State changes in Mediterranean ecosystems

An analysis of wildfire activity and vegetation changes in the San Francisco Bay area

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Bachelor thesis

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Summary

Wildfire activity is increasing due to climate change as are global temperatures and drought. Mediterranean ecosystems are highly prone to wildfire and when disturbance is large enough state shifts can occur including deforestation. Plant types have different responses to disturbance: Seeders perish but regenerate from seed banks while resprouters can survive and generate new shoots from dormant buds. In this thesis, transitions between different vegetation types based on these fire response strategies are analyzed in the San Francisco Bay area. SPSS analysis were conducted using vegetation data from 1940 and 2010 as well as data on large wildfires, altitude and local climate to determine possible correlations between these variables. Significantly more state transitions occurred in areas where wildfire was present compared to where it was not. Additionally, succession from shrubland to resprouter forest occurred less often under the influence of wildfire. In general the results show state transitions in mediterranean ecosystems to be complex processes affected by a large variety of factors. Mediterranean ecosystems are likely to respond to increases in wildfire activity and more vegetation change is expected to occur in the face of increasing climate change. Future research using more detailed vegetation and wildfire data over larger time scales will provide more insight into the dynamics of vegetation and wildfire.

Introduction

Increased wildfires activity, combined with global increases in temperature, drought and extreme weather conditions due to climate change raise questions about how ecosystems will respond to these changes (Stevens-Rumann et al., 2018; Intergovernmental Panel on Climate Change, 2019). Disturbance events like wildfires can lead to altered post-disturbance conditions which allow for the development of novel communities with different interactions between species and the environment (Donato, Harvey, & Turner, 2016). Under future climatic conditions, Mediterranean ecosystems could be especially vulnerable to these kind of state shifts (Baudena et al., 2020; Torres, Pérez, Quesada, Viedma, & Moreno, 2016). Besides the changes in temperature and drought, agricultural land abandonment and the resulting increases in fire hazard have been identified as one of the major drivers of land cover changes in Mediterranean ecosystems (Moreira et al., 2011).

Mediterranean ecosystems exist at several locations around the world including Europe, Chile, South Africa, Southern Australia and California (Esler, Jacobsen, & Pratt, 2018). These ecosystems provide a lot of beneficial goods and services such as wood, water regulation and erosion control (Fischlin et al., 2007) which are all at risk due to state shifts in ecosystems. Understanding the mechanisms behind the resilience of Mediterranean ecosystems is therefore critical for the conservation of biodiversity and ecosystem function in the face of global change (Spasojevic et al., 2016).

Non-anthropogenic deforestation in Mediterranean ecosystems is mainly driven by fire and drought (Karavani et al., 2018). While some research exists on these ecosystem state transitions (e.g..Buhk, Meyn, & Jentsch, 2006; Odion, Moritz, & DellaSala, 2010; Batllori, Ackerly, & Moritz, 2015; Karavani et al., 2018), a lot is still uncertain about this phenomenon. For instance to what extent these deforested states are maintained by fire and whether Mediterranean forests will be able to cope with future climatic change. Vegetation changes are hard to study as the ecological timeframe which is needed to study stability exceeds the available range of observations and systems that appear stable could actually be transient and maintained by disturbance (Fukami & Nakajima, 2011). In paleo-ecology however, changes in fire disturbance regimes have been linked to changes in vegetation cover using charcoal and pollen analysis (Colombaroli, Vannière, Emmanuel, Magny, & Tinner, 2008; Hallett & Walker, 2000).

Vegetation can have different ways of dealing with fire disturbance, the main responses being seeding and resprouting. Resprouters can regenerate from dormant buds after a fire while seeders can only recolonize from their seed banks (Pausas & Keeley, 2014). This thesis will focus on vegetation transitions between these different fire response types during the past century, specifically between the vegetation as documented in 1940 and that in 2010, in order to better understand the dynamics of vegetation and fire. The California bay area has been chosen as the research location, the open space areas (protected areas) within this region in particular. High fire occurrence rates cause the area to be prone to the previously described state changes. With regard to climate change, significant land cover change is predicted to occur here for temperature increases of more than 2 degrees (Hayhoe et al., 2004). These factors, combined with the recent increases of wildfire activity and the associated risks (Wells, O'Leary, Franklin, Michealsen, & McKinsey, 2004) make the California bay an excellent area for research on state transitions in Mediterranean ecosystems.

This paper seeks to contribute to a growing body of knowledge on state transitions in Mediterranean forests by answering the following questions: Are there state transitions regarding post fire response types of the vegetation cover in the California Bay area and if so, how are the changes in vegetation types related to wildfire activity? Additionally, how do altitude, slope and local climate affect the relation between wildfire activity and vegetation cover?

First an overview of the relevant concepts will be introduced in the theory section: post-fire recovery, the different aspects of fire regimes and their influence on vegetation, the concept of resilience and the theory on ecosystem state shifts, analyzed using a conceptual model. Finally, the impact of anthropogenic activity and of climate change on state shifts will be summarized.

Theory

Post fire strategies: resprouters and seeders

Plants can adopt different strategies with regard to post fire survival, the main ones being seeding and resprouting. A further subdivision can be made between obligate resprouters, obligate seeders, facultative seeders/resprouters and post-fire colonizers (Pausas & Keeley, 2014). Resprouting can be defined as “the ability to generate new shoots from dormant buds after stems have been fully scorched by fire” (Pausas & Keeley, 2014). The ability to resprout is recognized as a key functional trait and has been linked to the protection of buds mainly with use of bark or soil based on the bud location (Clarke et al., 2012).

Post-fire seeding can be described as “the ability to generate a fire-resistant seed bank with seeds that germinate profusely after fires” (Pausas & Keeley, 2014). Some species produce seeds that are able to germinate in response to heat (Serotiny), a rare but important trait among Mediterranean pine species (Gil, López, García-Mateos, & González-Doncel, 2009). Obligate seeders species have no means of resprouting and typically rely on their seed bank for post-fire re-establishment. Post-fire colonizers do not possess fire resistant seed banks and therefore rely on colonization from outside the burned area. Facultative seeders or facultative resprouters have both mechanisms of regenerating after a fire (Pausas & Keeley, 2014). Mediterranean forests are usually dominated by pine species (Pinus) which are obligate seeders or hardwoods, mainly oaks (Quercus) which are resprouters.

Fire regimes

Fire regimes can differ in several ways: frequency, intensity, fire type (crown or surface), seasonality and fire size (spatial extent). These parameters are largely determined by climate, vegetation, land use and topography (Morgan, Hardy, Swetnam, Rollins, & Long, 2001). Additionally, wind seems to be a major contributing factor to the fire size as is evidenced by the large southern California wildfires under the influence of Santa Ana windstorms (Keeley, Safford, Fotheringham, Franklin, & Moritz, 2009, p. 289). Lastly, disturbance from other sources also seems to have an effect on fire severity but not so much on the spatial extent of the fire in subalpine forest in Colorado (Kulakowski & Veblen, 2007, p. 764).

It is important to understand how different communities respond to different fire characteristics in order to accurately describe fire-vegetation dynamics (Buhk, Meyn, & Jentsch, 2006). High fire frequency can have a negative impact on pine species, the threshold being the point at which the fire return interval is shorter than the time it takes to regenerate the seed banks after a fire (Lloret & Vilá, 2003; Enright, Fontaine, Bowman, Bradstock, & Williams, 2015). High fire frequencies are generally linked to lower fire intensities compared to areas subjected to lower fire frequencies. This is because low fire frequencies allow for larger buildups of flammable biomass. Very high intensity fires can sometimes damage resprouters to the point at which they lose their regenerative capacity (Vlok & Yeaton, 2000), however, Mediterranean oak species are very resilient and will not easily succumb to fire (Baudena et al., 2020). Seeder species such as pines are less resilient to high intensity fires yet the seedbank of most species can even survive the high heat releases. This is the case for hard-seeded species but also for seeds of some opportunistic species (Buhk et al., 2006, p. 12).

When considering the type of fire, crown fires are usually of higher intensity than surface fires. Surface fires usually have little effect on tall trees, which leads to the main difference between these types of fires regarding post-fire conditions (Buhk et al., 2006).

The seasonality regarding fires also affects several of the already listed factors. Spring fires are mostly of lower intensity due to reduced moisture availability while summer fires can reach high intensity due to the accumulation of dry biomass during prolonged periods of little rainfall (Buhk et al., 2006).

Fire shape and size can influence the dispersal of species depending on the mechanisms used. Seeders without fire resistant seed banks have to rely on input from unburnt zones when recovering their population and are therefore prone to larger fires (Buhk et al., 2006; Pausas & Keeley, 2014).

