Chapter One

Introduction

1.1 Introduction

The term 'Land-Use/Cover Change' (LUCC) acknowledges the interaction of human (e.g. cognitive, social, economic) and biophysical (e.g. ecological, hydrological, atmospheric) processes and systems driving landscape change. It is a term that implicitly considers human-influenced landscapes as integrated socio-ecological systems. The terms 'Land-Use' and 'Land-Use Change' (LUC) consider human use of land and changes in the purpose for which that land is used. For example, land uses include 'agricultural' and 'recreational'. 'Land-Cover' and 'Land-Cover Change' (LCC) refer to the physical, biotic cover of the land, regardless of the human use (or lack thereof), and any changes occurring to it. For instance, land cover might be defined as *Pinus* forest or *Quercus* shrub-land. Because human activities and land-use often directly determine land cover, change in the former often results in change in the latter. However, changes in land cover as a result of biophysical processes may equally have consequences that in turn influence human land-use. This reciprocal change frequently demands that LUCC research crosses traditional academic subject boundaries, utilising theoretical and practical tools from multiple fields and requiring collaboration between 'experts' from different backgrounds, in a manner that has come to be known as 'interdisciplinary' (e.g. Naveh 2000). A primary concern of this thesis is how LUCC models can and should be developed, when an interdisciplinary approach is required to address a problem.

1.2 Mediterranean Landscapes

Prime examples of socio-ecological systems are found in the Mediterranean Basin where human activity has extended over millennia. The Mediterranean Basin is one of the most biologically diverse regions on the planet, with around 10% of all plant species on Earth in an area covering just 1.5% of the Earth's land surface (Blondel and Aronson 1999). High levels of habitat diversity and human activity, when combined with the heterogeneous nature of the Mediterranean climate and soil conditions produce a biological spatial 'patchiness' not found in more temperate, arid or tropical regions of the world (Blondel and Aronson 1999). Biophysical and human processes promote, and are shaped by, this biological patchiness, producing spatially heterogeneous, mosaic-like landscapes of land covers. The highly seasonally variable characteristics of the Mediterranean climate also make landscapes in these environments amongst the most fire prone in the world (Smith 1992). Wildfire disturbance, and subsequent patterns of vegetation regeneration that in turn influence future wildfire activity, typify the feedbacks between pattern and process emphasised by landscape ecologists (Turner 1989). When humans are present in a landscape they become part of this pattern/process feedback – human patterns of land-use influence where wildfires occur and how they spread. In turn, wildfire influences patterns of vegetation and its regeneration, and thus human land-use. Naveh (1994 p.164) identifies this very close interaction between humans, vegetation and fire, noting that fire has had vitally important roles in the evolution of both floral and cultural landscapes over previous millennia, serving "as a major driving force in the co-evolution of Mediterranean humans and their landscapes". The close relationship between humans and fire is highlighted by the fact that, while in certain regions of the world (e.g. boreal regions) wildfires are predominantly ignited by natural causes such as lightning, in the Mediterranean Basin humans are the principle cause (Vazquez and Moreno 1993).

Regardless of their interaction with wildfire, traditional human land-use alone has been an important disturbance (either directly or indirectly) of Mediterranean vegetation. For example, tree crops are cut for management purposes and land is widely used to graze livestock. A common human land-use in Spain is the open, savanna-like landscapes of managed oak woodlands known as *dehesa*. Grazing livestock on grasslands between trees in these woodlands prevents growth of saplings and has been shown to reduce the number and quality of acorns available to initiate future establishment, due to predation (Leiva and Fernandez-Ales 2003, Cierjacks and Hensen 2004). The maintenance of fields exclusively for cereal cropland prevents any succession-type processes to occur, reducing species diversity and above-ground biomass that otherwise might occur. However, recent technological and socio-economic changes in the Mediterranean Basin have led to increasing land-use change, and notably the abandonment of agricultural and managed land. Such abandonment may lead to increases in vegetation biomass – the fuel of wildfires. That is, changes in social processes are influencing spatial biophysical landscape patterns. These patterns will in turn affect biophysical processes (such as wildfire and vegetation dynamics) in the reciprocal manner described above. These reciprocal links between landscape patterns and processes (across the traditional boundaries of social and ecological study), non-linear feedbacks and historical landscape contingency mean potential changes are unclear, with a wide range of possible future outcomes. This thesis sets out to examine the potential consequences of these feedbacks over the coming decades for a Spanish landscape.

1.3 LANDSCAPE MODELLING

Naveh (2001) coined the term 'multifunctional landscape' to describe a system manifested by the interaction between cognitive, cultural, economic, and biophysical processes and phenomena. These systems of interaction will be termed here as 'socio-ecological' systems, of which 'landscapes' are a physical manifestation. The nature of the processes occurring in socio-ecological systems means that they function at scales on the order of the human observer. Thus the 'landscape scale' is taken here to be on the order of $1 \times 10^2 – 1 \times 10^9$ m^2 (100 m^2 – 1,000 km^2) and $1 \times 10^4 – 1 \times 10^9$ sec (1 day – 100 years). The 'multifunctional' nature of Spanish landscapes outlined above suggests an interdisciplinary approach must be taken to study the dynamics of these systems. Geography has traditionally been the academic discipline to address the processes and scales associated with socio-ecological systems and landscapes. More recently, landscape ecology has emerged as an interdisciplinary field of study as ecologists came to terms with the idea that there are few ecosystems remaining that are uninfluenced by humans and their activities (see Naveh and Liberman 1994). Geography considers questions of space, place and scale. Landscape ecology addresses ecological problems and questions of ecological structure and function, often incorporating human influence, but also stresses the importance of space, scale, and feedbacks between spatial pattern and ecological process. These disciplines provide the theoretical and academic background to this thesis. However, one of the primary aims of this thesis (see section 1.4 below) is to improve understanding of the impacts of agricultural and other LUCC on ecological patterns and processes, specifically wildfire regimes. Thus, whilst human activities (and changes in them) are important drivers of LUCC in these landscapes the focus of study here is on the ecological consequences of these changes.

At any point in time the state of heterogeneous and multifunctional landscapes, such as the study area examined here, is a result of spatio-temporal interactions and contingencies. These landscapes are complex in the sense that "the cause of a difficulty may lie far back in time from the symptoms, or in a completely different and remote part of the system" (Forrester 1969 p8). Space matters as much as time, patterns matter as much as process, social dynamics matter as much as ecological dynamics. How, then, should these landscapes be investigated and understood? Generally some sort of simplification, a model, of the actual system is required to investigate the system. Spatially-explicit modelling is a useful tool to examine the processes and phenomena occurring at the landscape scale as they provide a means to overcome logistical, political and financial constraints of empirical experimentation. Particularly, when a problem is not analytically tractable (i.e. closed form equations cannot be written down and are non-integrable) simulation models may be used to represent a system by making certain approximations and idealisations (Winsberg 1999). Spatially-explicit simulation models of LUCC have been used since the 1970s and have dramatically increased in use recently with the growth in computing power available. This thesis capitalises on previous LUCC simulation and contemporary technology to develop an integrated cellular-automata/agent-based model of vegetation dynamics, wildfire regimes and agricultural decision-making.

As implied above, by definition a model is a simplified representation, or theory, of how real world systems function. However, when the system being modelled contains many heterogeneous interacting components, the model itself can become complex (*sensu* Forrester 1969 above). In this case, analysing, interpreting and explaining both model dynamics and output becomes challenging. Furthermore, the epistemological issues raised by attempting to put boundaries on these systems so that they can be represented *in silico*, means deduction and falsification become less useful methods by which to assess the knowledge gained. As models of socio-ecological systems represent actual people in the real world, opportunities are present to engage with the subjects of the model to develop, assess and improve it. However, engaging non-modellers with models raises issues of expertise and the 'public understanding of science'. This thesis investigates not only the processes and routes of change in a Mediterranean landscape, but also the processes and routes of investigation themselves given the potential for model engagement with those being modelled.

1.4 Thesis Aims

This thesis has the following aims:

i) to examine interactions between human land-use/cover change and wildfire regimes in a Mediterranean landscape

ii) to explore and evaluate novel methods to 'validate' simulation models (and processes of modelling) of environmental change considering human activity

These aims will be achieve via the following objectives. To achieve aim i) a spatially-explicit computer simulation model will be developed to examine:

1. the processes of land-use/cover change in a traditional Mediterranean landscape
2. the reciprocal influences of land-use/cover configuration and composition on wildfire regimes
3. the reciprocal influences of human population (size and 'type' of inhabitant) on land-use/cover and wildfire regimes

To achieve aim ii) the thesis will:

1. consider the epistemological and sociological issues confronting attempts to model socio-ecological systems
2. examine the possibility of using local stakeholder input to 'validate' the construction of the model
3. reflect on the experience of developing a socio-economic simulation model to examine the interaction of LUCC and wildfire

1.5 Thesis Structure

The structure of this thesis may be broken into three sections (Table 1.1). The first section (chapters two and three) reviews the context of both the subject of study and the means of studying it. This necessary prelude to any modelling project examines the key features of the systems under study that will need to be incorporated into a model, and reflects on how previous modelling research has approached similar systems. Chapter two introduces and outlines the key features of the history and contemporary condition of Mediterranean landscapes that are pertinent for this work. Namely, the importance of human activity as a component of ecological processes in the landscape and the importance of wildfire and spatial heterogeneity are discussed. The study area, EU Special Protection Area number 56 (SPA 56) in central Spain to the west of the city of

Madrid, is introduced and described in detail. Chapter three then considers the broad array of tools and techniques that have been developed to investigate and model LUCC in many regions around the globe. These include transition matrix models, regression-based models, spatially explicit landscape simulation models, agent-based models and integrated ecological-economic simulation models. Empirical models are used to examine the data available for SPA 56, and result in the finding that observed changes suggest that increased scrub land area is likely to occur in the future. However, these empirical models are found to be inadequate to represent the spatial pattern-process feedbacks present in the study area, and a simulation modelling approach is suggested as being more useful. The components of this modelling are presented in section two of the thesis.

Table 1.1 Thesis structure. The thesis has three main parts – part one is a review of the subject of study and the means to study it, part two concerns the construction and results of the simulation model, and part three is an examination of the appropriate methods by which to assess models of socio-ecological systems.

Research Questions	*Chapter*	*Subject*
	ONE	Introduction
What are the important issues here?	TWO	Study Area
How should this LUCC be modelled?	THREE	Review of Modelling Methods
How should this model be built?	FOUR	Landscape Fire Succession Model
	FIVE	Agent-Based Model of LUCC
What does this model tell us?	SIX	Integrated Model Results
How to 'validate' socio-ecological models?	SEVEN	Model Validation Discussion
Do others think this model is any good?	EIGHT	Stakeholder Evaluation
What have we learnt?	NINE	Summary and Conclusions

The second section of the thesis thus describes the construction of, and initial results from, an integrated simulation model of LUCC and wildfire. The Landscape Fire Succession Model (LFSM) is presented first as this provides the basis of the ecological processes of land-cover change, upon which the Agent-Based Model (ABM) of LUCC imposes the effects of human activity (i.e. land use) on land-cover patterns. The relationship with wildfire is reciprocal however, as burning subsequently influences land-use decision-making because of changes in the spatial configuration of land-use/cover. Chapter four reviews previous models of Mediterranean vegetation dynamics and disturbance (i.e. wildfire) and uses this as the basis to describe and justify the structure of the LFSM. The literature review in this chapter highlights the fact that large spatial and temporal extents, combined with the limited available data for these environments, demand trade-offs between computer processing power and the level of detail at which the system can be represented. Thus, the LFSM developed here adopts a grid-based spatially-explicit approach utilising conceptual Plant Functional Types. A Rule-Based Community-Level Modelling strategy is adopted to represent changes across 11 land-cover categories using conditional rules regarding environmental conditions (e.g. water and light availability) and vegetation succession-pathways (i.e. secondary vs. regeneration). Wildfire is represented in the model using a cellular automata approach that explicitly considers ignition frequency and location as a function of human activity and vegetation. Wildfire spread is modelled as a function of vegetation, slope, and wind.

