but also in some Mediterranean-type ecosystems. Here, changes in vegetation dynamics are now limiting fire spread, and therefore buffering climate change effects on fire emissions. Most of these regions either face increasing fuel limitations or fuel limitation first appeared in the transient period (see Fig. 4-1 upper left panel). The opposite effect is seen in semi-arid regions of Africa and Asia, where enhanced vegetation productivity due to increased precipitation reduces fuel limitation of fire spread and changes fire occurrence (category 4; see also changes in fuel limitation in the corresponding regions, Fig.4-1 upper left panel). These patterns cannot be explained by climate-driven effects on fire alone, but together with vegetation dynamics.

If precipitation exceeds a critical level, fire emissions can be reduced also due to reduced length of fire season (category 5 and 6) as seen for large regions in western South America, Central Africa and Southeast Asia. Smaller increases in precipitation that are accompanied by dramatic increases in temperature reduce length of fire season more that biomass burnt below their historic average in some central-boreal and northern-boreal regions, but also in the temperate zone of North America. Looking at the climatic changes increases in fire emissions would be expected. Here, it could be assumed that increased water use efficiency seem to buffer climate effects and result in reduced length of fire season, especially in the northern part of the American mid-west.

Given the relative changes in length of fire season and biomass burnt, the actual contribution of biomass burning to net ecosystem production is of special interest. Including fire in the estimation of net ecosystem productivity reduces the increasing trend that would occur for the case *(NPP-R_h)*, in latitude bands, where fire is steadily increasing relative to its historical average. Depending on the actual amount of biomass burning the divergence of the ten-year running averages differ (compare Fig. 4-2 a)-d) with corresponding plots of biomass burning anomalies). In these regions the carbon balance is very likely to be very sensitive to changes in fire. The influence of fire vanishes in the mid- and high-northern latitudes, when fire is either largely fluctuating compared to its historic average (Fig. 4-2 e) and g)) or even decreasing (Fig. 4-2 f)). In the first case, both trends merge at the end of the simulation period or show no difference, when the amount of biomass is too small to have an effect on NEP. In the case of decreasing biomass burning, both trends diverge at the end of the simulation period, showing less influence of fire on the carbon balance under climate change conditions.

Temperate ecosystems in the southern hemisphere are characterised by a large increase in length of fire season and biomass burnt, which is also seen in the latitudinal anomalies of biomass burnt (Fig. 4-2 a)). A growing number of years have a NEP lower than the historic value, whereas the opposite case can be seen for NPP-r_h values. Here, fire is mainly responsible for changes in NEP under climate change conditions. These patterns are less pronounced in savannah regions (Fig. 4-2 b)), where climate change have qualitative different effects on fire (compare Fig. 4-1). Enhanced vegetation productivity together with decreasing fuel limitation seems to be responsible for increasing biomass burning in the tropical and subtropical regions between 20°S and 30°N (Fig. 4-2 c) and d)). Whereas the inner-tropical zone is facing carbon loss, the tropical zone northwards experiences increasing fire that still leads to diverging trends in NEP, but these ecosystems generally take more carbon up than under historic conditions.