Resilience of fire disturbed ecosystems

Resilience can be defined as the “magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behaviour” (Folke et al., 2002). When considering fire disturbance in ecosystems, the resilience of the system depends on how well the system can regenerate after a fire. This capacity to regenerate is partly determined by the individual resistance of the species inhabiting the system. The ability to resprout plays a big role as well traits influencing fire tolerance and resistance like fuel moisture content, bark thickness, leaf senescence and others (Karavani et al., 2018). Population level factors such as the ability to recolonize via seed banks, propagule establishment and survivability are also important with regard to resilience (Karavani et al., 2018). Research suggests that large scale functional diversity of regeneration traits (ability to resprout/seed, fire tolerance and resistance) is a stronger predictor for the recovery of system productivity after a fire than functional diversity of species richness or seed mass (Spasojevic et al., 2016).

State shifts in Mediterranean forests

When the resilience of a system is altered beyond regenerative capacity, state transitions can occur (Hosseini, Barker, & Ramirez-Marquez, 2016). Oak dominated forests are less prone to state shifts due to their resprouting ability while pine forests, which are seeder

species, are less resilient to state changes (Torres et al., 2016; Martín-Alcón & Coll, 2016). When pine forests, especially non-serotinous pines, are disturbed to the point at which adult trees die to fire and no seed sources are available, a state shift is likely to occur. The direction of change is then dependent on whether resprouters are present in the understory (Torres et al., 2016). If they are, resprouter dominance is generally the next state of the system, if not however, deforestation is most likely to occur leading to a state of dominance by Shrubs (fig, 1).

Availability of some nutrients such as nitrogen and phosphorus may be slightly increased for a short time after a fire while other nutrients might be less accessible due to the fire (Hinojosa, Parra, Laudicina, & Moreno, 2016). In general nutrient availability decreases, and this effect is further enhanced by the fact that burned soils are much more susceptible to erosion (Shakesby, 2011; Vieira, Fernández, Vega, & Keizer, 2015). Decreased nutrient availability may reinforce the transition towards non-forest states (Mayor et al., 2016). Resprouting forests can also be prone to deforestation, this occurs mainly under pressure of high fire activity in combination with drought and will be discussed later in this paper. When an ecosystem state shifts, the question is whether the alternative state can be considered as stable or not (Fukami & Nakajima, 2011).

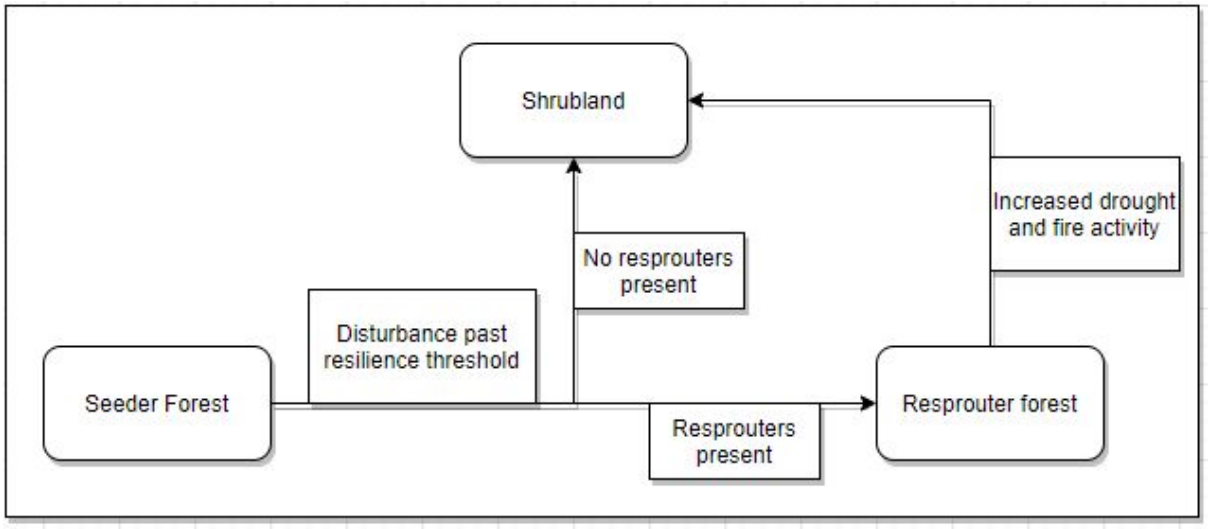

Figure 1: State shifts in Mediterranean ecosystems.

In order for a certain state to be stable, some form of negative feedback is necessary to recover from disturbances. In Mediterranean forests and shrublands such a fire vegetation feedback exists in the form of pyrogenicity: how well vegetation promotes fires and requires them for its sustainability. Shrublands are more pyrogenic while forests are more pyrophobic (Odion, et al., 2010), meaning shrublands are more prone to fire due to their low moisture content and high amount of flammable biomass which leads to higher fire frequency and intensity. In turn, this increased fire activity helps keep the shrubland in place by killing young trees, thus keeping succession at an early stage where shrubs can dominate (Santana, Baeza, Marrs, & Vallejo, 2010). The decrease of soil quality and fertility after a fire may reinforce this stabilizing feedback (Mayor et al., 2016). As opposed to shrubs, forests are less prone to fire, leading to lower fire activity which helps keep the late successional forest in place.

Human−fire interactions

Wildfires are often being managed by forest services or land owners. Prescribed burning and fire suppression are used to try to gain control of wildfire activity in an area. However, when small fires are being thoroughly suppressed in a certain area, this can lead to large buildups of dry biomass which can then instigate very large fires that are beyond extinction. In Catalonia only a fraction of the total fires is larger than 500 ha (0.2%) while they are responsible for 60% of the total burned area. This phenomenon, so-called the “fire paradox” leads to uneven fire size distributions in areas where fire suppression takes place (Piñol, Beven, & Viegas, 2005). Other research shows that population density and distance to WUI (the zone in which human development and vegetation connect) seem to lead to higher fire frequencies (Syphard et al., 2007). Additionally, forest roads seem to have a restricting effect on fire size (Narayanaraj & Wimberly, 2012).

Climate change and drought

Climate change is leading to higher temperatures and longer drought (IPCC, 2019). Mediterranean forests are especially susceptible to these changes according to the “intermediate-fire productivity” hypothesis (Pausas & Ribeiro, 2013). This hypothesis assumes a trade-off between aridity and productivity whereby areas with intermediate productivity and aridity are more prone to fires than areas with either high productivity and low aridity such as rain forests or low productivity and high aridity like deserts. High productivity means more fuel to burn but when conditions are moist, fire isn’t likely to occur. For areas with low productivity and high aridity, not enough fuel is present for high wildfire activity even though conditions are favourable for fire to occur.

Research in Spain suggests that during the last century, a shift in the fire limiting factor could have occurred (Pausas & Fernández-Muñoz, 2011). A large increase in fire frequency and size was observed around 1970 and attributed to an increase in fuel loads. Climatic conditions such as temperature and humidity appeared to be strongly related to post-1970 fire activity, thus suggesting a shift from fuel limitation to drought limitation which is expected to cause additional increases in fire activity in the face of further climatic change.

Drought affects species composition directly as an increase in water stress can lead to lower rates of growth, survival and recruitment. Even exhaustion of resprouting capacity has been reported in Mediterranean ecosystems (Pratt et al., 2013). A protection mechanism to counteract this exhaustion has been identified in *Quercus subpyrenaica* whereby premature leaf withering protects the stem from xylem cavitation (Peguero-Pina et al., 2015, p. 1919). Furthermore, drought weakens tree resistance to attacks by pests and pathogens which would also decrease resprouting capacity (Oliva, Stenlid, & Martínez-Vilalta, 2014). Increases in drought and fire combined can lead to serious alterations in Mediterranean ecosystems, promoting the occurrence of the state shifts discussed earlier (StevensRumann et al., 2018; Baudena et al., 2020; Batllori et al., 2018; Enright et al., 2015).

Methods

Study area

The study area for this report is the San Francisco bay area in California, specifically the southern parts of Napa and Sonoma county as well as Contra Costa county on the opposite side of the bay (see fig, 2). The area has a Mediterranean climate with dry and hot summers and mild and wet winters (Santos, Thorne, Christensen, & Frank, 2014). Its highly diverse in terms of vegetation types and significant land cover change is predicted to occur here for temperature increases of more than 2 degrees (Hayhoe et al., 2004). The area is sloped upwards slightly from the pacific ocean in the west towards the rocky mountains in the east.

Selection of plots to be studied

For this project, data about open spaces in the bay area as well as data on wildfires were made available by prof M. Santos (University of Zurich). Within the three counties mentioned above many open space areas have been acquired since the 1800's and preserved ever since (Santos et al., 2014a). In California, open space areas are generally protected from harmful human activities. However, some form of active management can be present at times. In order to minimize land use changes during this study's timeframe, only those open space areas that were already established before 1940 and are still open space now are considered for research. Based on this premise three areas were selected: Sugarloaf ridge state park, Lake Curry and Mount Diablo state park (see Figure 2).