This LFSM is coupled with the ABM of LUCC described in chapter five to produce an integrated socio-ecological simulation model. As with chapter four, chapter five first reviews the literature on modelling spatial land-use decision-making before then presenting the chosen model structure and rationale. The literature review in this chapter suggests that the spatially heterogeneous nature of environmental resources and land-tenure in the study area makes land-use decision-making highly dependent upon individual farmers' circumstances. Few of the previous ABMs of land-use decision-making in the literature have explicitly considered spatial land-tenure structure as a factor influencing LUCC. The ABM developed in this thesis considers two 'types' of land-use decision-making agent with differing perspectives; 'commercial' agents are perfectly economically rational, whilst 'traditional' agents represent the part-time or 'hobby' farmers that manage their land because of its cultural rather than economic value. These agents decide whether to place each piece of land they own in 'crops', 'pasture' or 'unmanaged' land-use categories as a function of their perspective, their age, the state of the market, the current land-use/cover, and the spatial configuration of their land. Each of these land-use decisions has consequences for the wildfire risk in the landscape by modifying the spatial configuration of land-cover (i.e. wildfire fuel).

The integrated model assumes that if an area of land is under human management, the land-cover is specified by that management. Thus, at each timestep the ABM determines locations of agricultural land-cover. The state of remaining unmanaged areas is determined according to the vegetation dynamics represented by the LFSM. Wildfire ignition and spread are then simulated as a function of the vegetation (land-cover) and human population in the landscape. Agricultural land-covers are assumed to provide decreased wildfire risk relative to the others considered, with resulting implications for spatial wildfire spread. After burning the resulting land-cover configuration and composition influences ecological succession and agricultural decision-making in the next timestep. Ecologically, wildfire is assumed to be stand-replacing, removing all vegetation. Vegetation regeneration then proceeds dependent upon successional pathway, availability of seed sources and other environmental resources. For agricultural decision-making, the burning of a particular piece of land will have consequences for the finances of an agent as income will be lost whilst the land is in a burnt state and costs are incurred re-cultivating the land. In this manner wildfire reciprocally influences both land-use/cover patterns and processes.

Chapters four and five conclude with initial results from testing and sensitivity analyses of the individual model components. Chapter six then presents the use of the integrated model to examine potential LUCC and the consequences for wildfire regimes as a result of different scenarios of economic and demographic change. Simulation results indicate that mean largest wildfire and mean total burned area will increase if agricultural activity declines. Furthermore, the integrated model suggests that changes in land-cover composition are driven more by human activity than wildfire. Chapter six also examines simulations not considering human activity over longer temporal extents (i.e. centuries rather than decades). These simulations show the scaling of wildfire sizes varies as a function of total and maximum land-cover flammability probabilities.

The third section of the thesis considers how interdisciplinary models of socio-ecological systems, such as the one developed in chapters four and five, should be assessed given the nature of systems they represent. Chapter seven discusses the epistemological and sociological issues that arise when ‘open’, middle-numbered systems are modelled. Middle-numbered systems, typified by the systems geographers and landscape ecologists study, are those which can be described neither by the methods developed for few interacting objects (such as Newton’s laws of motion) nor by those developed for many, many interacting objects (such as statistical mechanics). Objects, and interactions between objects, are rarely homogenous in middle-numbered systems. Specifically, the nature of these systems raises the epistemological problems of equifinality and the potential of committing the logical fallacy of deeming a model of a real-world system invalid (i.e. false) if it does not reproduce the empirical data, or valid (i.e. true) if it does (termed 'affirming the consequent' by Oreskes *et al.*, 1994). As a consequence, and in the context of continuing calls for greater public participation in science leading to the development of 'post-normal' science, a shift in emphasis in model validation away from establishing the 'truth' of the model's structure via mimetic accuracy (precisely reproducing previously observed empirical events) and toward ensuring trust in the model's results via practical adequacy (ensuring a model addresses relevant questions) is suggested. Chapter eight takes the model outside the 'ivory tower' to engage with local stakeholders in the study area to examine the potential of the validation criteria suggested in chapter seven for model validation and evaluation. The stakeholder validation exercise described in chapter eight highlights several shortcomings in the ABM representation of land-use decision-making but does suggest that is acceptable to the degree that it might be used in future regional planning. However, two shortcomings in this pubic engagement exercise are evident regarding stakeholder understanding of models and stakeholder attitudes toward future landscape change. Finally, chapter nine takes a narrative approach to summarise and reflect upon what was learnt during the research presented in the thesis. In particular, it suggests that ABM approaches to examining LUCC will need to be developed in different ways depending upon two distinct uses; one where the model is used for essential system-level understanding of the processes of LUCC, and a second where explicit representation of LUCC in the context of a specific place is intended.

CHAPTER TWO

SPANISH LANDSCAPES AND CHANGE

2.1 INTRODUCTION

Mediterranean-type ecosystems occur in regions characterised by a Mediterranean climate (Aschmann 1973). The five regions of the world with Mediterranean climates and associated Mediterranean-type ecosystems are the Mediterranean Basin, California, southern Chile, southern Africa and south and south-western Australia. Di Castri (1991 p.3) broadly defines a Mediterranean climate as "a transitional regime between temperate and dry tropical climates, with a concentration of rainfall in winter and the occurrence of a summer drought of variable length". The Mediterranean Basin possesses such a climate, characterised as it is by strong seasonal variability (Wainwright and Thornes 2004). Habitat diversity and human activity, when combined with the heterogeneous nature of the Mediterranean climate produce a biological patchiness not found in more temperate, arid or tropical regions of the world (Blondel and Aronson 1999).

Ecological and anthropic processes promote, and are shaped by, this biological patchiness, producing heterogeneous, mosaic-like landscapes of land-covers. That is, feedbacks between spatial landscape patterns and processes mean that any changes in these processes will have impacts on spatial landscape pattern, in turn precipitating further changes in processes. Recent changes in human activity have been observed in Mediterranean landscapes with corresponding changes in land-use and land-cover. This chapter opens with a very brief outline of the ecological processes occurring in Mediterranean landscapes. A more in-depth examination of the conceptual models used to understand vegetation dynamics in these regions is presented in chapter four alongside the rationale for the construction of the biophysical model. Instead, here emphasis is placed on the importance of the lengthy influence of human activities in these landscapes and the potential impacts recent changes in these activities may entail. Specific consideration is given to the case of Spain, in which the study area is located. The potential ecological implications of these changes are discussed, followed by a description of the study area and the data available to characterise it.

2.2 ECOLOGICAL DISTURBANCE

According to van der Maarel (1988 p.11), the word 'succession' in an ecological sense means, "the successive occurrence of plant communities at a given site" where plant communities are "interacting populations growing in a uniform environment and showing a floristic composition and structure that is relatively uniform and distinct from surrounding vegetation". The Clementsian view of succession (Clements 1916) emphasises succession as being a highly deterministic process of vegetation change in which the floristic composition (at a given site) converges toward a stable 'climax' plant community. This conception of a permanently stable, climactic endpoint to the process inherently views nature as being able to maintain a static equilibrium (in both time and space). This traditional (Clementsian) view of plant succession led ecologists to perceive succession in Mediterranean landscapes as characterised by the replacement of early-successional pines by oak climax communities that established themselves in the pine understory (Figure 2.1, Barbero *et al.* 1990b, Zavala *et al.* 2000). Over recent decades ecologists have abandoned this idea of static equilibrium and have begun to consider the importance of spatial pattern alongside temporal pattern (Perry 2002).

Figure 2.1 *Pinus* with *Quercus* understory An example of the classic conceptualisation of Mediterranean succession.

Ecological disturbances are relatively discrete events in time that act to remove organisms and change the physical environment or its resources (Pickett and White 1985). Thus, traditionally ecologists have perceived ecological disturbances to be ‘unnatural’, disturbing the ‘balance of nature’, as they act to disrupt the inevitable march towards a stable climax plant community (e.g. see Perry 2002). However, disturbance is now seen as playing a central role in many ecosystems – some disturbances are promoted by the biotic components of the environment, and at broader spatial scales and over longer periods of time disturbance may maintain floral composition and structure in a dynamic equilibrium (Holling 1973, White 1979). This perspective is imperative in the Mediterranean basin, where climatic stress and human activity mean frequent disturbance by burning and grazing. These environmental stresses produce characteristic Mediterranean community structures driven by competition for water and light in wake of these disturbances and along resource gradients (Vila and Sardans 1999).

In Mediterranean-type ecosystems high rainfall during late autumn/early winter months allows vegetation to flourish following long hot, dry summers. By the late summer the following year, vegetation (and climate) conditions have reached a state where widespread wildfire occurrence is inevitable. So frequent and ubiquitous is fire in the Mediterranean landscape, that many species have developed ‘fire-adaptive traits’. Although widely used, this term has been criticised by Whelan (1995) because species’ responses vary according to different fire conditions (intensity, frequency, etc.) and because it implies fire is the only selective force producing the fire-adapted characteristic. Generally, there are two strategies adopted by vegetation in response to this frequent burning – post-fire seedling establishment (‘seeder’ strategy) and post-fire vegetative regeneration (‘resprouter’ strategy). These alternative strategies are the result of a trade-off between expending resources on the maintenance of the current plant generation versus the next (Bellingham and Sparrow 2000). Serotinous pine species are representative of the seeder strategy and are typical in Mediterranean-type environments. These species store mature seeds in a canopy seed-bank (i.e. pine cones) ready for post-fire dispersal and germination (Barbero *et al.* 1998). Late-successional, evergreen oak species adopting the resprouting strategy are also widely found and are the other half of the traditional conception of Mediterranean succession. The importance of this conceptual pine-oak, seeder-resprouter model as a response to frequent burning is highlighted and discussed in more detail in chapter four.

Mediterranean ecosystems have also traditionally been disturbed (either directly or indirectly) by human activity. Tree crops are routinely cut for management purposes and land is widely used to graze livestock. Historically humans have used fire to clear land and improve grazing:

> *Typically, shepherds burned as they departed the mountains, in advance of the rains, and what escaped the autumnal flames they captured in the fires of the spring. The tradition was already ancient, inherited from the vanguard of the Neolithic revolution whose fires and flocks had begun a revolution in land use that endured, in many places, for 10,000 years.*
>
> Pyne (1997 p.89)

A common human land-use that persists in Spain today is the open, savanna-like landscape of managed oak woodlands known as *dehesa* (see section 2.3.2 below). Grazing livestock on grasslands between trees in these woodlands prevents growth of saplings and has been shown to reduce the number and quality of acorns available to initiate future establishment (due to predation Leiva and Fernandez-Ales 2003, Cierjacks and Hensen 2004). The maintenance of fields exclusively for cereal cropland prevents any succession-type processes to occur, reducing species diversity and above-ground biomass that otherwise might occur. The importance of human disturbance becomes even clearer when the longevity and extent of the human presence and influence in the environments is properly understood.

2.3 HUMAN PRESENCE

2.3.1 Introduction

In the contemporary Mediterranean Basin, Margaris *et al.* (1996) suggest that the vast majority of landscapes have been modified by human activity and can currently be described as being in one of three states – cultivated, semi-natural (non-cultivated but managed), or built upon. The traditional land-uses in Spain, and the causes of recent change in them, are now briefly highlighted.

2.3.2 Traditional Land-Use in Spain

Although dates are still contested, evidence suggests early human occupation of the southern European Mediterranean during the middle Pleistocene, around 1 million years before present (Oms *et al.* 2000). Evidence suggests that the westward spread of agriculture and associated sedentary settlements began in the eastern Mediterranean Basin *circa* 8,000 BCE, arriving in Spain *circa* 6,000 BCE (Wainwright and Thornes 2004). Human modification of landscape processes and patterns (primarily patterns of vegetation) grew slowly as initial technological constraints restricted change. A later 'secondary products revolution' adopted domesticated animals (e.g. horses, cattle) for transport and power (notably for use with ploughs which arrived in Spain *circa* 3000 BCE) and promoted more extensive and efficient use of landscape resources (Sherratt 1981). Gilman and Thornes (1985) suggest that irrigation (ditches and canals) in dry-land areas in Spain also came into use around this time, and evidence for the use of fire for agricultural land clearance has been proposed (see Wainwright and Thornes 2004). Further mechanisation of agricultural practices (beyond the basic plough or threshing sledge) was slow in Spain until the late 19th century, when global trade and better technology improved agricultural techniques (Grove and Rackham 2001). Nevertheless, in many areas where relief prevented the use of machinery, water is scarce or economic development was simply slow, traditional agricultural practices and land-management techniques dating back thousands of years persisted into the 20th century.