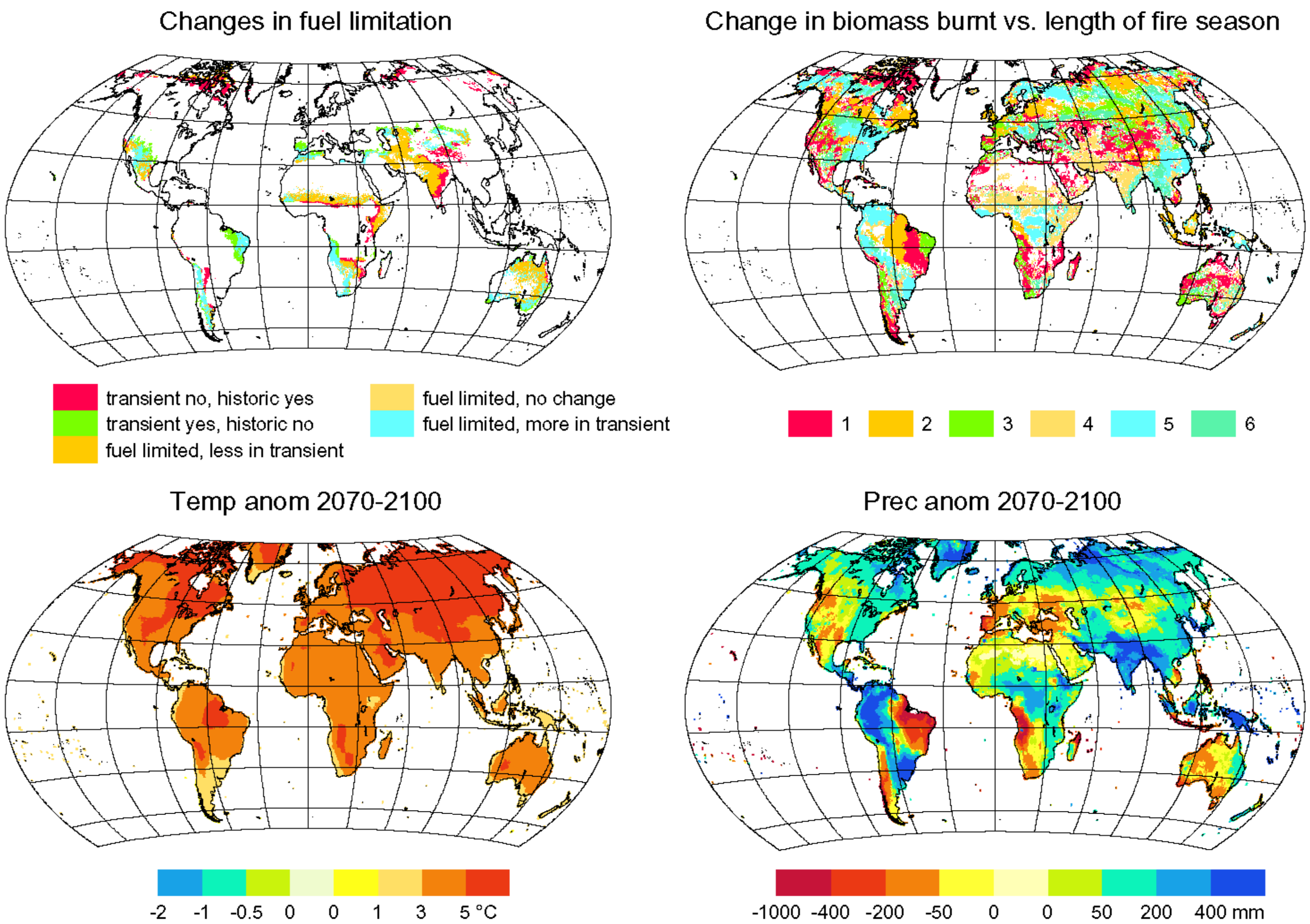