Historic vegetation data in the bay area have been recorded during the decade 1930-1940 in the Wieslander Vegetation Type Mapping Project (Kelly, M., B. Allen-Diaz, and N. Kobzina., 2005) (hereafter: Wieslander data). Within each of the three parks, clusters of consecutive open space areas have been traced including Wieslander plots which represent the vegetation type categories relevant for this research: Seeder dominated forest, Resprouter dominated forest, Mixed forest and Shrubland. The Wieslander plots were very variable in size and for an adequate covering of each of these vegetation type categories in a number of cases several Wieslander plots were needed. Within the selected open space areas no Wieslander plots which were categorized as Seeder dominated forest were present, but Mixed Forests include Seeders as well as Resprouters.

Modern vegetation has been documented during the decade 2000-2010 in a map by the Bay area Upland Habitat Goals under the R5 CALVEG classification system (BAUGH, 2011) (hereafter: Calveg Data). The Calveg plots often are small and the number of Calveg plots per Wieslander plot frequently is significant and in such cases for feasibility reasons a selection of Calveg plots had to be made. However, the selection of Calveg plots should still allow an adequate comparison between 1940 and 2010. As a rule of thumb it was decided that the Calveg plots should cover about 50% of the Wieslander plot area. Wieslander plots as well as Calveg plots regularly stretch out outside the Open space areas, and in such cases their area within the Open space was leading.

A total of 27 Wieslander plots have been selected, see Figure 3. In this figure the red areas stand for hardwood forest, the green areas for mixed forest and the orange ones for shrubland. Within these Wieslander areas a total of 95 Calveg plots have been selected.

Data Collection

For the vegetation data used in this project two measurements are used. The first consists of historic land-cover recorded in the Wieslander data. The second consists of the modern land-cover recorded in the Calveg data. For categorising the dominating vegetation types on the Wieslander plots the WHR1-types are used. For the Calveg plots the SAV-cover types are used and in addition the SRM-types for detailing the composition of Shrubland are used (see Kelly et al. 2005).

Data on wildfire activity in the research area during the years 1950-2010 documented by the US forest service is retrieved from the databasin website: https://databasin.org/datasets/bf8db57ee6e0420c8ecce3c6395aceeb. Climate data including average annual measurements of minimum and maximum temperatures and precipitation during the periods 1950-1955 and 2000-2005 in the study areas is retrieved from https://cal-adapt.org.

Finally, altitude data with a spatial resolution of 90m DEM is collected from https://databasin.org/datasets/78ac54fabd594db5a39f6629514752c.

These data on open spaces, vegetation types, wildfire activity and altitude have been made visible in ArcGIS pro. The resulting database is used to gather the following data, captured in a form prepared for this purpose. The form with all data filled in is added as Appendix 1.

Per selected Wieslander plot:

-ID-number (VTM_ID) (visible in pop-up screen after clicking on the plot in ArcGIS)
-Area (in ha) (the same)
-Vegetation type category (WHR1_type) and added Latin names (the same)
-Altitude – first determined in ArcGIS at the beginning and the endpoint of the longest axis of the plot (Walt1, Walt2) and of a transverse axis (Walt3, Walt 4) and then taking the average of these four altitudes
-Lengths of both axes – by using the measurement tool in ArcGIS (Length12, Length34)
-Slope along the longest axis – calculated by Wslope12=100*(Walt2-Walt1)/Length12.
-Slope along the short axis - calculated by Wslope34=100*(Walt4-Walt3)/Length34.

Per selected Calveg plot:

-ID-number (ObjectID) (visible in pop-up screen after clicking on the plot in ArcGIS)
-Area (in ha) – by clicking on the plot and using the measurement tool in ArcGIS
-Cover type category (SAF Cover type, for Chaparral specified by SRN-type)
-Occurrence of wildfire by activating these data and clicking on the plot in ArcGIS

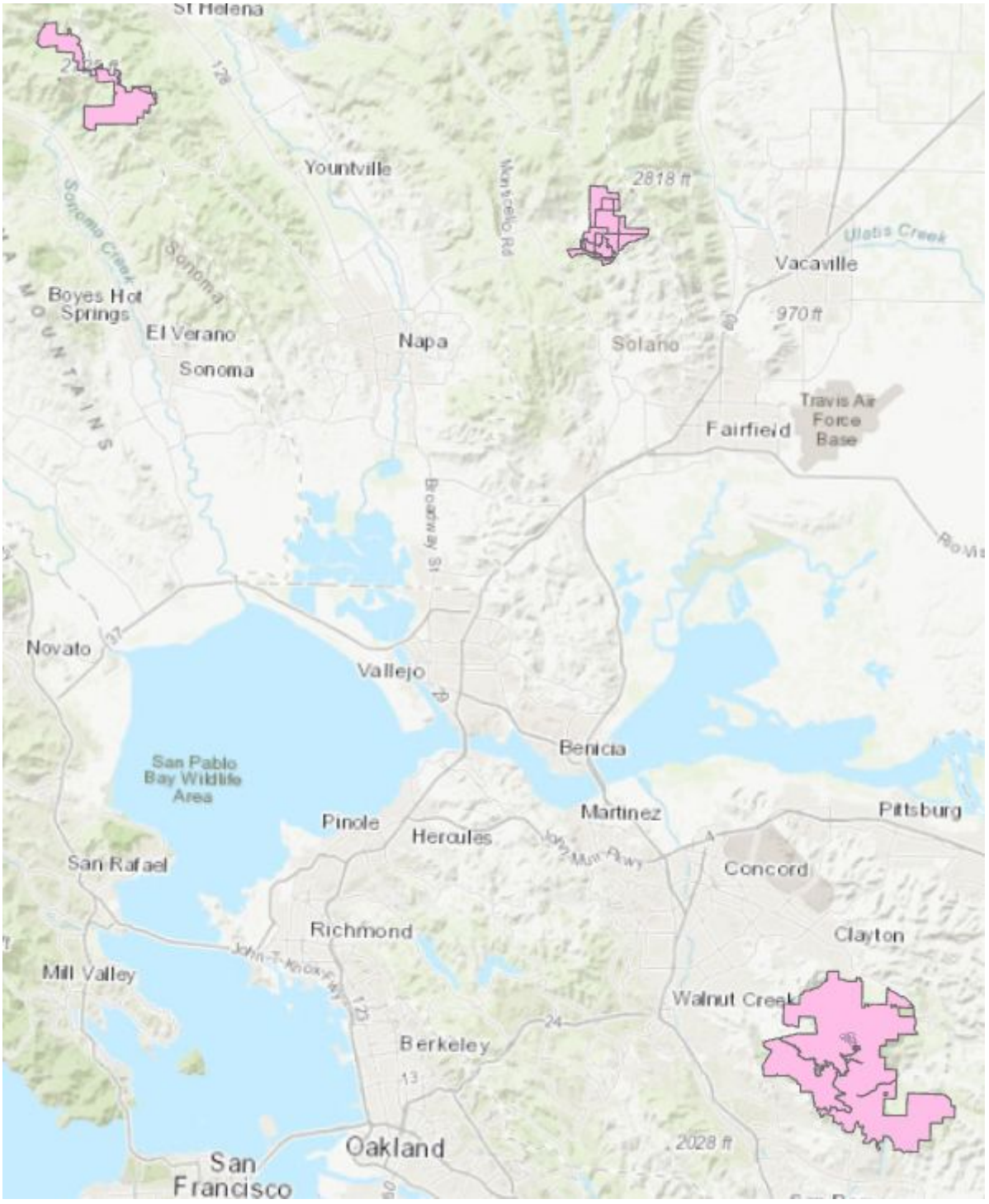

Figure 2: The study area and the selected open space areas established before 1940

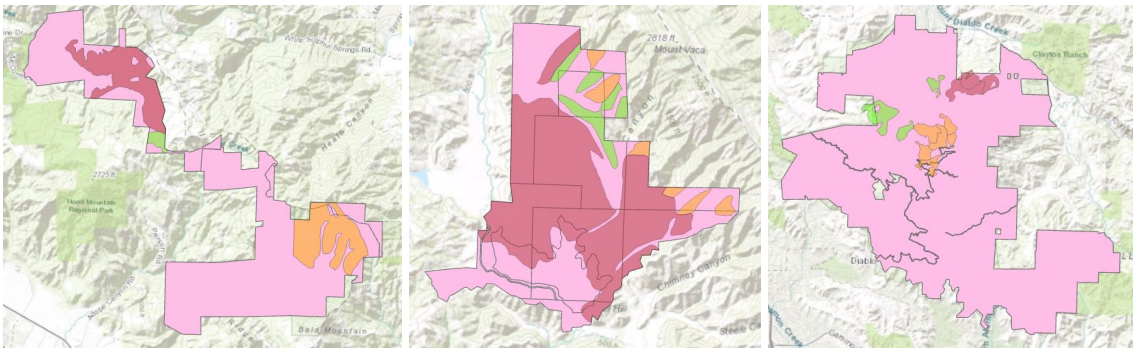

Figure 3: Selected study areas and their Wieslander plots from left to right: Sugarloaf ridge, Lake Curry and Mount Diablo. In the Wieslander data: red = Hardwood, green = Pines and orange = Shrubland. For scale see figure 2.