Traditional Spanish agricultural land-use may be split into two broad categories: managed, multi-use woodland and monoculture agriculture. Periodic cutting and felling of trees, allowing re-growth from 'stools' (a permanent, live tree base), maintains oak woodland at a low density (between 40 – 50 trees ha^{-1}) with space between available for alternative uses such as cultivated cereal or native grass for pasture (Joffre and Rambal 1993). Trees may be 'coppiced', 'pollarded', 'shredded', or in the case of *dehesa*, 'lopped' (Grove and Rackham 2001), reducing above-ground biomass. These agro-sylvo-pastoral systems have existed in Spain over an extended period of time, possibly since 4,000 BCE (Stevenson and Harrison 1992). Stevenson and Harrison (1992) examined the evidence from pollen cores in southern Spain, and suggest *dehesa* gradually developed over 3,000 years before a phase of forest destruction by fire (and possibly overgrazing) resulting in less productive scrubland. From *circa* 500 BCE *dehesa* systems began to recover from this destruction, to the well-defined states observed in the 19th – 21st centuries. Over the centuries it has been suggested that *dehesa* has come to influence hydrological processes in Mediterranean semi-arid environments (Joffre and Rambal 1993, Joffre *et al.* 1999). Grazing in the space between and under trees prevents invasion of shrub species, and so maintains areas with a rich diversity of native grass species and reduced above-ground biomass (Rackham 1998). *Dehesa* has multiple farming functions and products; sheep, goats and cattle graze grasses to produce wool, meat and dairy products; pigs graze acorns from the trees; cork is harvested where cork oak are present; cereals are harvested where planted; and wood from the management practices of pollarding or lopping is collected (Joffre *et al.* 1999, Allen 2001, Grove and Rackham 2001). *Dehesa* is found on rolling or flattish ground and may "march over hill and valley, over pasture, arable or even roads as far as the eye can see" (Grove and Rackham 2001 p.197, e.g. Figure. 2.2)

Figure 2.2 Example *dehesa* found in SPA 56. The land between trees in the foreground of this *dehesa* landscape is being used for arable crops in contrast to the areas in the distance that are reserved for grazing.

Agricultural products produced in Spanish monocultural landscapes include cereals, olives and vines. Central Spain's semi-arid climate (mean annual rainfall is 400 – 800 mm depending on altitude), intra- and inter-annual rainfall variability, irregular and often sharp relief, variable soil quality and high mean altitude (88% of the country is between 200 – 2000 m ASL) make for adverse farming conditions in many areas and have led to a "mosaic of farming landscapes with an uneven production capacity and a complex social and environmental composition" (Peco *et al.* 2000 p.146). These poor farming conditions led to low mean cereal (mostly wheat and barley) yields in Spain compared to northern European countries, but comparable to the yields of other southern European countries (e.g. Portugal, Italy and Greece – Pinto-Correia 1993a). Much of this cereal is used for pig, cattle and chicken feed. On land unsuitable for cereals – due to poor quality soils or lack of water – olives are often found (Allen 2001). Vines for wine or grape production are usually irrigated in some way (Grove and Rackham 2001) and grown both as a part of traditional small holdings and over larger spatial extents (Allen 2001). Harrison (1988) suggests that olives and vines were first grown for agricultural production (rather than collected from woodlands and meadows) sometime between 800 – 450 BCE. Recently, these cereals, olives and vines were estimated to cover 60% of the country's agricultural land (see Peco *et al.* 2000). The remaining area is devoted to almonds, figs, and fruit and vegetable crops (some of which are grown intensively in greenhouses).

2.3.3. Traditional Spanish Land Tenure

The social and economic imbalances leading to the Spanish Civil War in 1936 have been characterised by several historians as a result of the unequal distribution of land ownership and resulting social structure (Malefakis 1970, Brenan 1990, Carr 1993). The distribution of property throughout the 20th century has been described as 'sub-optimal' with regard to production (Simpson 1995) and the regional variation between extremes in spatial characteristics of land holdings is often highlighted (Malefakis 1970, Brenan 1990, Carr 1993). Traditionally, land holdings in the north and centre of Spain (notably Galicia) are small (less than 10 hectares) and often unable to support a single family (*minifundia*), whilst southern Spain is composed of large farms and land holdings (greater than 100 hectare, *latifundia*). The large *latifundios*, that blocked much agricultural and social reform in the years prior to the outbreak of civil war, were justified at the time by the necessity to offset poor yields due to low rainfall (Simpson 1995). More recently, this has been disputed and the link between aridity and farm size dispelled (Malefakis 1970). Contemporary statistics suggest that over three quarters of land holdings in Spain are currently 'small' according to the above definition, with less than 2.5% being 'large' (Ministerio de Agricultura 2003). The highly fragmented nature of land tenure, present in the study area here, means LUCC will occur at a fine spatial resolution, complicating both its representation and implications due to pattern-process feedbacks (as discussed in chapters three and five).

2.3.4 Contemporary Spanish LUCC

Land-use change has occurred across much of the Mediterranean since the 1950s due to advances in agricultural technology and socio-economic changes (Margaris *et al.* 1996). Simpson (1995) suggests that in Spain the adoption of artificial fertilisers and labour saving machinery was limited before the 1950s because of a combination of the poor performance of fertilisers on dry-lands, the negative economic impacts of the Spanish Civil and Second World wars during the 1930s and 1940s, and the persistence of cheap manual labour within the Spanish economy. In Spain, industrialisation and the increasing importance of the newly capitalist agricultural market following the Second World War have led to agricultural expansion and intensification in many areas, but abandonment of agricultural land in, and migration away from, marginal agricultural areas (Peco *et al.* 2000). In the interval 1970 – 1991 the irrigated land area in Spain increased by a third and the level of mechanisation almost tripled (see Peco *et al.* 2000). Despite this intensification, agricultural land abandonment has accelerated over the last 25 years in marginal areas. These marginal areas are generally steep and rocky and characterised by poor, shallow and dry soils making the introduction of modern, mechanised farming practices difficult, and in turn preventing the increases in production possible in other areas (Pinto-Correia 1993a). Grove and Rackham (2001) suggest, however, that easier ways of making a living, depopulation, and improvements in transport reducing the need for locally grown food, are the underlying causes of land abandonment, exacerbated and reinforced by technological changes.

Pinto-Correia (1993b) outlines two potential pathways by which *montado* (the Portuguese equivalent of *dehesa*) may become abandoned. First, grazing of the grass or shrub understorey may be reduced and less care taken to maintain tree crops. Progressively, tree crop yields fall simultaneously with the invasion of shrubs and biomass accumulation in the understorey. Exploitation of the land now becomes more difficult and management reduces even further until the land is totally abandoned. Alternatively, intensification by increased mechanisation demands a reduction in tree density and grazing is excluded to allow planting and harvest of cereal or other soil crops. Yields of new crops may remain low (or fall after initial increases) as soil erosion increases and soils become depleted. Land abandonment follows as the environmentally unsustainable land-use change produces economically unsustainable yields. Clear differences in appearance are evident for well-managed landscapes (e.g.

Figure 2.3a) compared with abandoned landscapes (e.g. Figure 2.3b) due to the invasion of shrubs (resulting in increased biomass).

Figure 2.3 Land abandonment in SPA 56. a) a well managed landscape and b) an abandoned landscape.

2.3.5 European Agricultural Policy

Alongside changes in agricultural practices due to technological advances, it is important to highlight the importance of European Union agricultural policy as a driver and accelerator of both intensification and abandonment. Spain entered the EU in 1986, thereby coming under the influence of the Common Agricultural Policy (CAP). CAP subsidies led to widespread reparcelling of agricultural lands in Spain, putting pressure on the small family-based farming units that have been characteristic for hundreds of years. In Spain it is estimated that 75% of subsidies for production have gone to the larger 20% of farms (Gravina 2003). Combined with technological advances this has promoted the aggregation of small farms into medium and larger farms that can benefit from greater capital and technological input. Set-aside subsidies have also complicated matters in Spain where it may be difficult to distinguish between abandoned land and set-aside, hindering management efforts. Furthermore, the CAP does not distinguish between farmers for whom farming is their sole livelihood and farmers who have secondary employment and farm as a secondary income. It is suggested these aspects of the CAP have contributed to decreases in agricultural employment, numbers of small farms, and increases in land abandonment (Gravina 2003).

Recently the EU has recognised that agriculture is an important component in natural resource management and rural economic sustainability and development, and begun implementing reforms (‘Agenda 2000’ in March 1999 and further reforms in 2003) to make policy (more) agri-environmentally oriented. These reforms attempt to promote the market and reduce trade distortion, strengthen rural development policy through measures to make rural areas more economically competitive, and to maintain and preserve the environment. Measures proposed by the Spanish government to achieve these aims include compensation for farmers whose production is restricted by natural resources (e.g. areas of steep relief), support for farmers to retire if land is made available for a new land holder, support for farmers who present improvement plans for their land, and support for young farmers (less than 40 years old) wishing to establish farming activities. These measures are important in a country where in 1999 over 40% of farms were owned by farmers older than 55 years with no relative in line to inherit the business (Ministerio de Agricultura 2003). Thus, many of these measures are designed to reduce migration out of, and abandonment of land in, marginal agricultural areas. These agricultural policy changes accompany the 6th EU Environment Action Programme, established in 2002 with aims to break the perceived link between economic growth and environmental damage. Key aspects of this programme relevant here include objectives to protect and restore ‘natural’ systems’ functions and to halt the decline of European and Global biodiversity. These aims hope to be achieved by extending the Natura 2000 network (see section 2.5.2 below), by extending programmes to promote sustainable forest management and by introducing measures to protect and restore landscapes.

2.3.6 Summary

Socio-economic and technological changes since the mid-Twentieth century have led to changes in land-use. Recent EU policy moves aim to protect and maintain the environment in light of these changes, but will need to be allied with appropriate management practices based on sound understanding of the processes in the environmental systems under treatment. In Mediterranean Basin ecosystems this understanding should not preclude consideration of the integral nature of humans within them, such has been the extended interval over which traditional land-uses have been employed. The spatial configuration and composition of land ownership is also important for studies of LUCC in Spain, characterised as it is in the north and centre by small, fragmented land holdings.

2.4 POTENTIAL CONSEQUENCES OF LUCC IN SPAIN

2.4.1 Introduction

MacDonald *et al.* (2000) highlight four potentially detrimental environmental consequences of land abandonment; desertification and soil erosion, reduced biodiversity, disruption of landscape (ecological) structure and function, and increased natural hazard risk (particularly fire in Mediterranean regions). These latter issues (landscape structure and wildfire hazard) are of primary interest to this thesis, but also have influences on the other potential consequences listed. These potential consequences are now examined in turn.

2.4.2 Desertification

Desertification is an environmental issue of concern in the Mediterranean, and large projects have been funded to examine it (e.g. the EU funded MEDALUS project). Thornes (2002) defines land degradation as the reduction or loss of biological and economic productivity due to LUCC or another physical process. However, the processes causing desertification are as poorly understood and confused as the concept of land degradation itself (Wainwright and Thornes 2004). Indeed some authors have been led to question whether the observed phenomena associated with desertification are not ongoing processes of change that occur ‘naturally’ in Mediterranean regions (e.g. Grove and Rackham 2001). Nevertheless, several studies have recently examined the effects of land-use on soil erosion and land degradation, illustrating differences in soil protection afforded by different land-covers (e.g. Kosmas *et al.* 1997, Dunjo *et al.* 2003). Generally these studies suggest that land-use practices that involve clearing ground cover to expose bare soil and/or utilise machinery, such as the cultivation of vines, lead to the greatest soil losses. Possible increases in fire frequency related to abandonment (see section 2.4.4 below) may have severe impacts on soil erosion and land degradation, especially if individual fire events coincide with intense rainfall events (De Luis *et al.* 2003, Pardini *et al.* 2003, 2004).

2.4.3 Biodiversity

In a long-range study of future global biodiversity Mediterranean ecosystems were forecast to be one of those experiencing the greatest proportional change in biodiversity by 2100 with LUCC being a major contributor (Sala *et al.* 2000). Across all 24 of their pan-European study areas, (MacDonald *et al.* 2000) found only one (on the Greek mainland) in which the impacts of land abandonment upon biodiversity were positive. However, other authors suggest that in the majority of Mediterranean regions abandonment will have positive impacts upon levels of biodiversity (Baudry 1991, Gonzalez Bernaldez 1991). MacDonald *et al.* (2000) themselves note that in Spain, partridge habitat lost during land reforms that reduced hedges and field margins because of land regrouping, may be restored and thereby reintroduce habitat diversity. Further, numerous studies in the Mediterranean Basin have shown that fire promotes vegetation species diversity in the few years following fire, before species numbers slowly decline (e.g. see Figure 2.4, Trabaud 1994). Following removal of the canopy and larger individuals, there is increased potential for competition between species for increased resources (e.g. light, nutrients in remaining ash etc.) leading to a flourish in the number of species. With time however, Trabaud (1994) suggests that the dominant species prior to the fire will return. Thus, the impacts of land abandonment on biodiversity levels are likely to rest upon consequent processes that act to change landscape heterogeneity and habitat diversity in time and space.

