22 Fire Spread Regulated by Weather, Landscape Structure, and Management in Wisconsin Oak-Dominated Forests and New Jersey Pinelands

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Abstract This study is to predict fire spread behavior and burned area across two fire-prone landscapes with contrasting vegetation (Oak-dominated ecosystem in WI vs. pine-dominated ecosystem in NJ), fuel-type composition, and land-use history regulated by the effects of weather, landscape structure and land management by combining simulations from three models (FARSITE, HARVEST, AND FRAGSTATS) under different scenarios. The results demonstrate: ① substantial differences in fire-spread patterns between the two landscapes were observed when holding weather conditions constant

J. J. Qu et al. (eds.), *Remote Sensing and Modeling Applications to Wildland Fires*

and excluding roads, indicating that landscape fragmentation is a main controlling factor on fire spread at the landscape level; ② roads functioning as barriers could significantly reduce the burned area from fire spread; and ③ Harvesting effects showed different trends, depending on landscape fuel type composition and weather conditions. At 4% harvesting intensity, both clustered and dispersed methods showed no significant impact ($a = 0.01$) on reducing the mean burned area across the more fragmented WI landscape, but showed significant effects on fire spread in the less fragmented NJ landscape in summer when weather was hot and dry.

Keywords Fire spread, burned area, landscape, fire model and weather condition

22.1 Introduction

There is a growing concern over catastrophic fires in the U.S. in recent years (http://www.usgs.gov/themes/Wildfire/fire.html). These fires can degrade environmental quality and wildlife habitat, destroy buildings and cost human life, reduce wood production, and increase firefighting costs. For example, the unusual fire season of 2000 consumed 3 million hectares across the country. Nearly a billion and a half dollars alone were spent on fire suppression in 2002—the second largest fire season in the last 50 years (Graham 2003).

At landscape level, wildfire spread and behavior are determined by many factors, including weather, fuel loading and type, topography, landscape structure, road density, and human activities and land use (Pyne et al., 1996; Rollins et al., 2002). While the physics of fire spread are well known at fine spatial and temporal scales (e.g., individual trees and forest stands, seconds to hours) (Rothermel 1983), examination of factors determining the variability of fire spread patterns over entire landscape is more complex and deserves greater attention.

The prediction of burned area (BA) across the landscape is a very important element in planning fire suppression efforts and reducing wildfire damage. Fire effects are often examined at the stand level after a single fire event and are difficult to link with landscape-scale processes without the application of models (Keane et al., 1989). Modeling is an efficient and practical approach to help understand and predict fire effects at landscape scales or larger because models simplify complex processes and systems for improved human understanding (Kercher and Axelrod 1984; Hunt Jr. et al., 1999; Finney and Andrews 1999; Miller and Urban 1999). It also allows us to test various fire cause-and-effect scenarios from economic, physical, social, ecological, and management perspectives that are almost impossible to be discerned in the field.

Timber harvesting is a major human-derived disturbance that alters landscape structure, thus affecting fire spread. Previous reports on the subject have generated conflicting, sometimes contradictory findings. On one hand, some recent data (1980 through 1999) suggest that logging activity increase BA (http://www.wildrockies.org/wildfire/), because logged areas tend to increase rate of fire spread and fire intensity by increasing fuel loading and creating additional corridors (Huff et al. 1995; Anderson 1982). On the other hand, harvesting can fragment the fuel complex and disrupt local fire growth, thereby increasing fire suppression effectiveness (Stratton 2004). Nevertheless, quantifying the effects of different harvesting methods on fire spread across landscapes is rare.

This study will examine the effects of weather, landscape structure, and land management on fire spread in two landscapes. We combine 3 models to conduct hypothetical simulations over a 15-day period to obtain a more-complete picture of how changes in landscape patch heterogeneity and fuel type composition could affect landscape fire spread. This Chapter is designed to answer four specific questions from a landscape perspective: ① How fire spread is affected by landscape structure? ② Does harvest increase or decrease surface fire spread across the landscapes? ③ Is there a significant difference in harvesting methods on fire spread? And ④ does the above influences on fire spread vary by seasonal, and, if so, to what degree?

The definition of landscape structure in this study refers to: ① the heterogeneity and spatial arrangement of different fuel patches across the entire landscape, and ② fuel type compositions across the landscape; because changes in either one can significantly affect the simulation results.

22.2 Methods and Materials

22.2.1 Study Areas

Two temperate forest landscapes in the eastern USA (Fig. 22.1) were selected for examining landscape-level effects on fire spread and behaviors. Both are fire-prone ecosystems with contrasting vegetation (a Oak-dominated ecosystem in WI vs. pine-dominated ecosystem in NJ), fuel-type composition, and land-use history (a highly fragmented and managed landscape in WI vs. a less fragmented landscape without harvesting activity for the last 100 years in NJ).