Data analysis

Since this thesis mainly focuses on the influence of wildfires on vegetation, the encountered vegetation types within the selected open space areas were placed in one of four categories based on fire response mechanisms: seeder dominated forest, resprouter dominated forest, shrubland and other (see Appendix 1). In the available vegetation data, the vegetation type classification used in 1930-1940 during the Wieslander project differs from the one used in the 2000-2010 data. For comparison between the two datasets to be made possible, both classifications have to be converted to the four fire response categories described above.

The Calveg plots will be analysed for changes in vegetation type composition between the 1940 and the 2010 measurements. Wildfire activity, area/altitude/slope and climate will be investigated for relationship to the possible changes in vegetation types. Univariate descriptive analyses are conducted for frequencies, ranges, totals and averages. Bivariate descriptive analyses are conducted using crosstabs and plots. Tests for statistical significance are conducted for the occurrence of specific vegetation type category transitions (crosstabs) and for the relationship between these transitions and wildfire activity (t-test) with Pearson chi^2 tests, supported by other tests. Differences between the three Parks in the occurrence of transitions, and differences between the vegetation type categories in the altitude and slope of the plots, are analysed and tested for significance with univariate analysis of variance (one way Anova). All the analyses mentioned so far are conducted with IBM SPSS version 25.

The possible predictors for vegetation type category transition are spread out over different aggregation levels, and relations among predictors could vary within higher aggregation units as well, which could negatively impact the accuracy of regular regression and covariance analysis. Therefore, additionally the data are analysed according to an explicit multilevel model with multilevel analysis, using the program MLwiN (Goldstein et al., 1998). The response variable (occurrence of transition) is binary (0/1). Binary variables are not normally distributed but binomially. However, with the numbers of cases (95 for analysing the occurrence of 'no transition' and from 26 to 35 for analyzing the main transitions) and the chances involved (varying from 24% to 54%), the binomial distribution more or less agrees with the normal distribution, which is the default assumption within MLwiN.

Results

Vegetation types and plot areas

Table 1 shows per park per fire response category in 1940 the number of the selected plots and their total area in hectares in both the Wieslander data (1940) and the Calveg data (2010).

*Table 1: Number of plots and their total area (in ha) per park per fire response category in 1940 *)*

Plots	Park	Seeder forest (not found)	Resprouter forest	Mixed forest	Shrubland	Other (not selected)
Wieslander	Sugarloaf		1 (162 ha)	1 (13 ha)	1 (164 ha)	
	Lake Curry		1 (600 ha)	5 (84 ha)	6 (50 ha)	
	Mount Diablo		3 (119 ha)	3 (107 ha)	6 (172 ha)	
Calveg	Sugarloaf		10 (52 ha)	2 (11 ha)	10 (72 ha)	
	Lake Curry		14 (149 ha)	16 (50 ha)	6 (20 ha)	
	Mount Diablo		10 (64 ha)	8 (63 ha)	19 (106 ha)	

*) Wieslander plots (1940) were selected within the open space areas of the three parks, Calveg plots (2010) were selected within these Wieslander plots as described in the Method section.

Figure 4 shows the two areas (of Wieslander plots and Calveg plots, in SPSS named Wsurface and Csurface, respectivly) plotted together. Each point represents one Calveg plot.

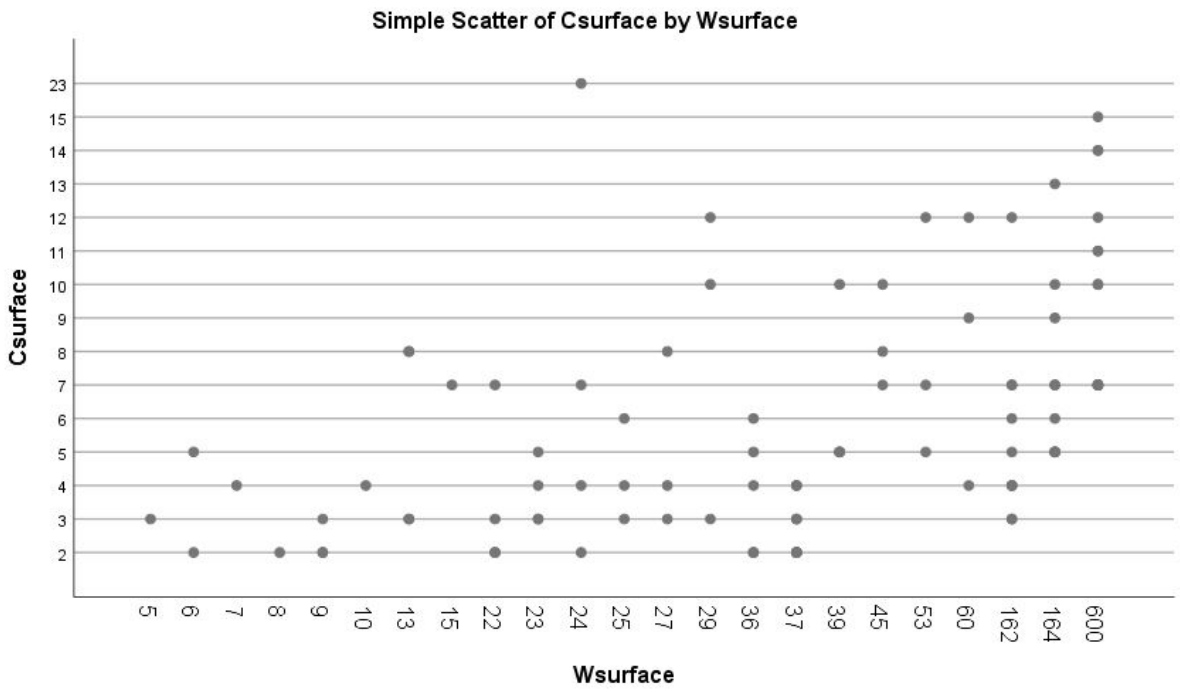

Figure 4: Areas of Wieslander plots (Wsurface) and Calveg plots (Csurface) in relation to each other. Each point represents one Calveg plot.

Transitions between vegetation types

Table 2 shows the transitions between the fire response type categories between the 1940 Wieslander data and the 2010 Calveg data.

Table 2: Transitions between fire response categories between 1940 and 2010 in both number of plots and amount of hectares

Wieslander plots 1940		Fire response categories on Calveg plots 2010				
Vegetation type category		*1 Seeder forest*	*2 Resprouter forest*	*3 Mixed forest*	*4 Shrubland **	*5 Other*
1 Seeder forest	(not found)					
2 Resprouter forest	2 (880 ha)	1 (5 ha)	**12 (95 ha)**	9 (55 ha)	2 (13 ha)	10 (86 ha)
3 Mixed forest	9 (204 ha)	3 (23 ha)	14 (40 ha)	**4 (18 ha)**	3 (10 ha)	2 (8 ha)
4 Shrubland	13 (386 ha)	3 (11 ha)	19 (113 ha)	5 (14 ha)	**7 (55 ha)**	1 (12 ha)
5 Other	(not selected)					
		6 (39 ha)	45 (248 ha)	18 (87 ha)	13 (78 ha)	13 (106 ha)

*All Shrubland in the Calveg data showed to consist of resprouters (see Appendix 1)

It is assumed that the method for selecting Calveg plots within the Wieslander plots (focussed on decent coverage by number and spread) leads to a representative sample.

From the table about two thirds of the area of the Wieslander plots appear to have made a transition to a different fire response type based on the Calveg data. The most common transition (in both number of plots and total area) is that of shrubland to resprouter forest.

The spread of the three vegetation type categories from the Wieslander data over the five cover types found in the Calveg data is shown in Table 3, with observed and expected values.

Table 3 Crosstabs of fire response categories in 2010 and vegetation type categories in 1940
The categories 1 through 5 for Calveg type and Wieslander type correspond with those in Table 2

			Ccovtype					
			1	2	3	4	5	Total
Wvegtype	2	Count	1	12	9	2	10	34
		Expected Count	2,5	16,1	6,4	4,3	4,7	34,0
	3	Count	3	14	4	3	2	26
		Expected Count	1,9	12,3	4,9	3,3	3,6	26,0
	4	Count	3	19	5	7	1	35
		Expected Count	2,6	16,6	6,6	4,4	4,8	35,0
Total		Count	7	45	18	12	13	95
		Expected Count	7,0	45,0	18,0	12,0	13,0	95,0

,04	Value	df	Asymptotic Significance (2-sided)
Pearson Chi-Square	17,390 [a]	8	,026
N of Valid Cases	95		

a. 10 cells (66,7%) have expected count less than 5.