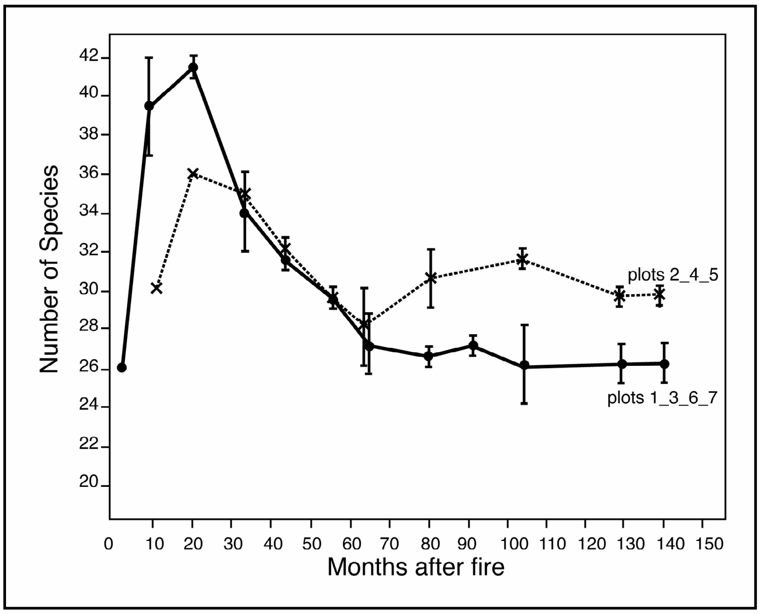

Figure 2.4 Vegetation species richness over time following fire. Data are for a dense *Quercus ilex* coppice. Source: Trabaud (1994)

2.4.4 Landscape Structure and Natural Hazard Risk

Landscape (ecological) structure and function and increased natural hazard risk are closely related in the Mediterranean because of the important ecological role that fire plays in these environments (section 2.2). Increases in flammable biomass, due to biomass accumulation following abandonment of agricultural or managed land (see section 2.3.4 above), is likely to be related to increases in fire risk and occurrence (Moreira *et al.* 2001, Romero-Calcerrada and Perry 2002). During recent decades wildfire in the Mediterranean has increased, both in terms of individual occurrences and total area burned (see Figure 2.5, Moreno and Oechel 1994, Pausas and Vallejo 1999, Vazquez and Moreno 2001). Some authors suggest that human influence has caused these changes in wildfire activity (Moreno 1996) while others have found changes in climatic variables are related to these increases (Vazquez and Moreno 1993, Pinol *et al.* 1998). Alternatively, it has been suggested that a combination of the two is the likely cause – De Luis *et al.* (2001) suggest a combination of changes in rainfall and land abandonment during 1961 – 1990 have made conditions for wildfire more favourable.

Land abandonment is not necessarily a uniform occurrence over large areas of land. As previously described (section 2.3.3) land ownership in Spain can be highly fragmented, often resulting in highly spatially heterogeneous land-use decision-making. Grove and Rackham (2001) have observed that if people are unable to continue cultivation of all of their land they may retreat to land easier to manage. This, along with fragmented land ownership, can result in isolated patches of land that remain cultivated amongst larger areas of abandoned land, and *vice versa*. Likewise, more marginal land within a farm may be abandoned in favour of intensified use of other parts of the farm to maintain overall income (Pinto-Correia 1993a). This spatially variable pattern of abandonment contributes to, and may even exacerbate, the highly fragmented, heterogeneous and mosaic-like land-cover pattern in the landscape. However, increased fire occurrence, resulting from increases in fire risk due to abandonment, may act to reduce this heterogeneity by converging smaller patches into larger burned areas (Perez *et al.* 2003). Theoretically, this homogenisation of the land-cover will promote the spread of disturbance (e.g. see Turner *et al.* 1989) leading to the occurrence of larger fires. Other authors suggest that actually it is land abandonment and the coalescence of unmanaged vegetation patches that will act to cause homogenisation, while fire *increases* heterogeneity of the landscape (Lloret *et al.* 2002). However, Lloret *et al.* (2002) found that for their study area in north-east Spain, the increase in heterogeneity caused by fires did not outweigh the homogeneity caused by agricultural abandonment and corresponding coalescence of natural vegetation patches. These previous studies in the Mediterranean Basin agree that landscape homogeneity has increased but a lack of consensus remains regarding the sequence of events and the processes that have caused this, and therefore for what will happen in the future.

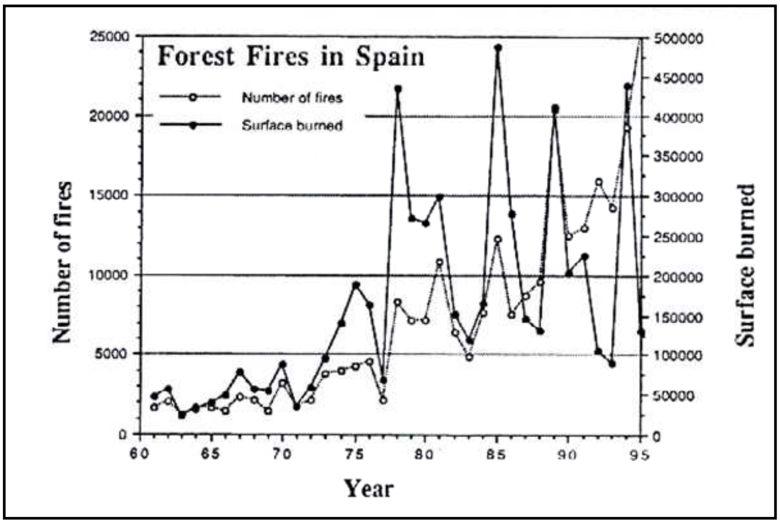

Figure 2.5 Recent trends in Spanish wildfire occurrence. Increases are observed in both occurrences of individual fires and total annual area burned over the last decades. Source: Moreno (1996)

The exact changes in landscape structure that may occur, due to changes in landscape pattern and reciprocal changes in landscape process and function in the future, remain unclear and may vary non-linearly with causal variables or be contingent upon changes at specific locations in the landscape. For these reasons, investigation into the impacts of LUCC on wildfire will need to consider all drivers of change (i.e. biophysical and socio-economic) as spatially explicit processes.

2.4.5 Summary

Recent technological and socio-economic changes in the Mediterranean Basin have led to increasing change in land-use, and notably the abandonment of agricultural and managed land. Pattern-process interactions between wildfire, vegetation dynamics and human land-use are unclear due to non-linear feedbacks and historical landscape contingency (i.e. contemporary and future events are influenced by those in the past) with a wide range of possible future outcomes.

2.5 EU Special Protection Area 56

2.5.1 Introduction

Agri-environmental policies have been implemented recently to address the conservation and land management issues discussed in the previous section. The European Community (EC) 'Bird Directive' (79/409/EEC) was passed in 1979 to prevent pollution in, or deterioration of, habitats of species in danger of extinction, vulnerable to specific changes in their habitat or considered rare because of small numbers or limited local distribution. In 1994 the directive was updated (by EC 'Habitats' Directive 92/43/EEC) to ensure that any plans or projects not directly linked to the management and conservation of the species and habitats protected are assessed for their impacts upon the area and its conservation objectives. If likely impacts are negative, such projects will be prevented unless designated as imperative for reasons of overriding public interest. SPAs are now an integral part of the Natura 2000 network. In Spain over 50% of land falling into the Natura 2000 scheme is covered by farmland habitats (EEA 2005).

The specific landscape that this project focuses on is the EU Special Protection Area number 56, 'Encinares del río Alberche y Cofio' (SPA 56), in central Spain (Figure. 2.6). SPA 56 covers approximately 83,000 hectares and is located 40 km to the southwest of the city of Madrid. Lying on the southern slopes of the Sierra de Guadarrama and Sierra de Gredos, altitudes range within the study area from 600 m ASL in the southeast to 1300 m in the northwest. These altitudinal differences impose variation upon the local climate, which is Mediterranean in nature and characterised by a prolonged hot, dry summer. Mean annual rainfall is around 700 mm but ranges from 800 mm in the higher altitudes to 400 mm in the lower-lying areas. Mean annual temperatures present a similar spatial pattern, ranging from 10°C in the upper altitudes to 16°C in lower areas. The SPA mainly occupies the middle basin of the River Alberche but is also drained by the rivers Cofio and Perales. Granite, gneiss, Neogene and Quaternary sediments dominate the lithology of the area.

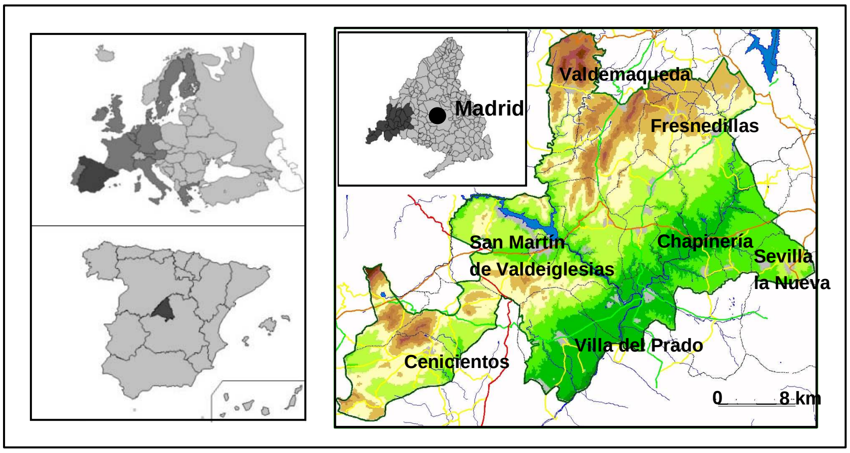

Figure 2.6 Location of SPA 56. The SPA encompasses all or part of 19 municipalities in the Autonomous Region of Madrid.

2.5.2 Habitats

SPA 56 is an area of outstanding ecological and aesthetic value and contains a wide variety of plant species, animal and bird habitats and human land-uses. The area is home to a highly diverse range of wildlife and has been described as one of the most outstanding ecological enclaves within the Madrid region (Martinez *et al.* In Press). Because of the diversity of habitats and environments in this area, in 1990 it was designated as an SPA under the EU Bird Directive.

The diverse habitats and environments present in SPA 56 include deciduous woodlands, pine forests, holm oak woodland, *dehesa*, meadows, grasslands, scrubland, rocky outcrops and crags, riparian habitats and others besides. SPA 56 is estimated to contain 30,000 ha (300 km^2, 37% of the total area) of 20 habitats included for protection in Annex 1 of the EC 'Habitats' Directive 92/43/EEC (Martinez *et al.* In Press). Oak species ('resprouter' – see chapter four), primarily *Quercus ilex* (Figure 2.7a) is widespread in the study area in both arboreal and shrub form (across ~ 25% of SPA 56), often in mixed stands with juniper (*Juniperus phoenicea*) or pine species, but also as wide expanses of *dehesa* (see section 2.3.2 and Figures 2.2 and 2.3a). Pyrennean oak (*Q. pyrenaica*) is also found in SPA 56, but is restricted to mountain slopes where it alttitudinally substitutes Holm oak. Pine ('seeder' – see chapter four) forest is also widespread (~ 20%), composed of a mixture of natural and plantation stands. *Pinus pinea* and *P. pinaster* are the dominant pine species, and they are mostly found in the northern areas and higher altitudes of the study area (Figure 2.7b). Other tree species found are chestnut (*Castena sativa*) and, in riparian areas, alder (*Alnus glutinosa*). Fragmented grassland composes less than 10% of the study area and is characterised by species of *Cistus* (e.g. *Cistus ladanifer* and *C. laurifolius*), lavender (e.g. *Lavandula pedunculata*) and *Genista* (e.g. *Genista hirsuta*). Exclusive grazing land (as opposed to *dehesa*) occupies around an eighth of the SPA and is generally found in mountainous areas and on steeper land. The remaining area (~ 15%) of SPA 56 is used for cultivation of cereals, vines, olives, almonds and figs. Intensive farming of fruit and vegetable crops, utilising greenhouses, has also recently emerged. Such agriculture is found in the low-lying areas with fertile soils in the south and west of SPA 56.