22.2.1.1 NJ Pinelands

The 38,150 ha sub-area of the Pinelands used in this study is flat with mean elevation of 39 m, ranging from 12 – 65 m. The majority of the study area is State Forest and a NJ wildlife management area, and some land is in federal ownership on Fort Dix Army Base. Thus, recent human interference (e.g. harvesting and urbanization) is minimal. The long-term (1930 – 2004) annual mean temperature of the area is 12.0℃, and annual precipitation is 1123±182 mm. The pine-oak

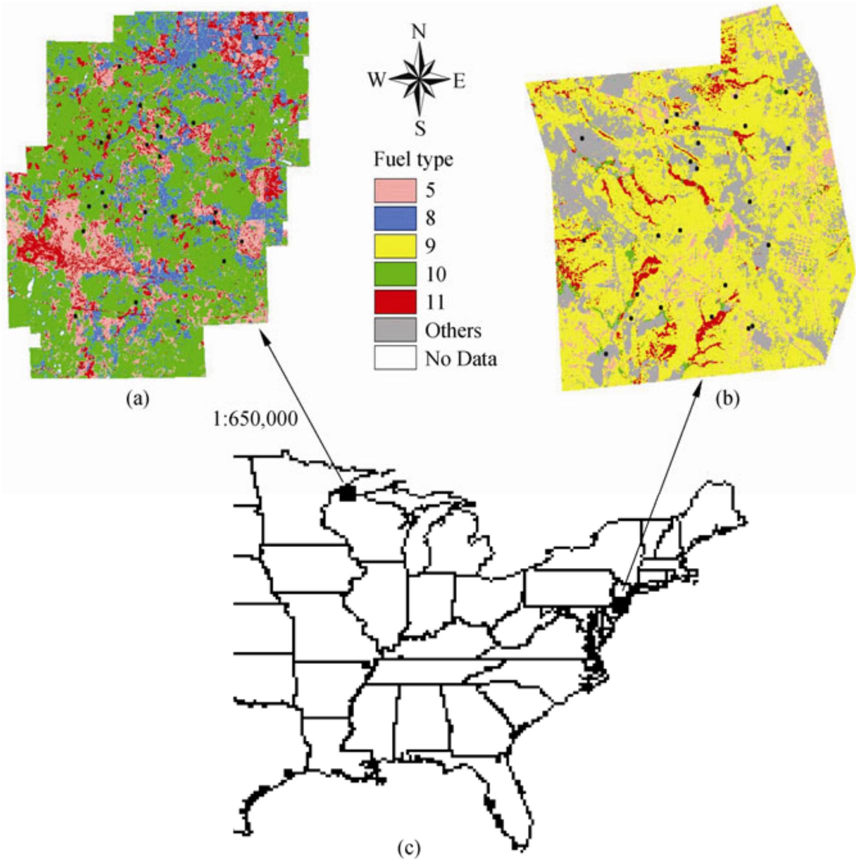

Figure 22.1 Spatial distribution of 24 fire ignition points in relation to fuel types in the Chequamegon National Forest (a) and New Jersey Pinelands (b) USA. Fuel type 5 = Brush < 0.8 m with scattered trees, 8 = Litter layer without under story, 9 = Pinelands/brush, 10 = Litter layer with under story, and 11 = light logging/Swamps. There are 5 more fuel types in the NJP landscape but all were grouped into the category of others to illustrate substantial difference in fuel type composition between the two landscapes

landscape is characterized by highly volatile fuels and sandy soils with a low water holding capacity (http://www.fs.fed.us/ne/global/research/fire.html).

New Jersey's Pinelands occupy 22% of the state (http://www.fs.fed.us/ne/global/pubs/maps_posters/pdfs/firepine.pdf) and depend on fire to regenerate and maintain its composition and structure. Sixty-two percent of the Pinelands are upland forests comprised largely of three communities; ① Oak/Pine, consisting of black oak (Quercus velutina), chestnut oak (Q. prinus), white oak (Q. alba), and pitch (Pinus rigida) and shortleaf pine (P. echinata), ② Pine/Oak, consisting of pitch pine with mixed oaks in the overstory, and ③ Pine/Scrub Oak, consisting of pitch pine with scrub oaks (Q. ilicifolia, Q. marlandica) in the understory (McCormick and Jones

1973, Lathrop and Kaplan 2004). All stands have ericaceous understories, primarily huckleberry (Gaylussacia bacata) and blueberries (Vaccinium spp.). Sedges, mosses and lichens are also present. Wetland forests communities include Pitch Pine lowlands, mixed Oaks, and Atlantic White Cedar (Chamaecyparis thyoides) swamps.

22.2.1.2 Wisconsin Oak-Dominated Forests

The 39,350 ha study area is located in the Washburn Ranger District of the Chequamengon national forest (CNF) in Northern Wisconsin, USA. The topography of the area is flat to gently rolling with elevations ranging from 232 – 459 m. The area in general has deep, coarse-textured soils. The climate is characterized by a short/hot summer with a growing season of 120 – 140 days, and cold winters. Annual precipitation ranges from 660 – 700 mm (Albert 1995) while annual mean temperature is about 4.7℃. Historically, the area was occupied primarily by jack pine (Pinus banksiana Lambert), red pine (Pinus resinosa Aiton), and several oaks (Quercus spp.) and is now dominated by red oak (Quercus rubra L.), sugar maple (Acer succharum Marsh), paper birch (Betula papyrifera Marsh), and aspen (Populus spp.) (Radeloff et al. 1999). Six dominant land-cover types in the study area are Oak, jack pine, red pine, mixed Oak/conifer, regenerating forest/shrub, and non-forested bare ground (Bresee et al. 2004). The dominant fuel type in the area is forest with under story (50.2%) after the cover types were grouped into fuel types following Anderson's (1982) classification system.