The Pearson Chi-Square test is problematic here because of the many cells with expected value < 5. However, other tests confirm that the relationship shown in Table 3 is significant: Cramer's' V =.303, p =.026; Contingency coefficient = .393, p =.026; Lambda (symmetric) = .118, p = .016; Goodman and Kruskal tau (Calveg type dependent) = .043, p =.038.

Comparing the expected and observed values in Table 3, it shows that Resprouter Forest less often than expected did not transform and more often than expected transformed into Mixed Forest or Other, and Shrubland more often than expected did not transform or transformed into Resprouter Forest. The following situations occurred the most:

- No transitions between cover types (23 plots in total).
- A transition from Resprouter Forest to Mixed Forest (9 plots) or Other (10 plots).
- A transition from Shrubland to Resprouter forest (19 plots) or to Mixed Forest (14 plots).

The amount of these transitions that took place differs between the three study areas. In two situations the difference between the parks is statistically significant:

- In Sugarloaf State Park (Sonoma County) 50% of the plots retained the same cover type and in Lake Curry and Mount Diablo State Park only 16-17% did so (F (2,92)=5,64, p < .01).
- The transition from Mixed Forest to Resprouter Forest occurs mostly in Lake Curry 36% (13 plots), in Sugarloaf Park 0% and in Mount Diablo Park only 3 % (1 plot) (F (2,92)=13,18, p <.001.

Wildfires and state transitions

In about 80% of the Calveg plots, only one wildfire has occurred between 1950 and 2010 (in 1973 in Lake Curry, 1977 in Mount Diablo Park and 1994 in Sugerloaf Park). This happened on 30 of the 34 Calveg plots in which the vegetation in the corresponding Wieslander plots was categorized in 1940 as Resprouter Forest (88%), 24 of the 26 plots categorized in 1940 as Mixed Forest (92%) and 25 of the 35 plots categorized in 1940 as Shrubland (71%).

Looking at the Calveg data 12 of the 34 Resprouter Forest plots recorded in the Wieslander data are still classified as Resprouter Forest in the Calveg data (36%). For Mixed Forest 15% stayed the same and for Shrubland this was 20% (see Table 2). Assuming that during the period between 1940 and 2010 no multiple transitions between vegetation cover types have taken place, Resprouter Forest has been maintained most often.

The occurrence of wildfire is only moderately related to vegetation type changes. See table 4, including five 2x2 Crosstabs including significance tests.The number of plots on which a transition from Shrubland to Resprouter Forest occurred was proportionally lower in case of Wildfire (p=.001). The number of plots on which no transition has occurred was also just proportionally lower in case of Wildfire (p (Chi^2)=.045, (p (Fischer Exact)=.051).

Table 4 Incidence of wildfire and occurrence of transitions between fire response types, in number of Calveg plots (2010)

	Fire response category									
Wildfire	Same cover type		Resprouter Forest > Other		Mixed Forest > Resprouter Forest		Shrubland > Resprouter Forest		Resprouter Forest > Mixed Forest	
	0	*1*	*0*	*1*	*0*	*1*	*0*	*1*	*0*	*1*
0 (no)	9	7	15	1	16	0	8	8	16	0
1 (yes)	63	16	70	9	65	14	68	11	70	9
Chi ^{2}	4.003		0.374		3.326		10.823		2.014	
p Chi ^{2}	p=.045		p=.541		p=.068		p=.001		p=.156	
p FE *)	p=.051		p=.468		p=.061		p=.003		p=.175	

*) FE = Fischer Exact (1-sided

Altitudes and slopes

The Wieslander plots differ not only in area but also in altitude and slope. The average altitude of the Wieslander plots varies from 220 up to 1040 meters above sea level. The slope over the longest axis varies from -39% to 35% and that of the shorter axis varies from -20% to 54%. The mean altitude and the ranges in altitude and slope vary between the parks (See table 5).

Table 5 Variation in the average height (m) and in slopes (%) of the Wieslander plots per park.

Park	Number of Calveg plots	Mean of the average altitudes of the Wieslander plots	Range of the average altitude	Range of slope, longest axis	Range of slope, transverse axis
Sugarloaf	22	573	510 - 725	-5% to 10%	-12% to 6%
Lake Curry	36	288	220 - 415	-4% to 29%	-14% to 27%
Mount Diablo	37	737	415 - 1040	-30% to 35%	-29% to 54%

The slope over the longest axis is significantly correlated to the average height ($r=.450$, $p<.001$). For the shorter axis however, this is not the case ($r=.002$, $p=.98$). There is also no correlation between the long and short axis slopes ($r=.12$, $p=.24$).

Altitudes, slopes and vegetation types

The mean altitude of the Wieslander plots and the slopes along the long axis differ significantly between the three Wieslander fire response categories (see table 6). Seeder Forest and Shrubland occur more often on higher and more sloped surfaces.

Table 6 Height and slope according to vegetation type in 1940 and 2010

Vegetation type category 1940	*Number of Calveg plots*	*Altitude (m)*	*Slope (%)*
Resprouter Forest	34	414	4
Mixed Forest	26	415	6
Shrubland	35	724	17
Univariate Analyses of Variance		$F(2,92)=30,8$ $p<.001$	$F(2,92)=8,1$ $p=.001$

Vegetation category 2010	*Number of Calveg plots*	*Altitude (m)*	*Slope (%)*
Seeder Forest	7	730	24
Resprouter Forest	45	482	8
Mixed Forest	18	483	9

Shrubland	12	713	11
Other (Herbaceous)	13	474	6
Univariate Analyses of Variance		F (4,90)=4,4 p=.003	F (4,90)=2,2 p=.081

Area, altitude and state transitions

Analysis of possible relations between the altitudes and slopes and the occurrence of transitions between vegetation types is hard because slopes can differ locally. The altitude of the Calveg plots has been approximated using the mean altitude of the Wieslander areas. Table 6 shows the relations between the area of the Calveg plots, the average height of the corresponding Wieslander plots and the degree of vegetation cover change regarding the most common transitions (see Table 4). The transition from Shrubland to Resprouter forest is more common in higher areas while transitions between Mixed Forest and Resprouter Forest is more common in lower areas.

*Table 6: Area and approximated altitude of Calveg plots and incidence of vegetation type transition **

		Fire response category				
Calveg plot feature	The fire response category no or yes	Same vegetation type category	Transition of Resprouter forest to Other	Transition of Mixed forest to Resprouter forest	Transition of Shrubland to Resprouter forest	Transition of Resprouter forest to Mixed forest
Area (ha)	0 (no)	5,7	***5,8***	***6,7***	6,3	6,1
	1 (yes)	7,4	***8,8***	***2,9***	5,4	6,1
Average altitude (m) **	0 (no)	524	538	***563***	***498***	***548***
	1 (yes)	541	453	***330***	***653***	***344***

*) **Bold** = statistical significant (p<.05) according to an independent t-test
**) Represented by the average altitude of the corresponding Wieslander plot (see Method, Data Collection)

Climate features

Table 7: Historical observation data on average annual minimum and maximum temperature and precipitation for the three parks.

	Average annual minimum temperature (Fahrenheit)		*Average annual maximum temperature (Fahrenheit)*		*Average annual Precipitation (Inches)*	
Park	1950-1955	2000-2005	1950-1955	2000-2005	1950-1955	2000-2005
Lake Curry	40.3	43.5	68.7	71	30.2	35.1
Sugarloaf	39	42.8	67.4	69	45	48.1

Mount Diablo	46.6	50.3	70.1	73.2	24.5	23.9

The table shows a clear difference in temperature and precipitation between Mount Diablo and the other areas as well as a difference in precipitation between Sugarloaf and Lake Curry. The temperature doesn't differ significantly between Lake Curry and Sugarloaf. A clear increase in temperature can be seen in the 2000-2005 observations compared to those in 1950-1955. Precipitation seems to have increased in Sugarloaf and Lake Curry but not in Mount Diablo.

An integrated multilevel model for further analysis

In the data used, three different aggregation levels can be identified: the parks, the Wieslander plots and the Calveg plots. The different variables that have been included in this analysis can have an effect on fire response type transitions (the response variable) at different levels of aggregation: Climate features could differ between parks, altitude mainly differs between Wieslander plots and wildfire activity differs between the Calveg plots.
These interactions are summarized in the model seen in figure 5.