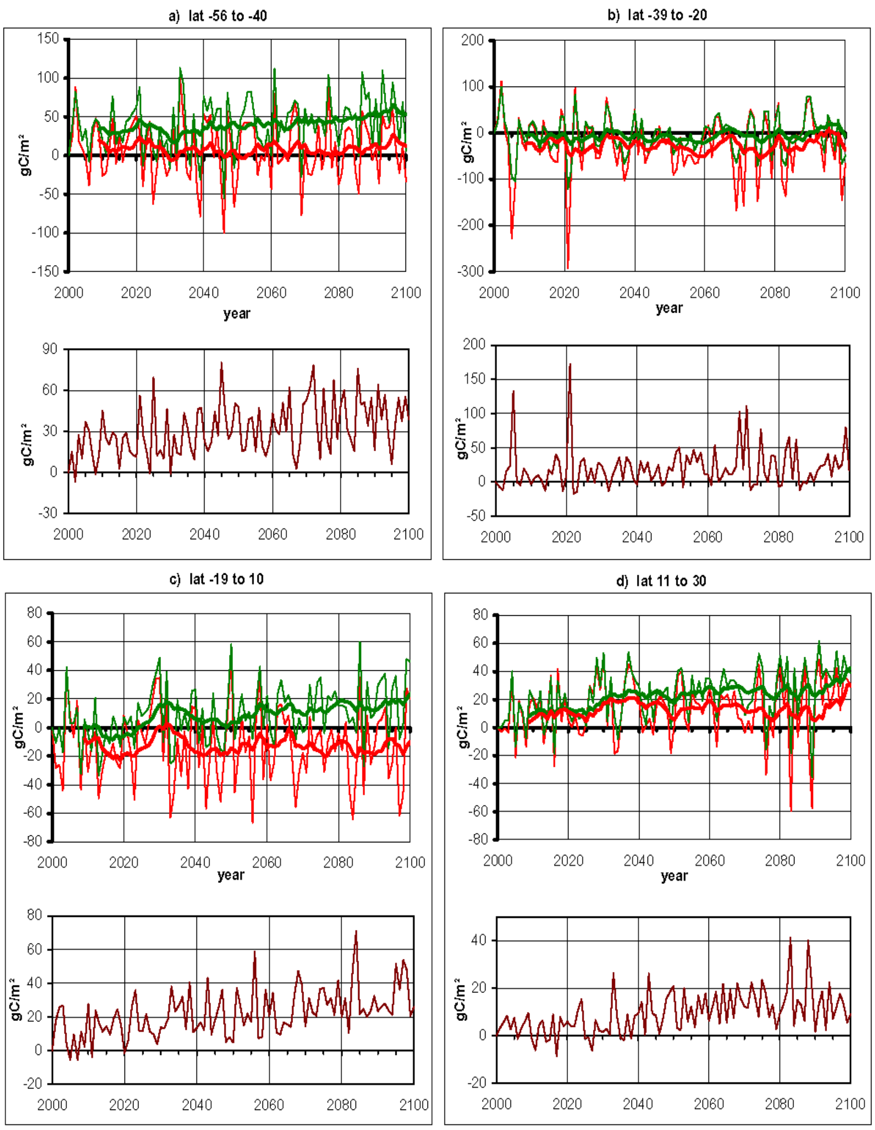