22.2.2 Study design

22.2.2.1 Fire Ignition Locations and Fuel Type Assignments

For each landscape, 24 fire ignition points were randomly generated after being stratified by major fuel types (13 nationally recognized fuel categories; Anderson 1982). There were 3 to 6 replicates for each fuel type depending on its weighted area of the total landscape. The fuel maps were developed using the 2001 land-cover map from Bresee et al. (2004) for the CNF and the 2001 land-cover map provided by the Grant F. Walton center for remote sensing and spatial analysis (CRSSA), Rutgers University (Lathrop and Kaplan 2004) for the NJP. Land cover, fuel type composition and patch mosaics differ appreciably between the 2 landscapes (Table 22.1, Fig. 22.1).

22.2.2.2 Management Scenarios and Simulations

We considered roads as fire barriers in our FARSITE simulations and compared these to the results from control runs without road effects. The CNF road map was provided by the USDA Forest Service (http://www.fs.fed.us/r9/cnnf/ftp/forest_files/). We used paved (road level B) and gravel with greater width (road level C) roads for comparison purposes. For the NJP road map, we used primary highway and secondary roads from the US Census Tiger Dataset (www.esri.com/data/download/census2000_tigerline/index.html#datainfo).

To detect effects of landscape structure and fuel type composition on fire spread (Question 1), hypothetical simulations were conducted by applying 15-day weather data collected in August 2 – 17, 2002 in both landscapes (i.e., weather was kept constant while varying the landscapes). In this particular test, roads were excluded to simplify the analysis. Additionally, we created two alternative fuel maps by assigning different fuel types to some land-cover types in the NJP to detect how changes in fuel-type composition could have on BA. One map had a similar mean rate of fire spread (MROS, 173 m/hr) to that in the control landscape of CNF (183 m/hr). The other map had a much higher MROS for NJP (334 m/hr) than that in the control landscape, which we think is closer to reality due to differences in fuel type composition (pine vs. oak) between the two landscapes. The MROS was calculated using the rate of spread defined by Anderson (1982) for various fuel types and multiplying by their corresponding fractions of land area to the entire landscape, then summed up to mean burned area (MBA, ha).

To illustrate how forest practices can affect fire spread across the landscape (Questions 2 and 3), we used the HARVEST 6.1 model (Gustafson and Crow 1996) to generate hypothetical landscapes imposing 4% cutting with two different methods, clustered (clearcuts) and dispersed (select cutting), in both the CNF and the NJP, then compared the results to those in the control landscapes.

To examine seasonal variation of fire spread (Question 4), we used daily weather data in August of 2002 (8/2-8/17) and April of 2004 (4/3-4/18, the April data in 2002 was unavailable and April data in 2003 had missing days) in the CNF recorded by meteorological equipments mounted on an eddy covariance flux tower (Noorments et al. 2004). The meteorological measurements were programmed to record data every 20 seconds and output 30-minute means to a data logger. In the NJP, meteorological data were collected from three weather towers, one located in each of the dominant upland forest communities; an Oak/Pine forest at Silas little experimental forest, a Pine/Oak forest at Fort Dix, and a Pine/Scrub oak forest located at the Cedar Bridge fire tower. Continuous meteorological measurements were made from each tower. Meteorological data was recorded with automated data loggers (CR23x, Campbell Scientific).

The selections of data periods (hereafter referred to as spring and summer, respectively) were determined using our best judgment on usefulness of weather data and its availability at each landscape. For example, August of 2004 in the NJP was affected by the hurricane season and there was a 1000-year storm event on 7/12 (21.7 cm of rain in one day) and consequently, June to July data were used for the NJP landscape. The simulated results from these data are useful for illustrating general patterns of seasonal fire spread and comparison between the two landscapes because fires are affected by weather in addition to other factors. We also conducted sensitivity analyses to examine the effects of wind speed on fire spread by increasing wind speed to 40 km/hr in both landscapes.

22.2.3 Model Linkage and Applications

22.2.3.1 FARSITE

Quantitative understanding of the interactions between landscape structure and disturbances such as fire spread is one of the major focuses of landscape ecology. Three models (FARSITE, HARVEST, and FRAGSTATS) were linked to conduct integrative analyses (Fig. 22.2). We used a fire growth model FARSITE (Finney 1998) to simulate fire spread across our landscapes. The model can predict both surface and crown fires, however surface fire spread was only used to simplify the analyses for comparison because most fires in the CNF are low-intensity surface fires (Sturtevant et al. 2004) and both landscapes are relatively flat. The required inputs for FARSITE include two ASCII files for weather conditions and five grid layers at 30 m resolution representing landscape structure, vegetation and topography. The five layers were: ① elevation, ② slope, ③ aspect, ④ fuel type, and ⑤ degree of canopy closure (Fig. 22.2). The topographic files were derived from 3-arc DEM data. The canopy closure file was developed and rescaled to 0% – 100% based on the Normalized Difference vegetation Index (NDVI) values (0 – 1) that were calculated from the red and infrared channels of the Landsat 7 data (Rouse et al. 1973). The Landsat TM-based fuel maps were developed from the land-cover maps in the CNF (Bresee et al. 2004) and NJP (Lathrop and Kaplan 2004) and categorized by Anderson (1982) fuel type (Table 22.1).