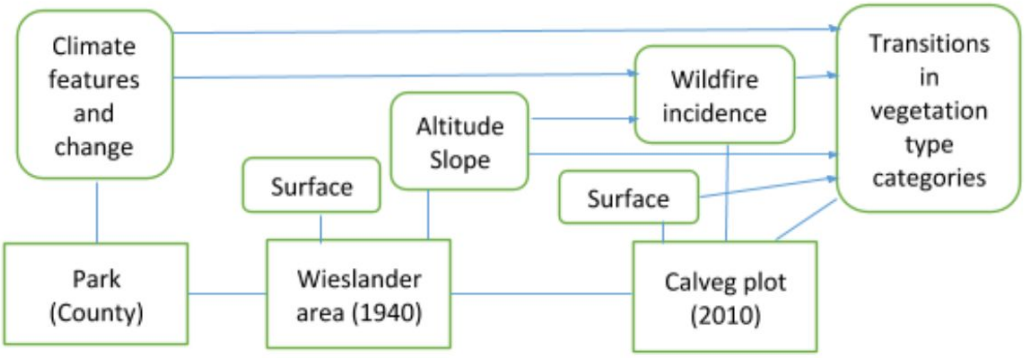

Figure 5: An integrated multilevel model with three aggregation levels and predictors on each level

To get a first feeling of the nature and significance of multilevel analyses in this research, a two-level model was analysed, with 'non-occurrence of transition' between 1940 and 2010 as response variable (Samecat), the Calveg plot area (Csurface) and the incidence of wildfire (Wildfire) as possible level-1 explanatory variable and the average altitude of Wieslander plot (WmeanAlt) as a possible level-2 explanatory variable.
The parameter estimates are shown below, together with their standard errors between brackets (the coefficients are significant at $p < .05$ if they exceed 1,96 times their standard error).

Samecat = 0,27 (0,21) (intercept) + 0,0 (0,0) WmeanAlt + 0,018 (0,012) Csurface
− 0,257 (0,142) Wildfire

This result shows that 'non-occurrence of transition' in vegetation type category can be a little bit predicted (or 'statistically explained') by the non-occurrence of wildfire and the surface of the Calveg plot, the latter not significantly. The mean altitude of the Wieslander plot does not contribute anything.

Adding the two slopes as possible explanatory variables, the estimates change a little and the coefficients of Csurface ($p < .10$) and Wildfire ($p < .05$) become significant, but the slopes themselves do not contribute anything:

Samecat = 0,35 (0,19) (intercept) + 0,0 (0,0) WmeanAlt + 0,021 (0,012) Csurface
− 0,283 (0,136) Wildfire + 0,005 (0,004) slope12 – 0,005 (0,003) slope34

These preliminary analyses have been done using the complete data file of 95 cases (Calveg plots). However, keeping the same category A between 1940 and 2010 or undergoing a state transition from A to B are only possible on plots who have been categorised in 1940 as covered by vegetation category A. Therefore the data file had to be splitted in three parts: Wieslander plots categorized as Resprouter forest, as Mixed forest, and as Shrubland, respectively (compare Table 2). In this way, keeping the same category can be analysed in a more pure manner. In addition, next to the relationships between the possible predictors shown in the model in Figure 5 and the non-occurrence of transition, their relationships to the four most common transitions (Resprouter to Mixed forest, Resprouter forest to Other, Mixed to Resprouter forest, and Shrubland to Resprouter forest) have to be analysed as well.

Each of these seven response variables (three for non-transition and four for the different transitions) were analysed using three consecutive models (see the Method section under *Data analysis*): first the variance components on the three levels (Null model), then adding the possible predictors on the first level (the Calveg plots), and after this adding the predictors at the second level (the Wieslander plots). The outcomes are given in Appendix 2.

These analyses on the three parts of the data sets separately (on the Calveg plots within Wieslander plots which in 1940 were categorised as Resprouter forest, Mixed forest and Shrubland, respectively) show that the explanatory value of the area of the Calveg plots (Csurface) and the incidence of wildfire for the non-occurrence of transition, which came forward from the preliminary analysis on the whole data set (see above), rests on the continuation of Shrubland for Csurface and on the continuation of Mixed Forest for non-occurrence of Wildfire.

In addition, the outcomes now show also the following. Analysing the second model (variance components with the predictors at level 1 added) Csurface to a certain extent predicts the transition from Resprouter to Mixed forest (the coefficient is negative, so this transition occurred more on smaller Calveg plots), and the incidence of Wildfire the transition from Shrubland to Resprouter forest (the coefficient is positive).

The analyses of the third model (with also the predictors at level 2 added) confirm the predictive value of the non-occurrence of wildfire for the continuation of Mixed forest (the coefficient of Csurface for the continuation of Shrubland being non-significant now), and the predictive value of the Calveg plot area for the transition of Resprouter to Mixed forest (negative, this occurred more on smaller Calveg plots), and now also significantly for the transition of Resprouter forest to Other (positive, this occurred more on larger Calveg plots).

Analysing the third model on the separate parts of the data set has also resulted in some significant coefficients for the two slopes of the Wieslander plots. However, these outcomes do not deserve much attention, as they are small and due to accidental variable values in small data sets.

The variances of the response variables at level 3 appeared to be small, particularly then possible predictors were added at level 1, and after adding predictors at level 2 no

variance at level 3 was left at all, so no possible predictors at level 3 were added anymore (as nothing was left to explain).

For reasons of comparison, the three models (see Appendix 2) have been additionally analysed on the whole data set (n=95), with Samecat and the four main transitions, as response variables. The outcomes of these analyses confirm the results described above: the non-occurrence of wildfire predicts the non-occurrence of state transition, the area of the Calveg plots predicts the transition from Resprouter to Mixed forest (negative: more transition on smaller plots) and the transition from RF to OTHER (positive: more transition on bigger plots), and the area and the two slopes of the Wieslander plots have no predictive value.

Discussion

This final section is concerned with the interpretation of the findings with the help of scientific literature as well as discussing the limitations of the research and implications for future research.

Interpretation of the results

The main questions this thesis set out to answer were: are there state transitions regarding post fire response types of the vegetation cover in the California Bay area and if so, how are the changes in vegetation types related to wildfire activity? Additionally, how do altitude, slope and local climate affect the relation between wildfire activity and vegetation cover?

The performed analysis indicate state changes in several directions have occurred and some significant relationships between variables were found. A notable amount of vegetation change in varying directions has been documented between the 1940 and the 2010 data: a distinct proportion of the mixed forest has changed to resprouter forest (about 54%); regular succession has been detected in in about 80% of the shrubland area, a large part of which manifested as resprouter forest; deforestation occurred as well but only in very small quantities (0.08% of resprouter forest and 1.2% of mixed forest). Based on just these findings, transitions from seeder towards resprouter forest seem to occur relatively often. This aligns with the theory which describes oaks to be more resistant to state changes than pines (Torres et al., 2016; Martín-Alcón & Coll, 2016).

Regarding Wildfires, the absence of state transitions was found to be a significantly predicted by wildfire absence. Succession from shrubland into resprouter forest occurred significantly less often when wildfire was present. This can be explained by the self sustaining feedback active in shrublands as described in the theory. Succession was significantly more common in higher areas while transitions between Mixed Forest and Resprouter Forest (in both directions) were significantly more common in lower areas. There is no readily available explanation for this however, research has shown topography to be related to climate and wildfire activity (Dillon et al., 2011). Climate varied between the parks as well as the different points in time. Conducting any analysis on its influence proved to be difficult due to the way the data is structured. It can be assumed however that variations in temperature and precipitation affect plant growth and survival and therefore resilience to state changes as well (Reyer et al., 2012).

Limitations of the research

Possible conclusions to be drawn from this study are limited in some respects. First of all, the Wieslander and Calveg data sets differ in terms of measurement accuracy using measurement by eye and by remote sensing respectively. The Wieslander plots are also significantly larger than those in the Calveg data quite possibly as a result of this difference in measurement techniques. For this reasons it's possible the Wieslander data has for example listed a certain area as Mixed forest where a more accurate satellite measurement would have listed several patches of pine and oak forest instead.

Additionally, on 79 of the 95 Calveg plots only one fire occurred based on the available data. On the rest of the plots no fire occurred. This meant the occurrence of wildfire was used as a yes or no variable, there either was a fire between 1940 and 2010 or there wasn't. As a result, the effect of fire frequency and seasonality could not be included in the analysis. These effects could nevertheless have been present considering smaller fires might very well have occurred in the research area which were not documented by the US forest service.

Thirdly, state transitions are processes which take place over a relatively long time period (Fukami & Nakajima, 2011). Nevertheless, within a period of 70 years (between 1940 and 2010) more or other changes could have taken place. There is, however, no sure way of knowing if documented state transitions are indeed related to fire regimes and climate change. Extensive documentation of human intervention within the study's time frame would be needed in order to confirm no other factors influenced the vegetation during this time. A time series analysis would be a more effective way of analyzing state transitions. This would however require more than two measurements for vegetation types. In spite of these limitations, the analysis conducted in this thesis have produced valuable results which show the complexity of the different factors involved in state transitions in mediterranean ecosystems.