Fig. 4-1 Interpretation of relative changes in biomass burnt versus relative changes in length of fire season between transient (2070-2100) and historic (1970-2000) average values (upper right); simulation using Glob-FIRM inside the LPJ-DGVM. For explanation of categories see Table 4-1, white: no changes. Upper left: Changes in fuel limitation that affects fire spread between transient and historic time period; white: no influence. Lower left: temperature anomalies. Lower right: precipitation anomalies.

The mid-latitudes have an exceptional pattern in NEP. Changes in biomass burning do not show a clear trend in the first decades of the 21st century, but mostly increase after 2050 (Fig. 4-2 e)). Thus, less carbon is lost compared to historic conditions in the first period, but increases with increasing fire. Therefore, the trend of both NEP and *(NPP-R_h)* estimates overlap in the last decades of the simulation period. The high variability in these estimates also imply that heterotrophic respiration as well as NPP is largely fluctuating, where fire seems to be a relatively small contributor. Most of the ecosystems in this latitude band are semiarid and some face increasing fuel limitation under changing climate conditions. Others show increases in biomass burning, which seem to be too small in absolute terms to have a visible effect on the balance (see Fig. 4-1).



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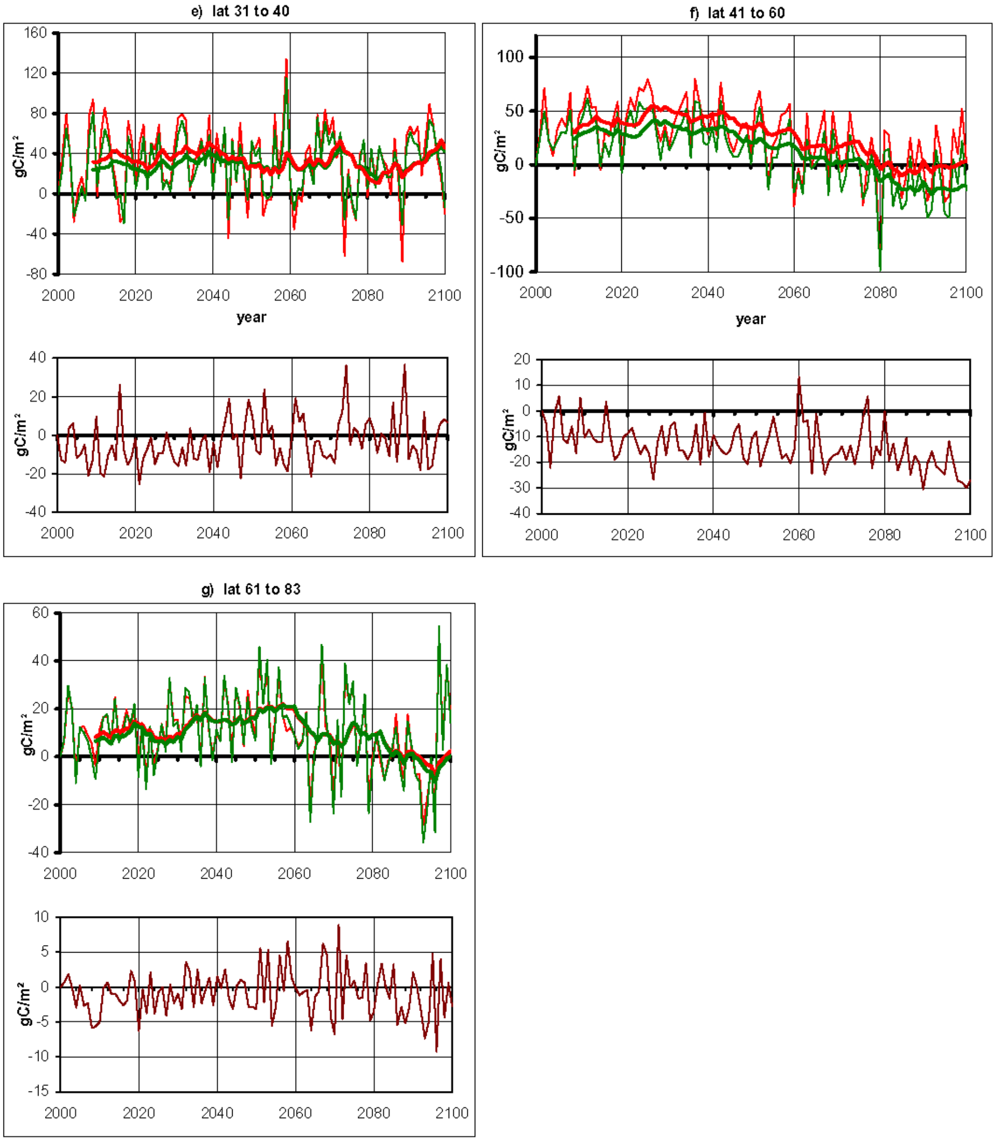

Fig. 4-2 Latitudinal changes in NPP-r_h (green) and NEP= NPP–r_h-BB (red) expressed as anomalies from the average over 1970-2000; anomalies in biomass burning (brown) averaged over the same latitudes complement each plot. Thick lines represent 10-year running averages. NPP- annual net primary production, r_h – heterotrophic respiration, BB – biomass burnt.

In most temperate and boreal ecosystems the extreme warming together with increasing precipitation in large regions leads to increases in NEP until 2060, the estimates thereafter fall below the historic average (Fig. 4-2 f)). Since, fire is decreasing less carbon is lost in NEP and the anomalies are larger than for *(NPP-R_h)* only. Many of these regions show increases in length of fire season, but smaller increases in biomass burnt, or have even a decreasing fire pattern (see Fig. 4-1). Climate warming in these regions, especially in boreal Eurasia, seem to increase heterotrophic respiration more than fire or NPP. Given the large carbon pool, this process seems to become the driving force of carbon loss.

In the high-northern latitudes, fire-related emissions start to increase in amount and variability after 2050 that are overridden by large fluctuations in NPP and heterotrophic respiration (see Fig. 4-2 g)). Both estimates show little difference in their quantity and variability. All this can be explained by the immense warming in these areas (see temperature anomalies in Fig. 4-1) and also relativise the dimension of increase in biomass burnt, driven by climate and enhanced vegetation productivity (category 1 in Fig. 4-1, upper right). Fire is increasing, but given the small quantities its influence on the carbon balance is relatively small.