It is also well known that weather and landscape structure interact to affect fire spread. Thus, separating these two effects is desired for increasing our understanding of fire ecology and providing insights for improved fire management at the landscape level. In each landscape, we ran the FARSITE model for 24 randomly selected fire ignition points stratified by major fuel types within a 15-day period.

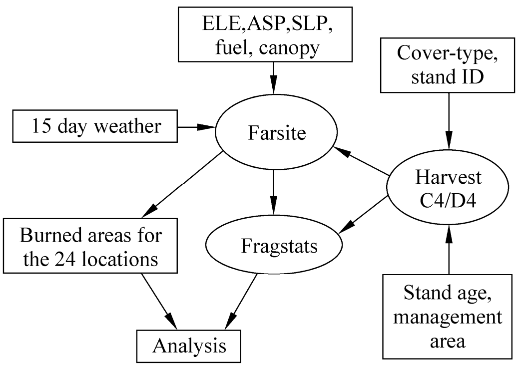

Figure 22.2 General flow chart of model linkages and simulations of fire spread in the Chequamegon national forest WI and New Jersey Pinelands. The rectangular boxes represent inputs and outputs, while the ellipse boxes represent the models used in the analyses

Table 22.1 Landscape percentages in major land-cover types and Anderson (1982) fuel types for the Chequamegon National Forest (CNF, oak-dominated forests), WI and the New Jersey Pinelands (NJP, pine dominated)

Landscape	Cover Type	Fuel Type	Proportion (%)	Definition
CNF	Regenerating forest/shrub, Jack pine	5	23.6	Brush < 0.8 m with scattered trees
	Red pine	8	16.1	Litter layer without under story
	Hardwoods and Mixed forests	10	50.2	Litter layer + under story
	Clearcuts	11	10.1	Light logging slash/Swamps
NJP	Lightly developed/unwooded, managed grasslands	1	0.5	Short grass < 0.3 m
	Lightly developed/wooded	2	1.2	Timber grass with under story, medium grass
	Croplands, tall grasslands	3	4.6	Tall grass > 0.8 m
	Upland Scrub/Shrub	5	7.5	Brush < 0.8 m
	Oak brush/slash	6	14.7	Brush/Cured slash
	Upland Pines/Coast plain	8	0.3	Litter layer without under story
	Upland Pine-Oak	9	61.7	Dense pinelands /under story brush
	Riverine/Palustrine mixed wetland	10	1.0	Litter layer + under story
	Wetland forest	11	2.3	Light logging slash/Swamps
	Moderately & highly developed	28	0.3	Urban

22.2.3.2 HARVEST

The HARVEST model is primarily a landscape-level, harvesting allocation simulator designed to evaluate alternative strategies of forest management and timber harvests and provide comparable predictions of the spatial pattern consequences of these alternative strategies (Gustafson and Crow 1996). Three fuel maps were created for each landscape; a control (fuel map with no harvest), and two fuel maps representing "clustered" and "dispersed" harvests with a cutting level of 4% using the model (http://ncrs.fs.fed.us/4153/Harvest/harvhome.asp). The harvesting units were set with an average of 10 ha with a 5 ha standard deviation, while minimum and maximum units were 1 ha and 20 ha, respectively, without buffer area. For the simulation, the HARVEST model randomly selected a place to clearcut (clustered and dispersed shape) according to the patch size and age of a stand, where dispersed is more irregular shape with more edge. After forest was cleared, the fuel type assignment was simply changed from forest categories 8, 9, or 10 to harvested category 11 (logging slash/swamp).

22.2.3.3 FRAGSTATS

We used FRAGSTATS (McGarical and Marks, 1995), a spatial pattern analysis program, to quantify the landscape structures in the CNF and NJP landscapes. The characteristics of quantified landscape structures were then linked to fire spread across the landscapes.

22.3 Results

Substantial differences in fire-spread patterns between the two landscapes were observed when holding weather conditions constant and excluding roads in our simulations. The MBA of 24 fires after a 15-day burning period was 3,867 ha on the CNF versus 4,177 ha on the NJP using the same weather inputs. This 8% MBA difference was surprising given that the MROS differed by 83% between the two landscapes (183 vs.334 m/hr). Spatial variation in BA across the NJP (930 and 1,773 ha, respectively) was much larger than that in the CNF (795 ha). This strongly indicates that land cover fragmentation is substantially controlling fire spread at the landscape level (Finney 2000) because the topographic differences were minimal between the two landscapes and road effects were excluded. Four landscape-level indices clearly demonstrated that the CNF landscape was roughly twice as fragmented as the NJP landscape (Fig. 22.3).