Implications of the research

In order to more fully understand vegetation and wildfire dynamics, more consistent measuring of vegetation and fire is required. Nowadays remote sensing data can be used to periodically measure vegetation in large areas using different vegetation indexes (Xue & Su, 2017). Especially in the face of further climatic change this will be a valuable asset in understanding more about the natural systems and how these are affected by disturbances. Remote sensing can also be used for wildfire detection and technological advances with regard to sensor systems like hyperspectral cameras, image intensifiers and thermal cameras allow for better documentation compared to the 20th century (Allison, Johnston, Craig, & Jennings, 2016). For future research on this topic the use of more detailed wildfire data as well as more data points for the vegetation data will provide more opportunities for research. Time series analysis including wildfire parameters like frequency and seasonality can lead to a better understanding of vegetation and wildfire dynamics which will help

advance our knowledge on how to protect mediterranean ecosystems in the face of future climate change.

Conclusion

The main purpose of this research was to gain a better understanding of wildfire and vegetation dynamics in mediterranean ecosystems, specifically those in the San Francisco Bay area. SPSS analysis were conducted using vegetation data from 1940 and 2010 as well as data on large wildfires, altitude and local climate the determine possible correlations between these variables. Significantly more state transitions occurred in areas where wildfire was present compared to where it was not. Additionally, succession from shrubland to resprouter forest occurred less often under the influence of wildfire. Some correlations were detected regarding topography as well. The transition from Shrubland to Resprouter forest was significantly more common in higher areas while transitions between Mixed Forest and Resprouter Forest (in both directions) were significantly more common in lower areas.

This study has provided useful insight into the dynamics of vegetation and wildfire in mediterranean ecosystems by examining the impacts of multiple factors on the occurrence of state transitions in the San Francisco Bay area. These type of transitions remain poorly understood and hard to study considering the large time scale at which they take place. In the future however, more detailed research will be made possible by using current and upcoming technology. It has become clear that more research into the different factors affecting mediterranean vegetation and wildfires will help advance our knowledge on the response of these systems to future climate change.

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Appendix 1 Data vegetation type categories

VTM_ID refers to Wieslander plots and ObjectID to Calveg plots.
Areas are in ha as far as within the Park Open space areas and the selected Wieslander plots.
Altitudes and lengths: the altitudes of both endpoints and lengths of the about longest axis (L) and a (perpendicular on it) short axis (S) of the Wieslander plots (which can have very different forms).
Modern cover type: 1 CON 2 HDW 3 MIX 4 SHB 5 Rest (herbaceous, barren, water, agriculture, urban).

Sonoma County - Sugarloaf Ridge State Park

Montane Hardwood

VTM_ID	Ha	Altitudes and lengths (in m)		Description (WHR1_type)	Type
243993	162	370 - 660 (3000) 480 - 530 (1000)		**Montane Hardwood** (white Oak) (Quercus garryana, Quercus kelloggii)	2
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
70874	11,5			California coast live oak	2
72192	5,5			California coast live oak	2
49197	3,1		1994	Herbaceous	5
71063	4,0			California coast live oak	2
76854	3,1		1994	California coast live oak	2
72107	6,7		1994	California coast black oak	2
63034	3,6		1994	California coast black oak	2
69387	4,0		1994	California coast live oak	2
49143	7,2		1994	Herbaceous	5
51392	3,1			Herbaceous	5
	52				

Douglas Fir

VTM_ID	Ha	Altitudes and lengths (in m)		Description (WHR1_type)	Type
130743	13.0	680 – 750 (680) 750 – 710 (330)		**Douglas Fir** (Pseudotsuga menziesii menziesii Arbutus menziesii, Quercus kelloggii)	3
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
60880	8,3			Mixed conifer and hardwood forest/woodland	3
60575	3,1			Mixed conifer and hardwood forest/woodland	3
	11,4				

Chaparral

VTM_ID	Ha	Altitudes and lengths (in m)		Description (WHR1_type)	Type
243828	164	660 – 570 (2000) 550 – 640 (1500)		Chamise-Redshank **Chaparral** (Adenostoma fasciculatum, Heteromeles arbutifolia, Arctostaphylos stanfordiana)	4
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
77074	9,9			California coast live oak	2
74587	5,7			California coast live oak	2
72255	5,1			California coast live oak	2
72771	4,8			California coast live oak	2
72239	4,9			California coast live oak	2
58171	13,3			Mixed Chaparral [SRM-type: Shrub oak mixed Chaparral: Quercus berbiridifolia of White oak)	4
67479	8,5			California coast live oak	2
70908	7,4			California coast live oak	2
58175	6,9			Hard Chaparral [SRM-type: Shrub oak mixed Chaparral: Quercus berbiridifolia of White oak)	4
72850	5,4			California coast live oak	2

	72				

Napa County - Lake Curry

Blue Oak Woodland

VTM_ID	Ha	Altitudes and lengths (m)		Description (WHR1_type)	Type
243857	±600	200 – 240 (3800) 140 – 300 (2400)		**Blue Oak Woodland** (Quercus douglasii,, Quercus lobate,Quercus wislizeni, Avena Barbata, Bromus Hordeaceus)	2
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
94812	14		1973 partly	Not forest or woodland (herbaceous rangeland)	5
99524	14		1973	California coast live oak	2
98405	12		1973	California coast live oak	2
94832	11		1973 partly	Not forest or woodland (herbaceous rangeland)	5
94807	15		1973 partly	Not forest or woodland (herbaceous rangeland)	5
97660	11		1973	California coast live oak	2
98093	10		1973	California coast live oak	2
99380	10		1973	California coast live oak	2
96020	7		1973	Blue oak- digger pine	3
96893	7		1973	Blue oak- digger pine	3
96698	7		1973	Blue oak- digger pine	3
95706	7		1973	Blue oak- digger pine	3
96018	7		1973	Blue oak- digger pine	3
96275	7		1973	Blue oak- digger pine	3
	149				

Blue Oak - Foothill Pine

VTM_ID	Ha	Altitudes and lengths (m)		Description (WHR1_type)	Type
131823 (1)	10	270 – 400 (800) 300- 300 (160)		(Quercus douglasii & Q wislizeni, Pinus sabiniana)	3
131822 (2)	9	350 – 380 (620) 320 – 380 (230)		(Quercus douglasii & Q wislizeni, Pinus sabiniana)	3
130509 (3)	37	300 – 230 (1600) 320 – 380 (300)		(Quercus douglasii & Q wislizeni, Pinus sabiniana)	3
131324 (4)	6	320 – 430 (400) 390 – 360 (210)		(Quercus douglasii & Q wislizeni, Pinus sabiniana)	3
	62				
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
99379 (1)	4		1973	California coast live oak	2
98430 (2)	3		1973	California coast live oak	2
98994 (2)	2		1973	California coast live oak	2
98794 (3)	3		1973	California coast live oak	2
98444 (3)	2		1973	California coast live oak	2
98116 (3)	4		1973	California coast live oak	2
94954 (3)	4		1973	Herbaceous	5
99540 (3)	3		1973	California coast live oak	2
98797 (3)	2		1973	California coast live oak	2
98470 (3)	2		1973	California coast live oak	2
99277 (3)	2		1973	California coast live oak	2
98087 (4)	5		1973	California coast live oak	2
	36				

Montane Hardwood

VTM_ID	Ha	Altitudes and lengths (m)		Description (WHR1_type)	Type
131941	22	280 – 420 (1250) 290 – 300 (250)		**Montane Hardwood** (Quercus wislizeni, Pinus sabiniana)	3
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
95562	7		1973	Knobkone Pine	1
100120	2		1973	California coast live oak	2
97118	2		1973	Blue oak – digger pine	3
100025	3		1973	California coast live oak	2
	14				

Mixed Chaparral *

VTM_ID	Ha	Altitudes and lengths (m)		Description (WHR1_type)	Type
131873	6	350 – 470 (420) 380 – 460 (300)		Quercus berberidifolia Quercus wislizeni frutescens	4
131884	15	340 – 430 (750) 370 – 380 (260)		Quercus wislizeni frutescens Quercus berberidifolia	4
131097	7	270 – 310 (440) 350 – 310 (320)		Quercus wislizeni frutescens Quercus chrysolepis nana Ceanothus cuneatus	4
131883	5	270 – 320 (390) 270 – 330 (240)		Quercus wislizeni frutescens Quercus berberidifolia	4
131882	9	280 – 430 (720) 300 – 340 (170)		Quercus wislizeni frutescens Quercus berberidifolia	4
131879	8	330 – 410 (400) 360 – 430 (300)		Quercus wislizeni frutescens	4
	50				
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
96526	2		1973	Blue Oak –digger pine	3
99086	7		1973	California coast live oak	2
95962	4		1973	Blue Oak – digger pine	3
99083	3		1973	California coast live oak	2
99856	2		1973	California coast live oak	2
98974	2		1973	California coast live oak	2
	20				