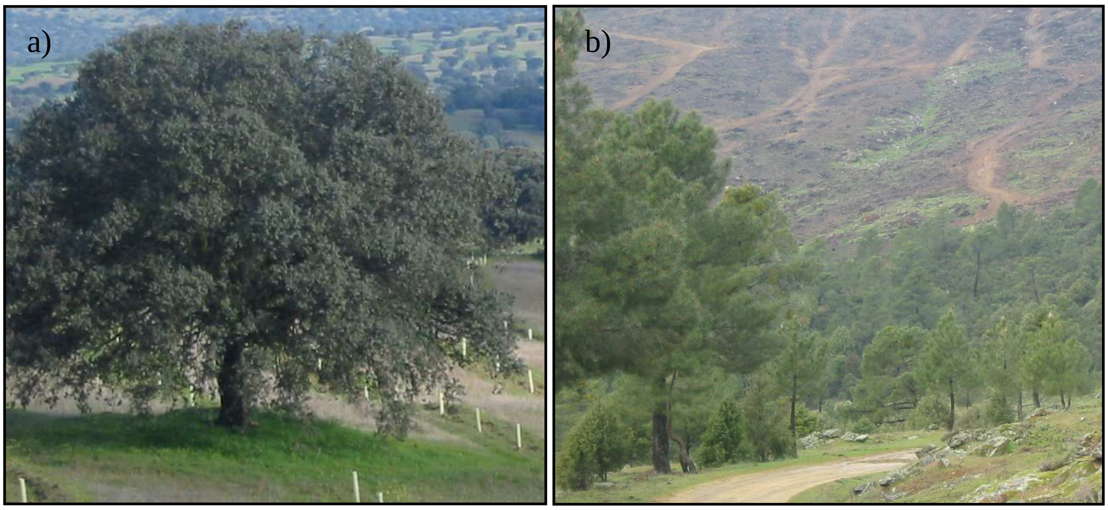

Figure 2.7 Predominant vegetation in SPA 56. Examples of a) Quercus ilex (arboreal form) and b) Pinus halepensis. Both species are common and widespread across Spain. Note the scar of a recent wildfire in the background of b).

2.5.3 Endangered Species

Much of the woodland-cover described above is vital habitat for protected birds and other endangered species. Over 150 bird species may be found in SPA 56 including populations of several species that are in grave danger of becoming extinct on local, national and European levels. The endemic Spanish Imperial Eagle (*Aquila adalberti*) is a notable example – with less than 400 individuals remaining this is one of the most endangered birds of prey in the world (Ferrer 2001). Other endangered species include the Black Vulture (*Aegypius monachus*), the Griffon Vulture (*Gyps fulvus*), the Eagle Owl (*Bubo bubo*) and the Black Stork (*Ciconia nigra*). As well as endangered birds, SPA 56 contains habitat and populations of endangered mammals including the Iberian Lynx (*Lynx pardina*), endemic to the Iberian Peninsula, and the Otter (*Lutra lutra*). To ensure their survival the habitats and environments that these species inhabit must be protected. Further, it is apparent that not only their habitat must be conserved, but also that of their prey. For example, both the Spanish Imperial Eagle and the Iberian Lynx preferentially prey upon the common rabbit (*Oryctologus cuniculus* – Ferrer and Negro 2004). Without the presence of adequate rabbit stock the conservation of Eagle and Lynx habitat is worthless, highlighting the importance of considering the ecology of SPA 56 as a whole when examining the impacts of future potential land-use change.

2.5.4 Human Activity

As noted above, around tone quarter of SPA 56 is devoted entirely to either agriculture or grazing land. *Dehesa* is used extensively across the area for the grazing of livestock, and pine forests are frequently taken advantage of for the production of timber, resins and pine kernels. The human population of the 19 municipalities composing SPA 56 was around 35,500 in 1999 (Romero-Calcerrada and Perry 2004). This number of inhabitants means a population density much lower than the average for the autonomous region of Madrid (in which the municipalities are located, 43 inhabitants/km^2 vs. 641 inhabitants/km^2), but highlights that SPA 56 is not simply a ‘closed’ nature reserve devoid of human influence. Human activity has produced a complex, multi-use landscape with the mosaic nature typical of Mediterranean landscapes.

Whilst restrictions on development and certain farming practices are in place, the inhabitants of the municipalities within the SPA are free to make their own choices about how to use their land and make a living. Despite an approximately 30% increase in population during 1985 – 1999, the area has experienced a decline in both the number of farmers and total cultivated area. This is likely to be due to a drop in the recruitment of younger people into the sector – mean farmer age has increased despite growth in the younger parts of the population as a whole. These younger workers have moved into other areas of the economy such as the construction and service sectors. Indeed, the suggestion is that the area might become part of Madrid’s commuter belt and/or ‘playground’, as the eastern side of SPA 56 is only a 45 minute drive from the centre of Spain’s capital city. As both the population of the city and the service sector of the region’s economy grow (allowing more free time for recreation) demand for recreation and residence outside the city is likely to increase. To highlight this, an increasingly evident trend across SPA 56 is the construction of secondary (holiday) residences in municipalities with recreation opportunities (e.g. Pelayos de la Presa and San Martin de Valdeiglesias near the reservoir and its marina, see Figure 2.8a) and primary residences in municipalities with good communication links with the city of Madrid (e.g. Sevila la Nueva and Villanueva de Perales – see Figure 2.8b). The EU itself cites economic diversification as a priority for action in its rural development programme for the Madrid region, and suggests that rural tourism in the region offers a real opportunity for development (EU 2000).

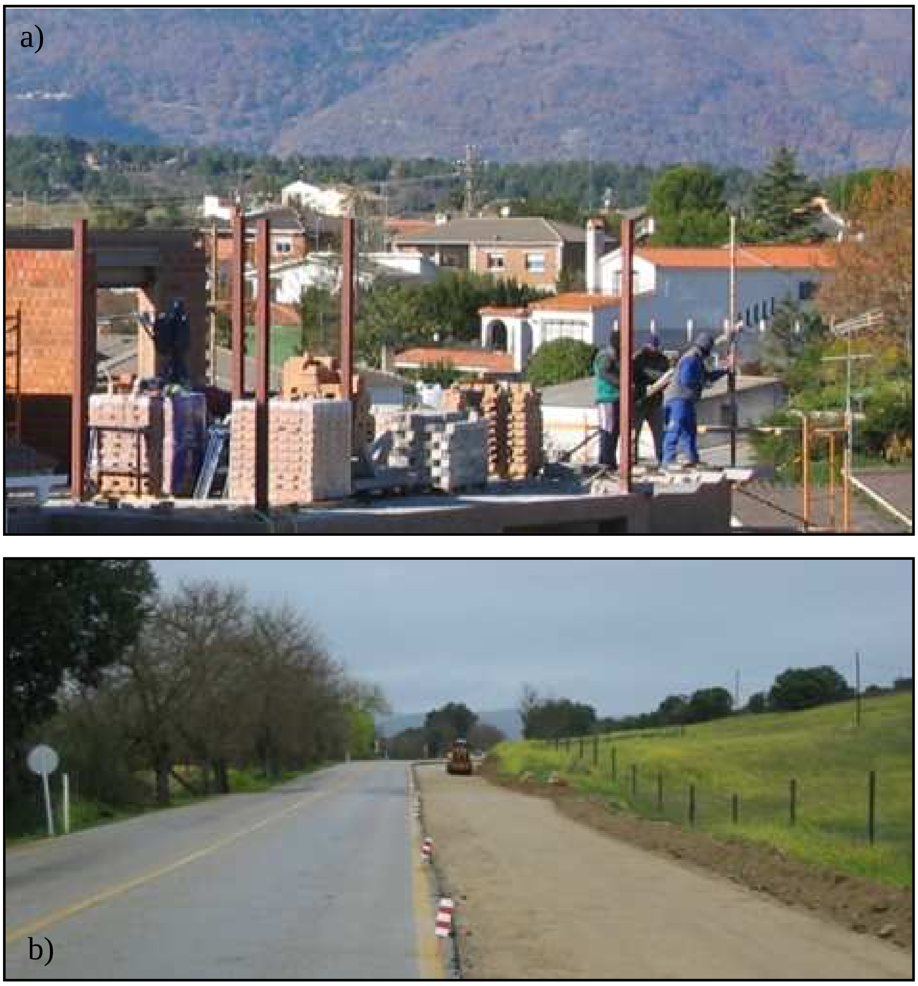

Figure 2.8 Evidence of socio-economic changes in SPA 56. a) Construction of new homes and b) widening of roads for improved communications with Madrid.

As with other parts of the Mediterranean (and the EU as a whole) these economic and social shifts have led to a decline in traditional agricultural practices and the abandonment of marginal land (MacDonald *et al.* 2000, Allen 2001). These current and future shifts depend on social and economic process and phenomena occurring at local (municipality/Madrid), regional (Spain/EU) and global scales, making projection into the future very difficult due to the high unpredictability and complexity of these systems, and their interaction across multiple scales. As discussed above (section 2.4), abandonment of land and traditional agricultural activities has potentially detrimental consequences for landscapes that have been used in such a manner for thousands of years. Romero-Calcerrada and Perry (2004) found significant compositional changes in the landscape of SPA 56 between 1984 and 1999 as scrubland replaced pasture due to progressive land abandonment (Figure 2.9). Using landscape pattern metrics and transition matrix models they found the configuration of land covers to be static through time, but that increasing rates of land abandonment (i.e. changes from pasture to scrubland) were evident in the landscape. Such changes are likely to affect the conservation of the protected species described above due to changes in their habitat, and that of their prey. Further, it is likely that shifts from pasture to scrub will increase wildfire risk due to increases in available fuel (Romero-Calcerrada and Perry 2002, Millington 2005). As with most Mediterranean environments fire is an integral component of the landscape's ecology, with fire occurring annually.

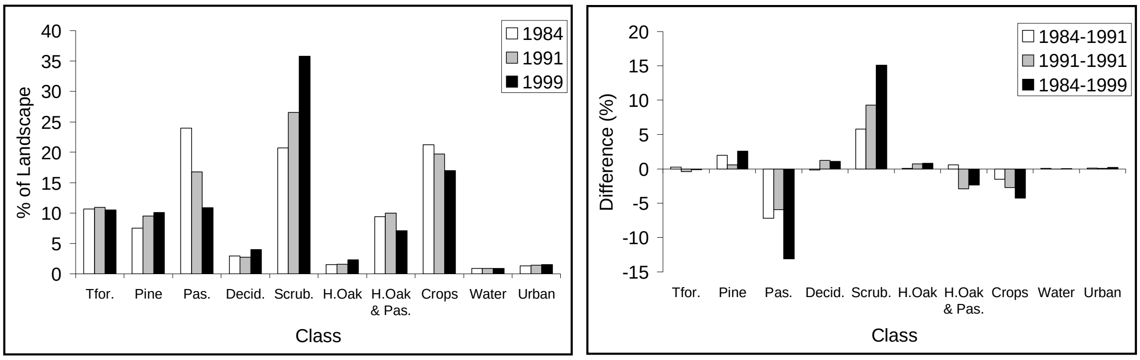

Figure 2.9 Observed land-cover changes in SPA 56 between 1984-99. a) Proportion of landscape for each land-cover class and b) Proportional change between years for each land-cover. Observed changes show a strong shift from agricultural land toward scrub and forest, as found by Romero-Calcerrada and Perry (2004).

2.5.5 Data Available for SPA 56

2.5.5.1 Introduction

The primary data available for SPA 56 are categorical land-cover maps for three different years (1984, 1991, and 1999). Climatic, topographic, land capability and social data for these years (or neighbouring years) are also available for use, derived and used previously by Romero-Calcerrada (2000). Aspects of the spatial structure of the landscape were derived from these and other thematic maps. All data maps are subset to cover the same extent and resolved to a 30 m resolution (grain), resulting in maps containing 900,453 data points (pixels). These data maps are used in the empirical LUCC modelling presented in chapter three, and the simulation model presented in chapters four and five, and are described now in turn.