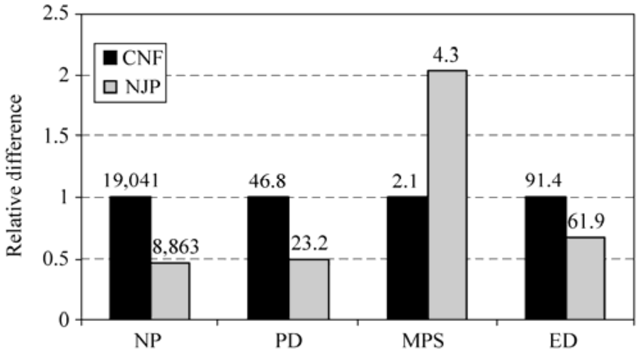

Figure 22.3 Relative differences ($VALUE_{NJP}$ / $VALUE_{CNF}$) of 4 landscape indices in the Chequamegon national forest (CNF) and New Jersey Pinelands (NJP). These were selected to illustrate the relationship between landscape fragmentation and fire spread. The values of indices in the CNF landscape are always expressed as 1. Absolute values are shown on top of each bar: NP = number of patches, PD = patch density (No./100 ha), MPS = mean patch size (ha), and ED = edge density (m/ha). Higher values of NP, PD, and ED or lower values of MPS indicate a higher degree of fragmentation for a given landscape

If the roads were considered as fire barriers during the simulations, the averaged BA in the CNF was 1,319 ha and 1,300 ha, respectively, for spring and summer periods without significant seasonal difference (Fig. 22.4). In the NJP, the averaged BAs were 1,503 ha and 2,515 ha for spring and summer periods, respectively.

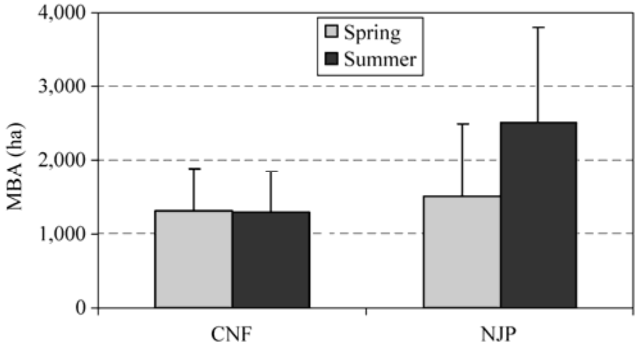

Figure 22.4 Comparison of seasonal changes of mean burned area (MBA, road effects were considered) of 24 fire ignition locations across the landscapes in the Chequamegon National Forest (CNF) and the New Jersey Pinelands (NJP). Vertical bars represent one standard deviation

Our simulations indicated that roads functioning as barriers could significantly reduce the BA of fire spread. In the CNF, road effects reduced the MBA by 71% and 66% for the spring and summer periods, respectively; compared to 25% and 30% for spring and summer, respectively, in the NJP. Much larger road effects in the CNF were partly due to the higher road density (0.63 km/km^2) than that in the NJP (0.33 km/km^2) (Table 22.2). In both landscapes, interacting with weather conditions further complicated the road effects on fire spread. The more favorite weather for fire spread, the larger road effects on reducing BAs are expected (Table 22.2).

Table 22.2 Seasonal weather variables and road density used for FARSITE model simulations of the 24 fires (considering roads as fire barriers) over a 15 day duration in the Chequamegon National Forest (CNF, WI) and New Jersey Pinelands (NJP). Model estimates of seasonal burned area are included. Numbers in the parentheses are the mean burned area (MBA) without considering road effects and the reduction in MBA by% if road effects were considered, separated by comma

	CNF		NJP	
	Spring 2004	Summer 2002	Spring 2004	Summer 2004
Mean temperature (℃)	4.6	18.1	10.4	23.1
Total precipitation (mm)	0	48	93	15
Mean wind speed (km/hr)	4.1	7.8	3.3	1.4
MBA (ha)	1,319 (4,561, 71)	1,300 (3,867, 66)	1,503 (2,015, 25)	2,515 (3,599, 30)
Road density (km/km^2)	0.63		0.33	

Harvesting effects showed different trends, depending on landscape fuel type composition and weather conditions. At 4% harvesting intensity, both clustered and dispersed methods showed no significant impact ($a = 0.01$) on reducing the mean burned area across the more fragmented CNF landscape, but showed significant effects on fire spread in the less fragmented NJP landscape in summer (Fig. 22.5) when weather condition (hot/dry) was more favorable to fire spread (Table 22.2). The clustered harvesting resulted in a slight reduction in MBA in NJ during the spring period. Relative changes in MBA caused by harvesting practices in a more fragmented landscape (CNF) were much smaller (< 2.5% in absolute value) than those in the NJP landscape with much less fragmentation and higher MROS (up to 17%, Fig. 22.5).