* In each Wieslander plot (all of them being rather small) only one Calveg plot has been selected

Contra Costa County – Mount Diablo State Park

Blue Oak Woodland

VTM_ID	Ha	Altitudes and lengths (m)	Description (WHR1_type)	Type
124927 (1)	53	500 – 680 (1600) 450 – 590 (580)	Quercus douglasii	2
125133 (2)	39	450 – 680 (870) 470 – 680 (640)	Quercus douglasii, Quercus wislizeni	2
243388 (3)	27	840- 510 (850) 680 – 590 (760)	Quercus douglasii, Quercus wislizeni Cercocarpus betuloides, Grass sp.	2
	119			

ObjectID	Ha		Wildfires	Description (SAF Cover type)	Type
36614 (1)	12		1977 (partly)	Not forest or woodland (Herbaceous rangeland)	5
36681 (1)	7		1977	Not forest or woodland (Herbaceous rangeland	5
38348 (1)	5		1977	Blue oak – digger pine	3
37725 (2)	5		1977	Knobcone pine	1
38765 (2)	5		1977	Blue oak - digger pine	3
37437 (2)	5		1977	Hard Chaparral [SRM-type: North Coastal Shrub: Diverse flowering evergreen drought tolerant plants]	4
36641 (2)	10		1977	Not forest or woodland (Herbaceous rangeland)	5
37623 (3)	8			Hard Chaparral [SRM-type: Chamise Chaparral]	4
36737 (3)	4			Not forest or woodland (Herbaceous rangeland)	5
39175 (3)	3			Blue oak - digger pine	3
	64				

Blue Oak - Foothill Pine

VTM_ID	Ha	Altitudes and lengths (m)	Description (WHR1_type)	Type
243385 (1)	± 60	580 – 800 (1400) 650 – 360 (1000)	Quercus douglasii, Pinus sabiniana, Cercocarpus betuloides	3
125110 (2)	24	810 – 630 (910) 660 – 800 (430)	Quercus douglasii, Pinus sabiniana	3
125145 (3)	23	430 – 520 (830) 430 – 280 (720)	Pinus sabiniana, Quercus douglasii, Quercus wislizeni	3
	107			

ObjectID	Ha		Wildfires	Description (SAF Cover type)	Type
37435 (1)	4		1977	Knobcone pine	1
37424 (1)	9		1977	Hard Chaparral [SRM-type: North Coastal Shrub: Diverse flowering evergreen drought tolerant plants]	4
37694 (1)	12		1977	Knobcone pine	1
37619 (2)	23		1977	Hard Chaparral [SRM-type: Chamise Chaparral: Ademostoma fasciriculatum]	4
39863 (3)	5		1977	Blue oak – digger pine	3
37592 (3)	3		1977	Hard Chaparral [SRM-type: North Coastal Shrub: Diverse flowering evergreen drought tolerant plants]	4
37880 (3)	3		1977	California black oak	2
36657 (3)	4		1977	Not forest or woodland (Herbeceous rangeland)	5
	63				

Mixed Chaparral

VTM_ID	Ha	Altitudes and lengths (m)		Description (WHR1_type)	Type
125214 (1)	45	640 – 940 (880) 830 – 760 (440)		Quercus wislizeni frutescens, Quercus berberidifolia, Ptelea crenulata	4
135234 (2)	36	760 – 1090 (930) 940 – 1020 (360)		Quercus wislizeni frutescens, Cercocarpus betuloides, Quercus berberidifolia	4
125233 (3)	24	710 – 950 (920) 850 – 820 (380)		Quercus wislizeni frutescens, Cercocarpus betuloides, Ptelea crenulata, Quercus berberidifolia	4
125232 (4)	13	950 – 1140 (570) 920 – 1140 (410)		Quercus wislizeni frutescens, Cercocarpus betuloides, Ceanothus cuneatus	4
125239 (5)	25	820 – 920 (700) 920 – 950 (460)		Quercus wislizeni frutescens, Cercocarpus betuloides, Umbellularia californica	4
125019 (6)	29	850 – 1080 (960) 920 – 830 (850)		**Chamise Redshank Chaparral** Adenostoma fasciculatum, Ceanothus cuneatus	4
	172				
ObjectID	**Ha**		**Wildfires**	**Description (SAF Cover type)**	**Type**
42901 (1)	8		1977	California coast live oak	2
37574 (1)	7		1977	Hard Chaparral [SRM-type: Chamise Redshank Chaparral: Ademostoma spersifolium]	4
37659 (1)	10		1977	Hard Chaparral [SRM-type: Shrub oak mixed Chaparral: Quercus berbiridifolia of White oak]	4
37660 (2)	5		1977	Hard Chaparral [SRM-type: Shrub oak mixed Chaparral: Quercus berbiridifolia of White oak]	4
42076 (2)	2		1977	California coast live oak	2
37741 (2)	4		1977	Knobcone pine	1
39199 (2)	2		1977	Blue oak – digger pine	3
43079 (2)	6		1977	California coast live oak	2
38042 (3)	4		1977	California black oak	2
37767 (3)	7		1977	California black oak	2
39640 (3)	2		1977	Blue oak - digger pine	3
37747 (4)	3		1977	Knobcone pine	1
37746 (4)	8		1977	Knobcone pine	1
23355 (5)	6		1977	Canyon live oak	2
38561 (5)	4		1977	Canyon live oak	2
38848 (5)	3		1977	Blue oak - digger pine	3
37572 (6)	3		1977	Hard Chaparral [SRM-type: Chamise Redshank Chaparral: Ademostoma spersifolium]	4
37656 (6)	10		1977	Hard Chaparral [SRM-type: Shrub oak mixed Chaparral: Ademostoma spersifolium]	4
36981 (6)	12		1977	Not forest or woodland (Herbaceous rangeland)	5
	106				

Appendix 2 Results of multilevel analyses

Estimates of variances, intercepts and predictor coefficients (Standard errors between brackets)
*Bold coefficients are statistical significant (p < .05) (> 1.96 * Standard error)*

Consecutive models for analysis			Same vegetation type category in 1940 and 2010			Vegetation type state	
			RF > RF	MF > MF	SHR > SHR	RF > MF	RF > OTH
Variance components model							
	Intercept		.352 (.161)	.357 (.230)	.200 (.068)	.253 (.104)	.294 (.078
	Total variance		.235	.221	.160	.194	.208
	Var at level 3		.063 (.064)	.141 (.129)	.000 (.000)	.016 (.027)	.000 (.000
	Var at level 2		.000 (.000)	.001 (.017)	.000 (.000)	.000 (.000)	.000 (.000
	Var at level 1		.172 (.044)	.079 (.026)	.160 (.038)	.178 (.045)	.208 (.050
Model with predictors at level 1		*Predictor*					
	Intercept		.293 (.335)	1.027 (.198)	–.210 (.198)	.991 (.293)	–.008 (.08
		Csurface	*.027 (.024)*	*–.005 (.012)*	***.057 (.022)***	***–.090 (.020)***	*.036 (.023)*
		Wildfire	*–.161 (.251)*	***–.921 (.195)***	*.123 (.146)*	*–.105 (.207)*	*.040 (.241)*
	Total variance		.220	.070	.135	.189	.196
	Var at level 3		,056 (.057)	.000 (.000)	.000 (.000)	.000 (.000)	.004 (.018
	Var at level 2		.000 (.000)	.000 (.000)	.000 (.000)	.083 (.066)	.000 (.000
	Var at level 1		.164 (.042)	.070 (.019)	.135 (.032)	.106 (.028)	.192 (.049
Model with predictors levels 1+2		*Predictor*					
	Intercept		.179 (.462)	1.491 (.675)	.055 (.300)	1.908 (.0364)	*–.906 (.49*
		Csurface	*.036 (.025)*	*.009 (.023)*	*.039 (.023)*	***–.101 (.020)***	***.068 (.027***
		Wildfire	*–.008 (.253)*	***–1.107 (353)***	*–.343 (.312)*	*–.217 (.199)*	*.210 (.272)*
		WmeanAlt	*.000 (.001)*	*–.001 (.001)*	*–.000 (.000)*	*–.002 (.000)*	*.001 (.001)*
		Wslope12	***.031 (.009)***	*–.003 (.010)*	*.015 (.010)*	***–.028 (.007)***	*.006 (.010)*
		Wslope34	***–.044 (.013)***	*–.005 (.006)*	*–.006 (.004)*	*.037 (.010)*	*–.008 (.01*
	Total variance		.147	.066	.119	.092	.170
	Var at level 3		.000 (.000)	.000 (.000)	.000 (.000)	.001 (.000)	.000 (.000
	Var at level 2		.000 (.000)	.000 (.000)	.000 (.000)	.000 (.000)	.000 (.000
	Var at level 1		.147 (.036)	.066 (.018)	.119 (.029)	.091 (.022)	.170 (.041