Regional environmental changes

Brandenburg

Fire and vegetation pattern were simulated under climate change conditions, which assume high increases in daily temperature range and mean temperature during the summer month, but decreasing precipitation during summer and increasing precipitation in winter month for Brandenburg. Choosing the A1FI SRES emission scenario implies that these drastic changes in climate are accompanied by extreme increase in atmospheric CO_2, which could possibly buffer increasing drought conditions, and thus reduce fire risk and effects on vegetation dynamics as seen in Fig. 4-3.

The climatic fire risk is constant over the first half of the 21st century, and decreases notably after 2065. Annual areas burnt follow the sequence of minima and maxima distribution of the climatic fire risk, but not in all years. Furthermore, by comparing the temporal dynamics of the fire danger index with area burnt (Figure 4-3a) and b), see cubic polynominal fit added to both graphs) one can see a time delay, where the fire danger decreases almost a decade after area burnt was already decreased. This underlines the importance that both, climatic fire danger and the vegetation impact on fire spread are important to consider in climate change studies. Vegetation can therefore buffer fire risk to a certain extent given the fact that CO_2 fertilisation increases water use efficiency of plants.

Thus, vegetation dynamics were less and less affected by fire during the 21st century (compare Figure 4-3 b) and c)). Broad-leaved summergreen woody PFTs increase foliar projective cover (FPC) only in the last decade of the simulation period. Variations are relatively small and not always caused by fire dynamics. Needle-leaved evergreen woody PFT show little change in their FPC under climate change conditions. Small decreases in the second half of the century favour grasses, allowing them to increase their FPC. In general, potential natural vegetation in Brandenburg is able to sustain climate change conditions.