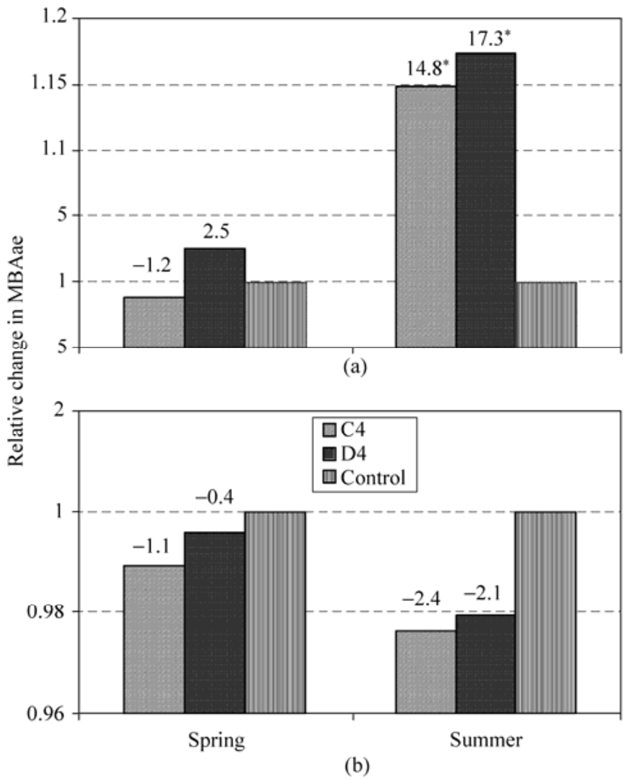

Figure 22.5 Effects of harvesting and harvesting methods on mean burned areas (MBA, relative change) in (a) New Jersey Pinelands; and (b) Chequamegon National Forest, WI; compared to the MBA in spring and summer for the control landscapes. Numbers above the bars indicate relative changes in %, compared to the MBA in control landscapes (scaled to 1). * Indicating that the difference in MBA is significant at level of 0.01

22.4 Discussion

Our simulations revealed that both landscape structure (e.g., spatial arrangement of patches and degree of fragmentation) and fuel type composition can significantly affect fire spread across the landscapes when weather is held constant. Our results clearly indicated that increases in both fragmentation and rate of spread generally increased the size of MBA. However, how these two variables interact across the landscape was the key to determine MBA. For example, the fuel-based MROS in the CNF was 183 m/hr, which resulted in a landscape mean burned area of 3,867 ha (Table 22.2). When we assigned the fuel types in the NJP with a similar MROS (173 m/hr) to that of the CNF, the mean burned area across the landscape was only 1,630 ha, 58% smaller than that in the CNF, reconfirming our previous conclusion that landscape fragmentation was another controlling factor for fire spread across the landscape. Quantified landscape indices showed that the degree of fragmentation in the CNF was, on average, about twice as that in the NJP (Fig. 22.3). If both the degree of fragmentation and fuels' MROS in landscape A is higher than those in landscape B, the difference in burned area should be enhanced between the two landscapes. If one factor is higher while the other is lower in one versus the other landscape, then the effects of the two factors on fire spread will be contradictory, hence moderating each another. Our results have demonstrated such combined effects. Because the NJP landscape was much less fragmented than that of the CNF landscape and the effect of increasing fuels' MROS overcame the effect of decreasing landscape fragmentation in the NJP (Fig. 22.3).

Spatial variation (measured by STD) of fire spread across the NJP landscape was higher than that in the CNF landscape. This is likely because there were 3 additional grass fuel categories (not presented in the CNF) in the NJP that have much faster fire-spread rates. If a fire reaches these fuel types it can greatly increase the size of its perimeter, even with the same weather.

While the combination of temperature and moisture conditions in the weather input was the most influencing environmental factor on fire spread across the landscape, wind speed can also enhance fire spread across the landscapes (Table 22.2). To better understand that relationship, we ran the FARSITE model in the control NJP and CNF landscapes by increasing wind speed to 40 km/hr in spring (when both landscapes had similar mean wind speeds (4.1 km/hr in the CNF vs. 3.3 km/hr in the NJP), Table 22.2) while keeping all other inputs as the same. Consequently, landscape mean fire-spread area increased by about 8 times in the NJP and 5 times in the CNF landscape, suggesting that an increase in wind speed has greater enhancement power on fire spread across landscape that possesses higher fuels' MROS. However, the simulated spread areas using 40 km/hr wind input should be interpreted with caution because in both landscapes it is rare to have such a high wind speed consistently over a 15 day period.

The actual fire-spread areas on the ground within the NJP landscape could be larger than our simulated results for two reasons. First, our simulations dealt with the surface fire only for comparison purposes because most fires in the oak-dominated CNF landscape are low-intensity surface fires (Sturtevant et al. 2004) while crown fires in the pine-dominated NJP landscape could be an influencing factor on fire spread. Second, Anderson's (1982) fuel classification system was primarily developed for the Western USA, thus, it may not quite match the unique characteristics of the vegetation and fuel types in the Eastern USA (http://www.fs.fed.us/ne/global/research/fire.html).

If roads are treated as fire barriers, our results as expected showed an inverse relationship between road density and mean area burned (Table 22.2). However, the degree of reduction in BA related to roads was not proportional to the change in road density between the two landscapes, nor had the same degree of reduction between seasons, even within the same landscape, indicating the road effects on BA are non-linear and complicated other interacting factors such as weather, landscape structure, and fuel characteristics. Furthermore, the road effects on fire spread could go in opposite directions (e.g., increasing fire spread) if human factors are considered. More roads often equates to increased area accessible by people, resulting in higher fire ignition frequency and, thus, increasing fire spread across landscapes (http://www.uecutah.org/where%20there%20is%20smoke.htm). For example, humans caused over 97% of all fire ignitions in the northern forests of the upper Midwest during the 1980's and 1990's (Cardille et al. 2001). Therefore, future studies on fire spread and behavior should incorporate human and social factors, especially in densely populated areas to improve our current understandings on the topic.