2.5.5.2 Land-Cover Maps

Three land-cover maps derived from Landsat Thematic Mapper imagery for the years of 1984, 1991 and 1999 were made available for this study (see Figure 2.10). These data were previously used by Romero-Calcerrada and Perry (2004) and details of the classification and manipulation of the remotely sensed imagery can be found therein. Originally the land-cover data described 15 land-cover classes, but Romero-Calcerrada and Perry (2004) re-classified it to eight land-cover classes for their purposes. Here the data were re-classified into 11 classes (see legend in Figure 2.10). These classes were deemed more suitable for the modelling exercise at hand, reducing unnecessary complexity in the model that 15 classes might produce, but improving potential for a more accurate mechanistic representation over the eight land-cover classes. The main point to make regarding this classification is the lack of a *dehesa* cover type. In this model "Holm Oak & Pasture" is the land-cover class that closest represents *dehesa*. This class has not been renamed *dehesa* as it includes some areas that are not technically *dehesa*, and excludes some areas that are (these fall into the Holm Oak land-cover class). This problem highlights the difference between 'land-use' and 'land-cover' – remotely sensed images (like those used here) are based on the spectral response of the land surface and therefore simply record land-cover. However, different land-uses may actually present very similar spectral responses and be grouped together during image classification as a single land-cover. In this case, for example, low density oak forest will present a similar spectral response to that of *dehesa*, but the human activity and use of the land may be markedly different between the two. Thus, the classes used here represent 'land-cover' rather than 'land-use'. Further, these classes will be the basis for the vegetation dynamics model to be developed and therefore are used in the exploratory data analysis here.

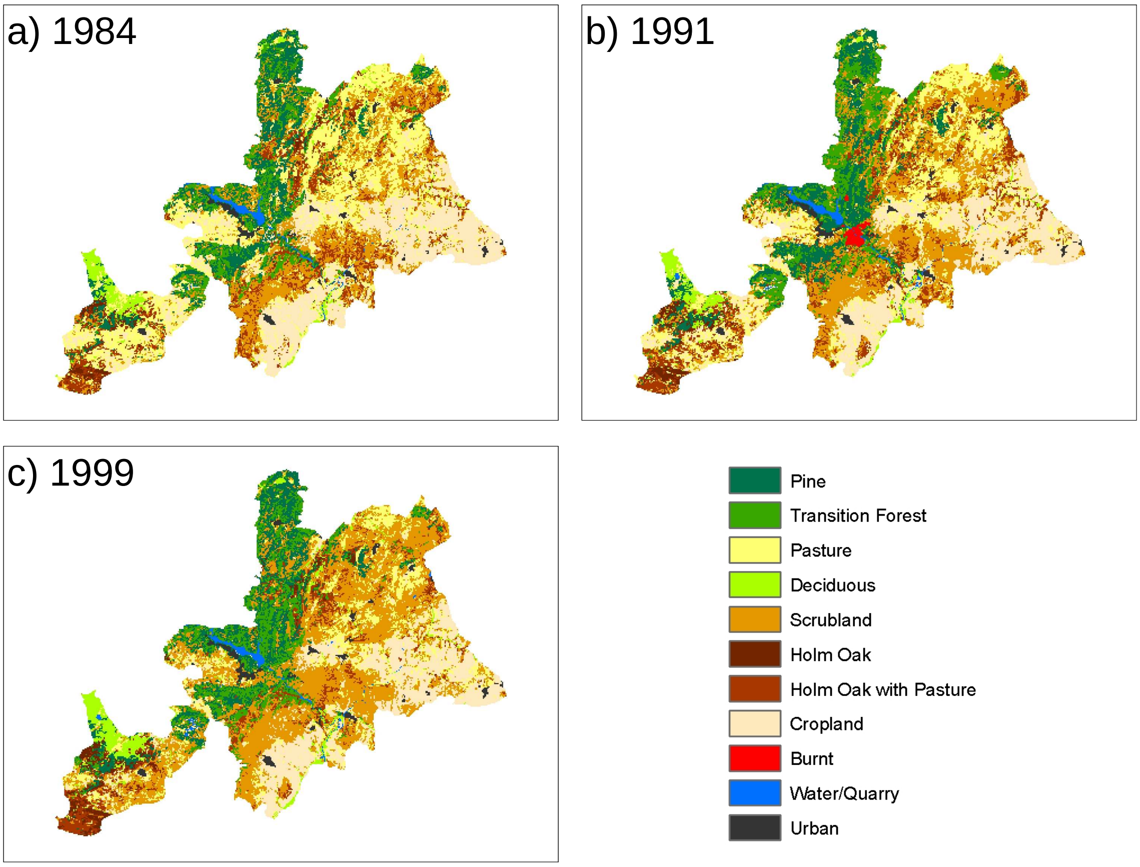

Figure 2.10 SPA 56 land-cover maps. a) 1984, b) 1991, c) 1999 These maps distinguish 11 classes of land-cover and were derived from Landsat TM imagery.

These land-cover data are in raster format (grids of square cells termed here as 'pixels') with a spatial resolution (grain) of 30 m. With land-cover maps (*LC*) for three years, three periods of land-cover change can be examined; 1984 – 1991, 1991 – 1999 and 1984 – 1999. Burned areas were removed from the land-cover maps for the empirical modelling (presented in section 3.3) as the focus was on the dynamics of LUCC rather than wildfire occurrence and spread (resulting in 884,501 pixels).

2.5.5.3 Climatic Variables

Maps of mean annual rainfall and temperature were derived from data collected at 21 weather stations located both inside and outside SPA 56 (see Figure 2.11a). Climate data were collected for years 1956 – 1995, though data are incomplete for some stations. Data were interpolated across the regions using the ancillary variables of altitude, slope, aspect, latitude and longitude (see Romero-Calcerrada 2000 for details). Rainfall data were binned into classes (see Figure 2.11b), with the median value of each bin used for pixel values here. Temperature values were rounded to the nearest integer value across the range shown in the legend for Figure 2.11c.

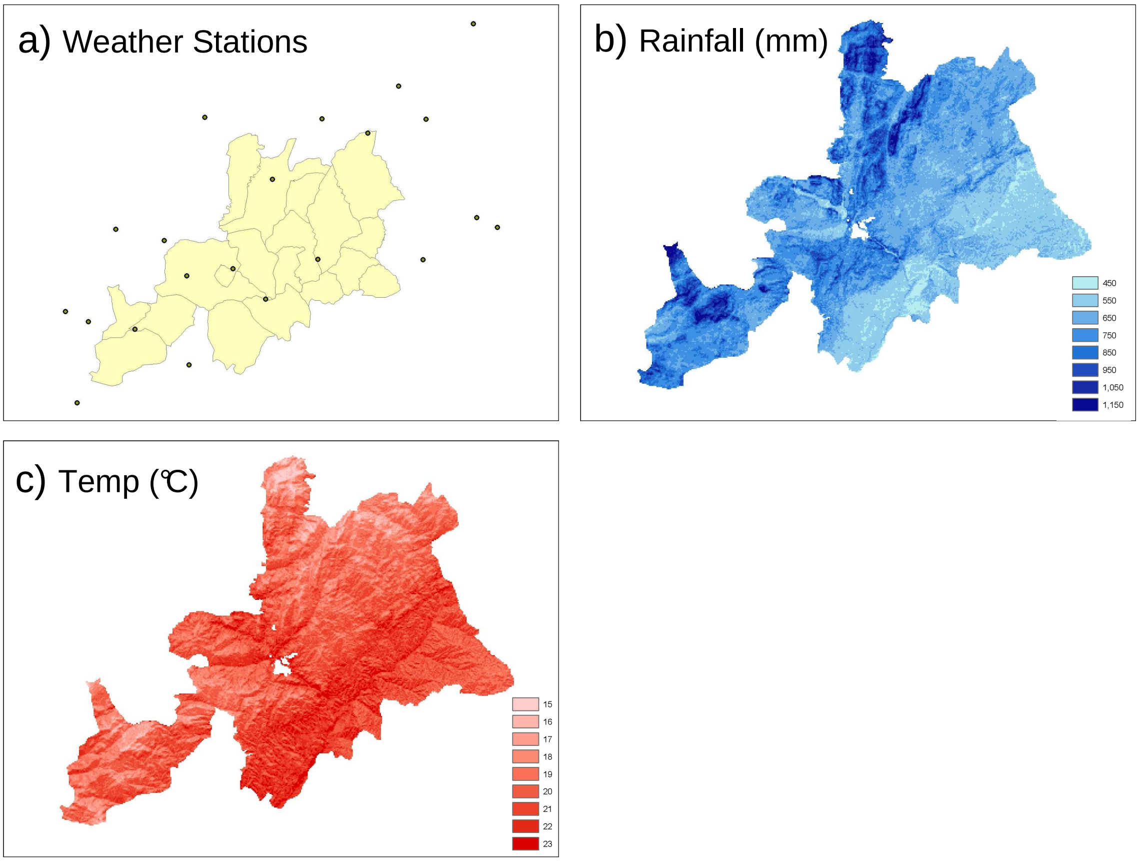

Figure 2.11 SPA 56 climate maps. a) weather station locations, b) mean annual rainfall, c) mean annual temperature Climate data were interpolated from the weather station data and rounded to the nearer integer value.

2.5.5.4 Topographic Variables

Aspect and slope surfaces were derived from a Digital Elevation Model (DEM) of the Consejería de Política Territorial de la Comunidad de Madrid (see Romero-Calcerrada 2000). The DEM had an original spatial resolution (grain) of 20 m but this was degraded to a 30 m resolution (in the IDRISI GIS package) to match the resolution of the other data layers. From this DEM, maps of slope and aspect were derived. Integer slope angles were derived for the range shown by the legend in Figure 2.12a. Integer aspect values (degrees from 0° = north-facing, through 90° = east-facing and 270° = west-facing) were derived for the range of values shown by the legend in Figure 2.12b. Geomorphological change is not considered in any of the models presented here and thus these topographic variables were assumed to remain constant throughout model runs.

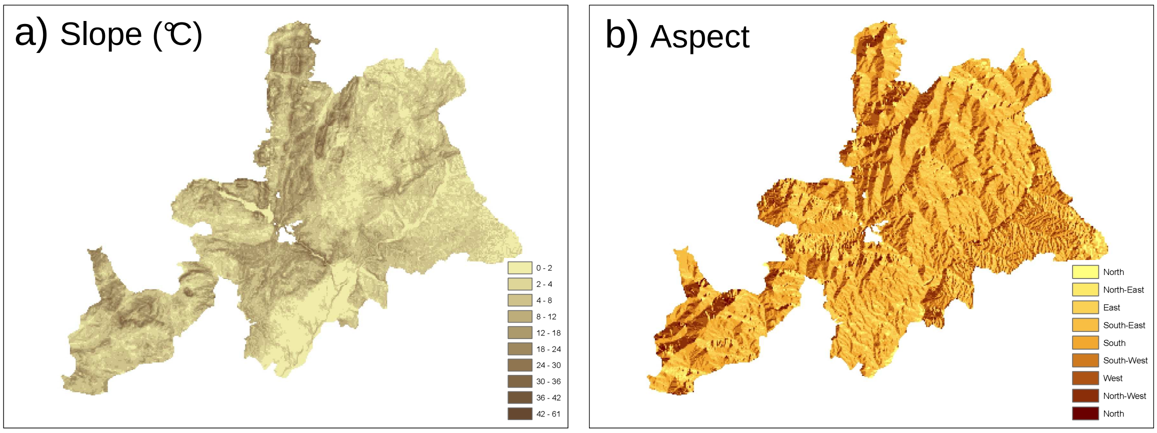

Figure 2.12 SPA 56 topographical maps. a) Slope, b) Aspect Data were derived from a DEM and rounded to the nearest integer values. Data are classified for display purposes only – individual values were used in analysis.

2.5.5.5 Land Capability

A qualitative land evaluation technique, based on the MicroLEIS expert system (de la Rosa 1990), was used to determine land capability. In this context, 'land capability' is an assessment of the suitability of land for sustainable agricultural development or use. Land capability is thus considered along a continuum of decreasing resource use from arable agricultural, through pastoral agriculture and forestry to 'un-useable'. A land evaluation method such as this is useful for locating and identifying areas most appropriate for human land-use and potential maximum level at which the land can be sustainably used. The FAO has recognised land evaluation in this manner as having an important role in land-use planning (e.g. FAO 1981, 1985).