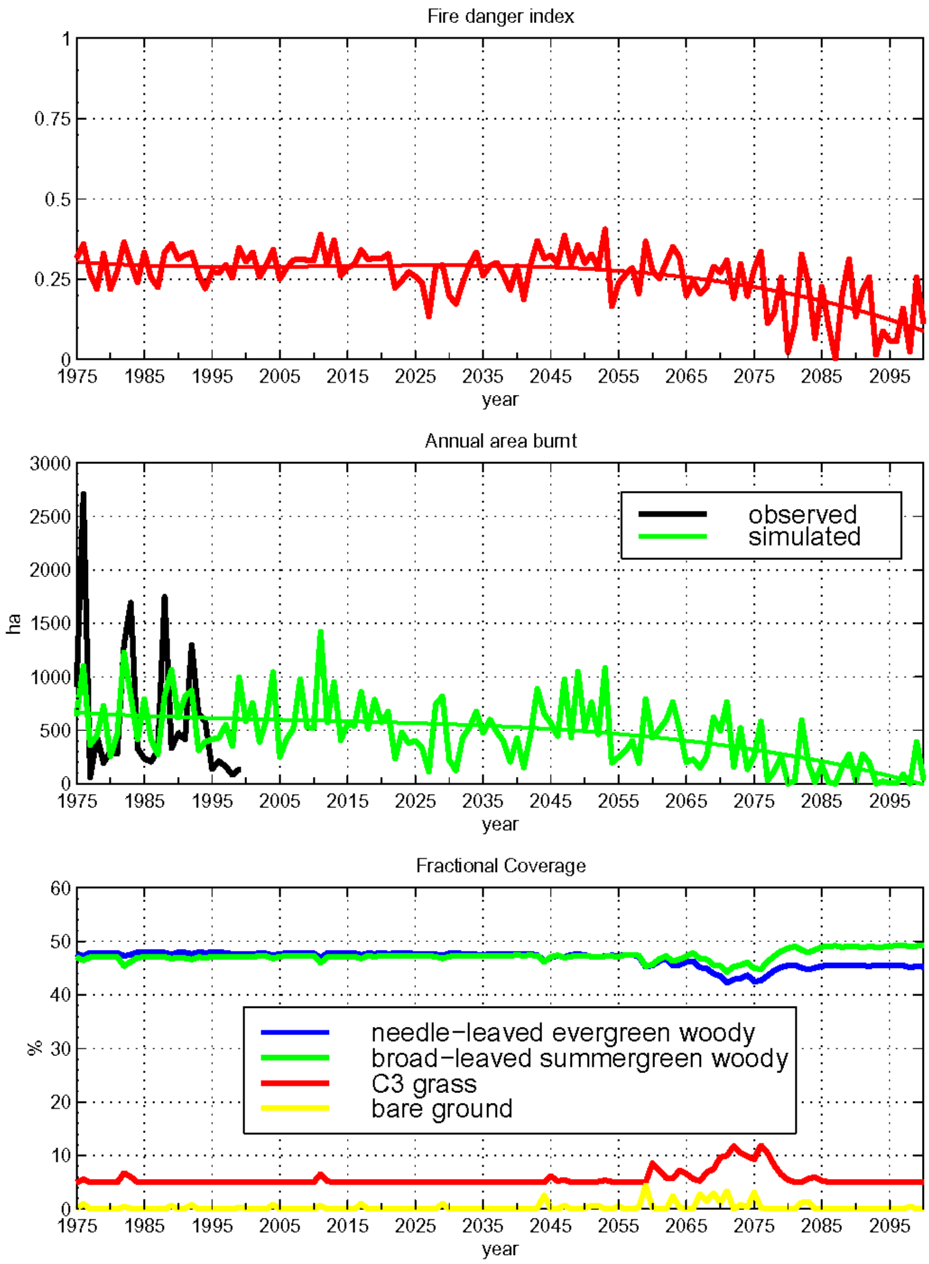

Figure 4-3 Temporal pattern of (a) fire danger index as used in Reg-FIRM, (b) area burnt and (c) foliar projective cover of PFTs under potential natural vegetation composition in forested areas of Brandenburg 1975-2100 at 10'x10' resolution using the HadCM3 climate change scenario with A1Fi SRES scenario.

Peninsular Spain

Under climate change conditions (HadCM3 A1FI scenario) peninsular Spain experiences large increases in summer temperatures, large decreases in precipitation mainly in the northern part of the peninsula that shorten the winter rain season. The effects on daily temperature range are relatively large compared to the other climate variables (Mitchell 2002). The potential of the vegetation to buffer these changes are possibly very low.