Seasonal changes in fire spread should be strongly associated with how energy and water conditions were combined. In the CNF, MBA only differed by 1.5% between spring and summer because the overall effects of whether conditions in spring (dry but cool) and in summer (warm but wet) on fire spread were contradictory to each other (Table 22.2). In the NJP it showed 67% difference in MBA between spring and summer because of sharp weather contrasts, with cool and wet springs vs. warm and dry summers (Table 22.2), thus enhancing the seasonal effects of weather on fire spread (Fig. 22.4). Our results suggested that differences in BAs between seasons should not be necessarily larger than the changes within the season across landscape. Such differences primarily depend on a combination of weather conditions, although climate can be used to differentiate the fire regimes in general. Spatial variations in BA for a given landscape seemed to be determined by landscape mosaic and fuel type composition but not by the variations in weather condition (Fig. 22.4).

The effects of harvesting (e.g., spatial arrangement, treatment method) on fire spread were complicated by interactions with landscape fragmentation, fuel type, and weather. In the CNF, both harvesting practices reduced burned area slightly and the cluster method showed more effects than the dispersed method did on fire spread. In contrast, harvesting practices tended to increase burned area and dispersed method showed more effects than clustered method did on fire spread in the NJP. Such influence was more evident and significant under weather conditions that are favorable to fire spread (Fig. 22.5). Following reasons could cause this inconsistency. When harvesting was conducted within the landscapes, it caused two changes simultaneously: ① the MROS was generally reduced in both landscapes because fuel type 11 (logging slash) had 24% lower fuel intensity than type 10 (occupying 50% in the CNF) and 20% lower than type 9 (occupying 62% in the NJP, Table 22.1), thus slowing fire spread; and ② both landscapes became more fragmented, which could increase fire spread. In the CNF, the effects of reducing MROS seemed to outweigh the effects of increasing fragmentation in an already highly fragmented landscape, resulting in less than expected corridor effects on increasing fire spread. Consequently, harvesting reduced the amount of area burned. In the NJP landscape where has been no harvests conducted in the past 100 years, the effects of increasing fragmentation across the landscape was larger than the effects of reducing MROS, thus, harvesting tended to increase fire spread. Furthermore, harvesting can cause more effects on fire spread under extreme weather conditions that are favorable to fire spread (Fig. 22.5).

Harvests can altered landscape structure in several ways such as fragmenting the fuel complex, creating open areas for easier fire spread, or disrupting local fire growth patterns (Stratton 2004). Some of these changes can be favorable or unfavorable for fire spread depending on the interactions with other contributing factors. In the CNF, our results suggested that the clustered method had greater effects on reducing fire spread than the dispersed method possibly due to the disruption of fuel connectivity in a already fragmented landscape. In the much less fragmented NJP landscape, however, dispersed method showed more effects on fire spread than the clustered method probably due to enhancing the corridor effects and improving the fire ventilation across a landscape having a much higher MROS. In both landscapes, harvesting practices showed more effects on fire spread in summer than in spring (Fig. 22.5). Evaluation of harvesting methods on BA was confounded by varying weather conditions throughout the year and interactions with other factors such as fuel type composition and landscape fragmentation; therefore, it deserves more studies in the future.

22.5 Conclusions

Burned areas in general were larger in the NJP landscape due to highly volatile fuels, compared to the CNF landscape when weather inputs were held constant. Our results indicated that fire spread was associated with landscape fragmentation. The combined effects of fuel-type composition and landscape fragmentation ultimately determined the differences in fire-spread areas and spatial patterns between the two landscapes if other controlling factors are considered constants.

In a more fragmented Oak-dominated landscape in northern Wisconsin, both clustered and dispersed harvesting methods with 4% cutting intensity could reduce burned areas up to 2.4%, compared to those in the control landscape. Although the clustered method reduced more areas both in spring and summer, such reductions were not significantly different ($a = 0.01$). In the less fragmented NJP landscape, 4% cutting could significantly increase burned areas compared to the control landscape, especially under extreme weather conditions that favor fire spread. Seasonal variation in fire-spread area is not necessarily larger than within-season variation in fire-spread area, depending on combinations of weather conditions during the period that fires occurred.

Our results showed that roads as physical barriers could significantly reduce fire spread across the landscape depending on existing road density. However, road effects on fire spread could have the opposite effect (e.g., increasing fire spread) if human factors (ignitions) are considered. Road effects can be enhanced when weather conditions are more favorable to fire spread.

This study suggests that effects of harvesting methods on fire spread are more complicated and vary with other fire controlling factors such as land-use history, fuel type composition and weather conditions. Thus, forest management planning should be flexible and aim to the characteristics of given landscape to minimize fire spread. As such, more studies on this from a landscape-oriented perspective are desired.