The land capability map for SPA 56 was produced by Romero-Calcerrada (2000) by synthesizing four factors – slope, soil type, erosion risk and bio-climatic deficiency. The MicroLEIS software (de la Rosa 1990) was implemented in a GIS, to score each variable on a scale from one (excellent agricultural land capability) to four (marginal land). Slope was considered to be a variable restricting land capability and scored accordingly. Erosion risk assessed soils' vulnerability or resistance to degradation, and was primarily based upon the proportion of vegetation cover. The soil factor accounted for useful soil depth, texture (percentage of sand, mud and clay), stoniness, and drainage. The bio-climatic deficiency map combined two variables; hydric deficiency and frost risk. Hydric deficiency was calculated as total annual rainfall divided by the total annual Potential Evapo-Transpiration. Frost risk was assessed by the number of months the mean diurnal minimum temperature was below 6 ºC. These four ‘limiting-factors’ maps were synthesized via a basic weighted overlay to produce the land capability map (*LCAP*). Urban/residential areas, quarries, reservoirs and other human-made areas were not considered by the land-use map and were coded as ‘No Data’. The resulting land capability map is shown in Figure 2.13. Each map pixel has an attribute referring to the class of land capability it belongs to (S1 = Excellent Potential, S2 = Good, S3 = Moderate or N = Marginal), plus a sub-class identifier(s), depending on the combination of limiting factors that impact the capability negatively (t = topographic; r = erosion risk; l = soil; b = bio-climatic deficiency). Each class was then converted to a numerical value on a scale from 0.0 to 2.0 to represent land capability.

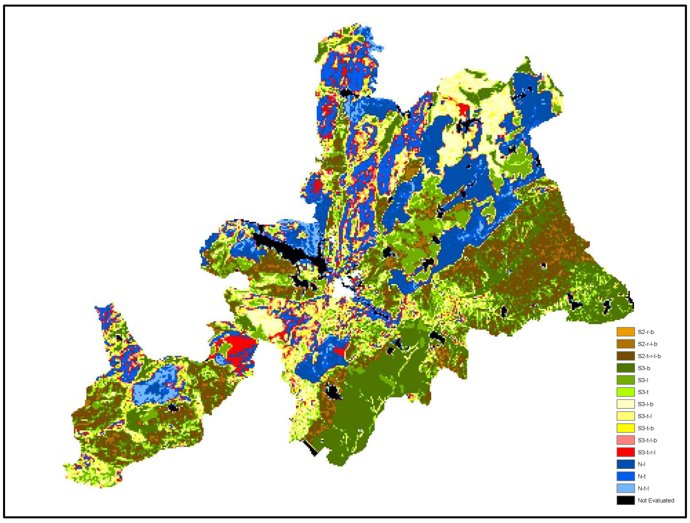

Figure 2.13 SPA 56 Land Capability map. Refer to text for an explanation of classified land capability type codes.

2.5.5.6 Spatial Landscape Data

Weaver and Perera (2004) highlight the problems of ignoring spatial dependence in empirical models of ecological change – by ignoring spatial dependence, empirical models typically fail to accurately reproduce a given landscape’s spatial structure. For each of the three land-cover maps, maps of Modal Neighbour (Moore neighbourhood), Patch Edge and ‘Distance to Patch Edge’ were derived. Pixels were defined as patch edge if any one of the eight surrounding pixels was not of an identical land-cover type. If all surrounding pixels are of the same land-cover type as the central pixel, that given pixel was designated as being in a patch. Maps were also created to quantify the distance from a given pixel to the nearest ‘urban’ pixel (*DURB*), nearest road (*DROAD*), and nearest watercourse (both ephemeral and perennial – *DRIV*). These maps were derived in the ArcMap GIS (ESRI 2002) based on maps for 1999, and were assumed to remain constant throughout the study period.

2.5.5.7 Social Data Variables

Five social variables are used in the empirical models – agricultural sector employment, mean farmer age, net migration, and population density. These variables were recorded and aggregated at the municipality level and then converted to raster maps with a spatial resolution of 30 m (e.g. see Figure 2.14).

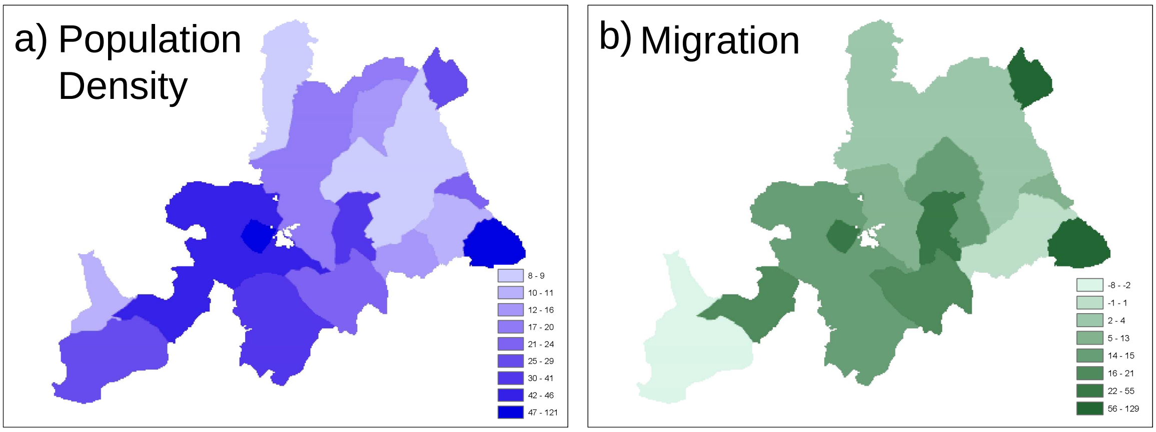

Figure 2.14 Example SPA 56 maps of social variables. a) Population Density (people/km^2), and b) Net Migration (people/yr).

2.5.5 Summary

The EU Special Protection Area number 56, 'Encinares del río Alberche y Cofio' is an area of outstanding natural beauty and contains several endangered bird and mammal species. SPA 56 contains multiple land-uses and-covers – stands of pine and oak, *dehesa*, and pasture and arable farmland. However, recent socio-economic changes (e.g. ageing farming population) have resulted in the replacement of pasture by scrubland due to land abandonment (see section 2.5.4 and Romero-Calcerrada and Perry 2004). The implications of these trends are unclear due to the feedbacks between ecological and human patterns and processes (noted above sections 2.2 and 2.4).

2.6 SUMMARY

This chapter has presented an introduction to the causes, processes, patterns and potential impacts of LUCC in the Mediterranean Basin, with specific regard to Spain and SPA 56. SPA 56 is characterised by two main types of vegetation (pine and oak) whose dynamics and succession-type behaviour are influenced by frequent disturbance. The main ecological disturbances in SPA 56 are fire, grazing and other human agricultural practices. Furthermore, the spatial nature of Mediterranean vegetation dynamics, succession and disturbance has been demonstrated. The long influence of humans (agriculture arrived in Spain *circa* 6,000 BCE) means that such an ecological system cannot be considered independently of human activity. These processes highlight recent shifts in ecological thinking, which now emphasises the need to consider disturbance and the feedbacks between spatial patterns and processes as central to the structure and functioning of ecological systems.

Grove and Rackham (2001) suggest that future threats to the sustainability and biodiversity of Mediterranean landscapes are likely to come from technological and socio-economic changes rather than climatic ones. Whether this proves to be a correct assertion remains to be seen, but LUCC has certainly been observed recently (1984 – 1999) in SPA 56, and suggests that agricultural abandonment has lead to commensurate increases in scrubland and decreases in pasture. As MacDonald *et al.* (2000 p.52) succinctly summarised it, "abandonment in its extreme form is associated with an inability to adapt farming and land management to social and economic pressures". In SPA 56 an ageing and diminishing farming population is driving the decline of the agricultural sector. Improvements in communications with the city of Madrid and the planning and building of new homes in SPA 56 suggest that these socio-economic and agricultural trends are unlikely to be reversed soon. The challenge is to manage these changes such that SPA 56 remains viable for population by both humans and the endangered species it is designated as being a haven for. Such management will have the greatest chance of succeeding if it is based on sound understanding of the ecological processes occurring in SPA 56, and does not preclude consideration of the integral nature of humans within them.

In general, previous studies suggest that the land-cover of Mediterranean landscape has become more homogenous, due to a combination of LUCC (predominantly agricultural abandonment) and increases in wildfire risk and occurrence. However, because pattern-process interactions between wildfire, vegetation dynamics and human land-use are unclear due to non-linear feedbacks and historical landscape contingency, a lack of consensus remains regarding the exact causes of homogenisation its implications for the future. This thesis aims to examine the impacts of potential future land-use/cover change upon wildfire regimes in SPA 56. The lengthy human presence in Mediterranean landscapes has resulted in systems in which ecological and cultural change are highly integrated and interactive. Human and ecological processes are perpetually re-shaping these landscapes, and in turn, themselves. The following chapter examines previous approaches and techniques that have been used to landscape change with varying levels of representation of human activity.

CHAPTER THREE

LUCC MODELLING

3.1 INTRODUCTION

From the discussion in the previous chapter it is apparent that Spanish landscapes exist as part of an integrated system of humans, flora, fauna and disturbance (predominantly fire and grazing). Humans and their cultural, social and economic activities are woven into the ecological fabric that covers the landscape. Changes in human behaviour are likely to precipitate changes in ecological structure and function. Feedbacks between and within socio-economic and ecological processes, make the direction these changes might take unclear. Odum (1992) proposed that in order to develop sustainable environmental system humans must acknowledge that they are part of that system and examine it as such. To examine potential ecological changes in the landscapes described above, where humans are so pervasive and where land is used for multiple purposes, socio-economic changes cannot be ignored and need to be addressed.

LUCC, its processes and impacts, occur over extended space and time extents from local to global and from years to centuries. The large time and space extents involved make empirical experimentation on landscape function and change virtually impossible because of logistical, political and financial constraints. Models ranging from purely statistical to process-oriented simulation modelling approaches can help to overcome these problems. However, integrating human activity into these models has provided modellers with some challenging methodological problems. It is widely acknowledged that the prediction of future human activity quantitatively at specific points in time or space with any useful degree of accuracy or confidence across long time intervals is an inscrutable problem (e.g. Allen and Strathern 2005) But if advances in understanding or estimation of future LUCC are to be made, some attempt to assess potential future human activity must be made. One problem is how best to do this. A second is then how to integrate this assessment with knowledge of environmental systems in order to gauge most accurately how future landscapes will be structured and function as a response to human activity. A third, and following, problem then remains regarding how to assess the reliability of the knowledge gained from these procedures. Modelling provides a means to overcome logistical and political constrainsts of studying these systems where direct empirical experimentation is very difficult, and has frequently been used in LUCC studies. In modelling terms, the problems listed above can be summarised as the need to find the most appropriate way to model the interaction of human and environmental systems and then how to assess appropriateness of this model and the procedure of building it.

In this chapter modelling methods that have been employed previously to examine landscape change are reviewed and synthesised to establish the most appropriate methodological approach to model the particular LUCC processes occurring in SPA 56. Specific modelling techniques are classed into two broad groups – empirical modelling and simulation modelling. These broad groups are considered in turn and an example of the former is presented for SPA 56. This example is used as the basis for discussion of the merits of the different approaches (empirical and simulation), how they complement one another, and how they might be used in this particular case study. Potential implications and questions that are raised regarding the validity, credibility and confidence of the results that arise from the proposed methodology for this case study are also briefly outlined ready to be addressed later in the thesis.

3.2 Empirical Modelling

Empirically-based models are 'black box' models – output results are produced from input data with no explicit regard for process. What happens inside the box is unknown (it is black and we cannot 'see in') as are the reasons why the output was produced from the input. Thus, they contrast with process-based, 'white box' models that (theoretically) consider all processes acting in the system under scrutiny. In reality white box models do not exist, as it is generally unfeasible to incorporate all processes (because models are simplifications of reality) and in any event, all acting processes may not be known. Therefore, models not defined as 'black' are usually some shade of 'grey' (in the range between purely 'black' or 'white'). Empirical models use previously observed relationships between variables in a system to predict future systems states according to the current state of system variables (i.e. the data about the system). The obvious limitation of such an approach is the model's inability to represent systems that are non-stationary, i.e. systems in which relationships between variables are changing through time. This assumption of stationarity rarely holds (Turner 1988). Applying empirical models to systems other than the particular one for which they were initially derived, and across scales, is also problematic. Two empirical approaches exist to forecast spatially explicit LUCC – transition- and regression-based models.