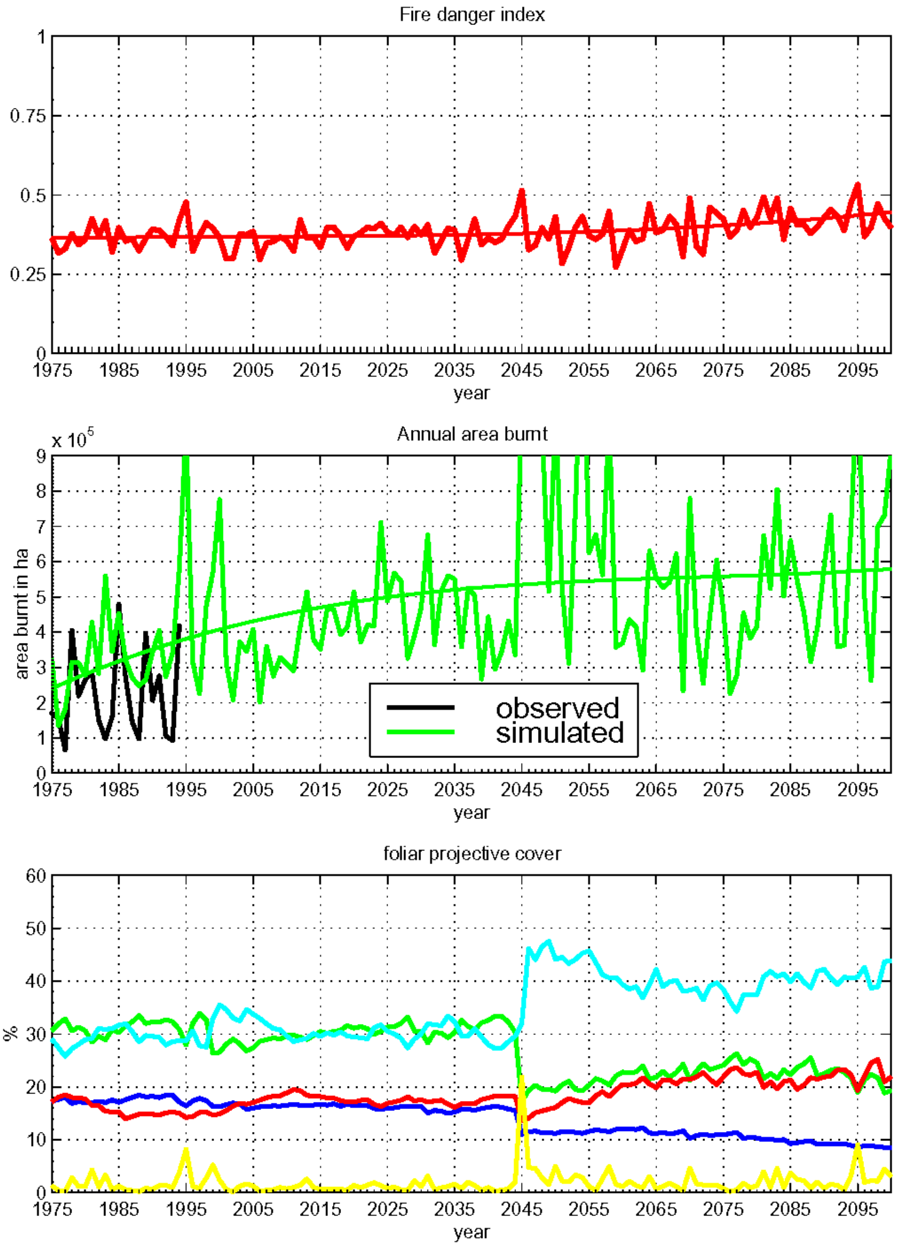

Figure 4-4 Temporal pattern of (a) fire danger index as used in Reg-FIRM, (b) area burnt and (c) foliar projective cover of PFTs in Peninsular Spain 1975-2100 (blue: temperate needle-leaved evergreen woody, green: temperate broadleaved evergreen, red temperate broadleaved summergreen, cyan C_3 grass, black: bare ground) at 10'x10' resolution using the HadCM3 climate change scenario with A1FI SRES scenario.

These effects of climate change conditions on fire are very large in peninsular Spain (see Fig. 4-4). Climatic fire danger increases notably only after 2040, whereas annual area burnt is almost doubled already in the first decades of the 21^{st} century. A large shift in fire is caused, when after 2045 grass cover increases and therefore principally changes the conditions for fire spread. Since grasses have a 8 times smaller fuel bulk density (see Table 1 in Venevsky *et al.* 2002) an increase in its fractional coverage can increase the rate of spread, and thus area burnt, also with smaller decreases in fuel moisture conditions as illustrated in Fig. 4-4. Here, changes in vegetation composition enhance climate change effects on fire, which leads to a new quality of the fire regime. These interaction change vegetation composition and reduce its potential to buffer climate change effects.

The climate change conditions reduce the cover of temperate needle-leaved woody plants, and also broad-leaved evergreen woody are decreasing in cover, which are replaced by grasses. Temperate summergreen woody take surprisingly an advantage under climate change conditions and its advantages in productivity seems to make them more competitive over the entire peninsula. This can be an artifact of the LPJ-DGVM that lets all Plant Functional Types to occur in any place, wherever its bioclimatic and biotic condition would allow their growth, but does not consider migration of plants. Realistically, broad-leaved summergreen woody plants would not be able to migrate so far in the time frame of the simulation experiment, therefore the ecological impact of this effect should be neglected.

Discussion

In this study the effects of climate change on fire under potential natural vegetation at the global and, considering human-ignition potentials, at the regional scale have been exemplarily shown. It was shown, how fire drivers could influence fire effects under changing vegetation and climate conditions using the fire models Glob-FIRM and of Reg-FIRM. The regionally different combinations of changes in fire drivers can have various effects on biomass combustion and also be influenced by other ecosystem processes that do not directly influence fire processes.

When calculating $(NPP\text{-}R_h)$ for major ecosystems, except for boreal and arctic regions increasing trends have been shown. But the consideration of dynamically simulated biomass burnt in the ecosystem carbon balance shows that fire can change the qualitative characteristics of the balance, such as trend and amplitude. Increases in vegetation productivity enhances also fire occurrence in moisture-limited ecosystems, changing the NEP from being increasing to rather constant levels. The carbon balance in boreal and temperate ecosystems is mainly driven by changes in heterotrophic respiration the influence of fire is small. Here, the conjunction with heterotrophic respiration is very sensitive in the temporal dynamics of NEP and underlines the importance to simulate both processes dynamically in vegetation models.