Acknowledgements

The joint fire science project (JFSP) and the Northern global change program (NGCP) primarily support this research. We are grateful to Kevin McCullough and Nicholas Skowronski in USDA forest service, newtown square, PA for providing us field data and GIS maps in the New Jersey Pinelands. We thank two anonymous reviewers for their valuable comments on the manuscript.

References

Albert DA, (1995), Regional landscape ecosystems of Michigan, Minnesota, and Wisconsin: a working map and classification. Gen. Tech. Rep. NC-178, USDA Forest Service North Central Forest Experiment Station, St. Paul, MN

Anderson HE, (1982), Aids to determining fuel models for estimating fire behavior. General Technical Report INT-GTR-122, USDA Forest Service

Bresee MK, Le Moine JM, Mather S, Brosofske KD, Chen J, Crow TR Rademacher J, (2004), Disturbance and landscape dynamics in the Chequamegon National Forest, Wisconsin, USA, from 1972 to 2001. *Landscape Ecology*, **19**: 291 − 309

Cardille JA, Ventura SJ, Turner MG, (2001), Environmental and social factors influencing wildfires in the upper midwest, United Stats. *Ecological Applications*, **11**: 111 − 127

Finney MA, (1998), FARSITE: fire area simulator—model development and evaluation. Res. Pap. RMRS-RP-4, USDA, For. Serv. Rocky Mt. Res. Station, Fort Collins, CO, USA

Finney MA, Andrews PL, (1999), FARSITE: Fire area simulator—a model for fire growth simulation. *Fire Management Notes*, **59**: 13 – 15

Finney MA, (2000), A spatial analysis of fire behavior associated with forest blowdown in the Boundary Waters Canoe Area, Minnesota Duluth, MN. USDA Superior National Forest http://www.superiornationalforest.org/ july4thstorm1999/bwcara/bwcawra.html

Graham RT, (2003), Influence of forest structure on wildfire behavior and the severity of its effects: an overview. USDA, Forest Service, North Central Experiment Station, 12

Gustafson EJ, Crow TR, (1996), Simulating the effects of alternative forest management strategies on landscape structure. *Journal of Environmental Management*, **46**: 77 – 94

Huff MH, Ottmar RD, Lehmkuhl JF, Hessburg PF, Everett RL, Alvarado E, Vihanek RH, (1995), Historical and current forest landscapes of eastern Oregon and Washington. Part II: potential fire behavior and smoke production. General Technical Report PNW-GTR-355, USDA Forest Service, Portland, OR

Hunt Jr. ER, Lavigne MB, Franklin SE, (1999), Factors controlling the decline of net primary production with stand age for balsam fir in Newfoundland assessed using an ecosystem simulation model. *Ecological Modelling*, **122**: 151 – 164

Keane RE, Arno SF, Brown JK, (1989), FIRESUM—an ecological process odel for fire succession in western conifer forests. General Technical Report INT-266, USDA Forest Service, Intermountain Research Station

Kercher JR, Axelrod MC, (1984), A process model of fire ecology and succession in a mixed-conifer forest. *Ecology*, **65**: 1725 – 1742

Lathrop R, Kaplan MB, (2004), New Jersey land use/land cover update: 2000-2001. New Jersey Department of Environmental Protection, 35

McCormick J, Jones L, (1973), The Pine Barrens: Vegetation Geography. New Jersey State Museum, 76

McGarical K, Marks BJ, (1995), FRAGSTATS: spatial pattern analysis program for quantifying landscape structure (version 2.0). USDA Forest Service, PNW-GTR-351, Portland, OR

Miller C, Urban DL, (1999), A model of surface fire, climate and forest patter in the Sierra Nevada, California. *Ecological Modeling*, **114**: 113 – 135

Noormets A, Chen J, LeMoine J, Rademacher J, (2004), Seasonal dynamics of ecosystem carbon fluxes in five managed Northern Wisconsin forests. *Global Change Biology*: Submitted

Pyne SJ, Andrews PL, Laven RD, (1996), Introduction to wildland fire. John Wiley & Sons, New York

Radeloff VC, Mladenodd DJ, He HS, Boyce MS, (1999), Forest landscape change in the northwestern Wisconsin Pine Barrens from pre-European settlement to the present. *Canadian J. of Forest Research*, **29**: 1649 – 1659

Rollins MG, Morgan P, Swetnam T, (2002), Landscape-scale controls over 20th century fire occurrence in two large Rocky Mountain (USA) wilderness areas. *Landscape Ecology*, **17**: 539 – 557

Rothermel RC, (1983), How to predict the spread and intensity of forest and range fires. Gen. Tech. Rep. INT-143, USDA Forest Service

Rouse JW, Haas RH, Schell JA, Deering DW, (1973), Monitoring vegetation systems in the Great Plains with ERTS. pp 48 − 62 in Third ERTS Symposium

Stratton RD, (2004), Assessing the effectiveness of landscape fule treatments on fire growth and behavior. *Journal of forestry*, **102**: 32 − 40

Sturtevant BR, Zollner PA, Gustafson EJ, Cleland DT, (2004), Human influence on the abundance and connectivity of high-risk fuels in mixed forests of northern Wisconsin, USA. *Landscape Ecology*, **19**: 235 − 253