In boreal ecosystems, the effect of climate warming on increasing length of fire season and thus biomass burning differ in their spatial occurrence from patterns of increasing carbon loss due to increasing heterotrophic respiration. Biomass burning has increased compared to its historic average, but less than the increase in length of fire season. The increase in living biomass is relatively small, and thus the amount of biomass that could be burnt per unit area. Although temperature-limited, the effect of climate warming on fire regimes of long fire return intervals seems to be limited. Only, if the temperature increase is so large that despite of increased precipitation and increasing vegetation productivity, therefore changing fuel limitation, the fuel gets sufficiently dry, so that both length of fire season and amount of biomass burnt increase compared to their historic value as seen in the high-northern latitudes. Its influence on NEP, however, is small. Increases in length of the fire season together with changes in seasonal pattern under climate change conditions were also simulated for boreal Eurasia and Canada by coupling the Canadian Fire Weather (FWI) system to four different GCMs (Stocks *et al.* 1998). Our results, however, seem to underline that increases in fire danger do not automatically lead to increases in area burnt that would also affect carbon storage. Environmental conditions that contribute to a specific fire regime and can be expressed in a specific fire return time, are likely to change, especially under climate change conditions. Interpretations of the influence of fire on NEP under these climate conditions by applying a constant area burnt (corresponding to a historic fire return time) as done in White (2000) still explain the contribution of vegetation and soil to NEP only, it neglects the dynamic interactions between these two processes and fire. It is very likely that patterns of carbon storage change with the consideration of dynamic fire module, as seen in this study. Given the intense climate warming, changes in heterotrophic respiration are reducing NEP below its historic average as simulated by the LPJ-DGVM, a result opposite to White's findings.

Effects of climate change and vegetation dynamics in moisture-limited ecosystems very much depend on the amount of precipitation change and to lesser extent on the amount of temperature increase. Three different effects can be seen semi-arid regions, which have one driving component, the vegetation productivity and the seasonality of dry seasons. Enhanced vegetation productivity that support fire spread due to increased fuel production has been simulated for tropical and subtropical ecosystems. Large increases in precipitation, however, can reduce the length of fire season and thus biomass burnt. On the other hand, vegetation cannot buffer increasing drought conditions even under increased atmospheric CO_2 concentrations with higher water use efficiency. Increasing fuel limitations affect fire spread and reduce biomass combustion. Effects of CO_2 fertilisation on water use efficiency under increasing drought conditions seems to be restricted to a specific combination of climate change and patterns of vegetation dynamics, also implying that vegetation has a limited potential to buffer such climate-related stress.

To simulate the historic pattern of human-dominated fire regimes, human ignition causes have been considered in the fire model Reg-FIRM. This was used in two regional climate change applications, taking a SRES emission scenario that assumes large increases in carbon emissions with corresponding consequences for climate change pattern. These experiments have shown that changes in the climatic fire danger are only one aspect of changing fire regimes. Furthermore, changes in fuel characteristics caused by shifts in vegetation composition can intensify trends given by the climatic fire danger, but already at an earlier stage than one would expect from analysing the trends in climatic fire danger alone. This is because vegetation composition has a direct influence on fire spread through fuel bulk density and vegetation and climate due to moisture conditions of dead fuel. If precipitation is reduced below a critical threshold (and changed in its seasonality), vegetation cannot buffer these effects with implications for vegetation and fire as seen in the simulation experiments. They underline the importance of this particular link; its implications can now be quantified in simulation experiments. Given the changes in climate, types of land use change that affects the use of fire as a management tool in landscapes are now even more sensitive elements. Changes in potential human-caused ignitions could even intensify simulated patterns of changing fire regimes, first of all in Peninsula Spain. These results show one example of possible effects of climate change in order to focus on changes in links between drivers of changes and affected environmental processes. Its reliability will have to tested in applications of other climate change scenarios. Nevertheless, they do underline the importance to investigate mitigation strategies to cope with effects of land abandonment, of pasture management and therefore plan forestry policy accordingly.

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