3.2.1 Transition-Based Models

Transition (matrix) models use previously observed land-use changes to derive probabilities for the likelihood of future changes. A transition matrix is produced that details probabilities of change for all possible state changes. For example, for a landscape with three land-uses, *i*, *j*, and *k*, probabilities are derived for all nine possible state transitions (*i* to *i*, *i* to *j*, *i* to *k*, *j* to *i*, *j* to *j*, *j* to *k*, *k* to *i*, *k* to *j* and *k* to *k*) according to the number of times each state change occurred in the last time interval. This transition matrix *P* is then multiplied by a vector x_t of current abundances for each state (land-use) to give the landscape abundances for the next time interval:

$$x_{t+1} = P.x_t \qquad \text{Eq. 3.1}$$

We can thus predict landscape composition at some period *n* time steps in the future as:

$$x_{t+n} = P.x_t{}^n \qquad \text{Eq. 3.2}$$

If Eq. 3.2 is applied iteratively the matrix projections will stabilise at the stable landscape land-cover distribution. This stable distribution will be the same irrespective of the initial vector. Such a LUCC transition matrix modelling exercise has already been undertaken in SPA 56 for the period 1984 – 1999 (Romero-Calcerrada and Perry 2004). The stationarity assumption did not hold for SPA 56 during 1984 – 1999. However, non-stationarity can be turned to an investigators advantage – any prediction of a currently observed landscape from transition probabilities calculated from a previous change can be used to examine how transition rates have changed (e.g. Chust *et al.* 1999, Romero-Calcerrada and Perry 2004).

This technique can also be used in a spatially-explicit manner. Turner (1988) highlights that the probability of a patch of land to change state is not solely determined by the patch's previous state, but is also influenced by surrounding cells. By incorporating this influence Turner produced a spatial transition matrix model for a Georgian, USA, county for the interval 1942 – 1980. Jenerette and Wu (2001) produced a similar spatial transition matrix model to examine LUC in central Arizona-Phoenix, USA, for the interval 1975 – 1995. These models are simple and quick to create and use, providing broad insight into current LUCC trajectories and their change, but allow little in the way of causal interpretation.

3.2.2 Regression-Based Models

Regression-based models of LUCC derive relationships between observed change (the dependent variable) and the values of physical, economic or social indicators (the independent variables) such as elevation, mean annual yield, or population density, at those locations of change. These relationships are then used to project future land-use/cover from current independent variable values. Because LUCC is usually a discrete change (e.g. from agricultural land to scrubland), logistic regression is an appropriate statistical model to use (Trexler and Travis 1993). This technique has been widely used in all manner of LUCC studies in many different environments with many different types of independent variable, ranging across: aspect, slope, soil pH, rainfall; intensity of cattle/goat grazing, level of farm mechanisation, agricultural value; ethnic composition; proximity to nearest urban centre/market etc., and characteristics or variable values in neighbouring cells (Bockstael 1996, Chomitz and Gray 1996, Turner *et al.* 1996, Wear and Bolstad 1998, Carmel *et al.* 2001, Schneider and Pontius 2001, Serneels and Lambin 2001, Muller and Zeller 2002, Soares *et al.* 2002).

Turner *et al.* (1996) utilised multinomial logit models to examine differences in land-cover transitions between land owner types in the southern Appalachians and the Olympic Peninsula, USA. Variables used included slope, elevation, distance to roads, and population density, and the authors suggest that their method is applicable to other landscapes. Wear and Bolstad (1998) performed a similar modelling exercise on the southern Appalachians, and found that their models correctly predicted between 68% and 89% of land-cover proportions. As with transition-based models, and empirically-based models in general, these models are useful to predict potential LUCC and explore the most important variables or important scales of study, but are of less use for explaining *why* changes are occurring. Furthermore, the presence of non-stationarity in a system (landscape) severely inhibits the ability of a model based on observed changes (and associated drivers of change) to accurately predict a future state. To explain the importance of relationships amongst independent variables and upon the dependent variable, a technique such as hierarchical partitioning is more useful (Chevan and Sutherland 1991, Mac Nally 1996, 2000, Millington *et al.* 2007).

3.3 EMPIRICAL MODELLING OF SPA 56

3.3.1 Introduction

To examine the potential of one of these empirical methods to model LUCC, incorporating socio-economic data as potentially important independent variables, an example of the application of the multinomial logistic regression model is presented. This exercise (much of which is published as Millington *et al.* 2007) also allows an initial examination of the available data to examine how data at different scales of resolution/aggregation influence LUCC modelling efforts. Carmel *et al.* (2001) used a logistic regression model in an Israeli landscape, attaining between 60 – 90% accuracy when predicting total landscape proportions of three Mediterranean land-uses. On a pixel-by-pixel basis their model successfully predicted the correct transition of slightly fewer than 55% of all pixels. From the literature it would seem that Carmel *et al.* (2001) are the only modellers to use logistic regression in a Mediterranean-type environment. Further, they modelled changes between just three land-cover types (tree cover, herbaceous cover, shrub cover). In contrast, models will be calibrated and used here to predict landscapes with 10 land-cover types.

3.3.2 Methodology

Generalized Linear Models (GLMs) are able to model non-normally distributed dependent variables, and thus overcome the problems of the assumptions of regular linear regression models (Venables and Ripley 2002). Quinn and Keough (2002) note that GLMs have three components: 1) a response (dependent) variable with a population distribution belonging to the exponential family, 2) the predictor (independent) variables, and 3) a 'link function' that links 1) and 2). For example, the logistic model (for multiple predictors) is:

$$\pi(x) = \frac{e^{\alpha + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_i x_i}}{1 + e^{\alpha + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_i x_i}} \qquad \text{Eq. 3.3}$$

where $\pi(x)$ is the probability that the response variable $y = 1$, α is the equation constant, and β_i is the coefficient of predictor variable x_i. Thus, the binary response variable is modelled as an odds ratio – the probability that y will be a member of one class relative to the other class (Trexler and Travis 1993). Rather than model Eq. 3.3 directly, the link function $g(x)$ allows the response variable to be modelled as:

$$g(x) = \alpha + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_i x_i \qquad \text{Eq. 3.4}$$

For a binomial response variable the logistic (logit) link is the natural logarithm of the odds ratio:

$$g(x) = \ln\left[\frac{\pi(x)}{1-\pi(x)}\right] \qquad \text{Eq. 3.5}$$

Thus, the logistic regression model is more easily solved as:

$$\ln\left[\frac{\pi(x)}{1-\pi(x)}\right] = \alpha + \beta_1 x_1 + \beta_2 x_2 + \ldots \beta_i x_i \qquad \text{Eq. 3.6}$$

The equation intercept (constant) α and variable coefficients β_i are estimated from calibration data using maximum likelihood techniques. Once these are known, the model can be applied to estimate future states based on an alternative data set considering the same variables.

This method can be extended for response variables with more than two categories by using the Multinomial Logit Model (MNLM) as a probability model to estimate the category of the response variable given the predictor variables. Consider *y* as a dependent variable that can potentially take one of *J* nominal categories, and that these categories are numbered from 1 to *J* (but not assumed to be ordered). Let

$$\mathrm{X}\Phi_m = \alpha + x_1\beta_1 + x_2\beta_2 + \ldots x_i\beta_i \qquad \text{Eq. 3.7}$$

where *m* is also a category of *y*. The probability of observing each category *m* can now be calculated (when $\Pr(y = m|\mathrm{X})$ is the probability of observing *m* given X, the set of predictor variables):

$$\Pr(y = m|\mathrm{X}) = \frac{1}{1+\sum_{j=2}^{J} e^{\mathrm{X}\Phi_j}} \text{ for } m = 1 \qquad \text{Eq. 3.8}$$

$$\Pr(y = m|\mathrm{X}) = \frac{e^{\mathrm{X}\Phi_m}}{1+\sum_{j=2}^{J} e^{\mathrm{X}\Phi_j}} \text{ for } m > 1 \qquad \text{Eq. 3.9}$$

To test the importance of predictor variables in these models statistically, the Likelihood Ratio (LR) statistic is used. The LR statistic is calculated by comparing the Residual Deviance (RD) of the full model (containing all variables) against a reduced model (full model minus the variable in question):

$$\text{LR Statistic} = -2(\textit{log-likelihood}_{reduced} - \textit{log-likelihood}_{full}) \qquad \text{Eq. 3.10}$$

The resulting statistic is compared to the χ^2 distribution, to examine whether the variable has a significant effect on the response variable (i.e. the variable coefficient is statistically different from zero), with:

$$\text{Degrees of Freedom} = (J - 1)(N_{Xfull} - N_{Xreduced}) \qquad \text{Eq. 3.11}$$

where N_X is the number of predictor variables.

Lennon (2000) highlight the pitfalls of not considering the spatial autocorrelation of data used in multiple regression analysis. Spatial autocorrelation was tested here for the response variable (land-cover) using a row-standardised Moran's *I* test (see Cliff and Ord 1973). The results indicated that autocorrelation decreased monotonically above a lag of eight map pixels (~ 240 m). Therefore, data were sampled at every 10th pixel in both *i* and *j* directions, resulting in 8,855 data points being used for model calibration.

Coefficients for the 12 predictor variables (see Table 3.1 and section 2.5.5) were estimated from the calibration data using the 'multinom' function in R (Venables and Ripley 2002). The Likelihood Ratio statistic was also calculated in R. A C++ program was written to apply the produced coefficient estimates to predict change/no change and future land-cover states based on equations 3.8 and 3.9. A second C++ program was written to produce error matrices comparing predicted versus observed landscape maps. Land-cover was projected into the future (the year 2014) by using values of the model coefficients, estimated for the period 1984 – 1999, with values of the predictor data as at 1999.

A pixel-by-pixel comparison of maps is used to assess the proportion of pixels in the predicted landscape whose composition was predicted correctly. The performance of groups of related variables was also assessed. Social, physical, and spatial variables

Table 3.1 Predictor variables (maps) used for empirical modelling. More details on this data is provided in section 2.5.5

Variable	*Unit of measurement*	*Year of data*
Socio-economic data		
AGRWK -Agricultural Workers	Percentage of population	1986, 1991
FMAGE - Mean Farmer Age	Years	1982, 1989
MIG - Migration	Persons	1988, 1991
PDENS - Population Density	Inhabitants/km^2	1985, 1991
Biophysical data		
ASPECT - Aspect	N, NE, E, SE, S, SW, S, NW	1995
LCAP - Land Capability	Ranked classification	1997
LC - Land-cover	Land-use	1984, 1991, 1999
TEMP - Temperature	Mean annual ℃	1965-1995
Spatial Data		
D_PE - Distance to Patch Edge	Metres	1984, 1991
D_RIV - Distance to River	Metres	1995
D_ROAD - Distance to Road	Metres	1995
D_URB - Distance to Urban Area	Metres	1995

were grouped and models run to examine how the three broad types of data influenced land-cover change (see Table 3.2 for the definitions of these models). Pontius *et al.* (2004) bemoan the inability of LUCC models to improve prediction above and beyond the predictive capability of the original landscape (i.e. if we simply assume no change). They stress that this fact should not be overlooked when validating or evaluating models. Thus, when evaluating the models, pixel-by-pixel accuracies are compared with the pixel-by-pixel accuracy of the corresponding *null model*. The null model assumes no change and is taken as simply the previous landscape upon which the model is based (i.e. for a 1984 – 1991 model the 1984 landscape map is the null model with which to compare other models). Results are presented for non-calibration pixels only, resulting in 875,646 pixels for evaluation.

Table 3.2 Model definitions for empirical models of similar variable types. Abbreviations are listed in Table 3.1.

Model	*Variables*
SocMod	AGRWK, MIG, PDENS, FAGE
PhysMod	ASPECT, LCAP, LC, TEMP
SpatMod	DROAD, DRIV, DURB, PD
PhysMod2	ASPECT, LCAP, TEMP

3.3.3 Results

All predictor variables used in the multinomial logistic models were found to be statistically significant according to the LR statistic ($p < 0.01$). On the basis of the pixel-by-pixel accuracy models achieved up to 57% accuracy (Table 3.3). These values are comparable with those of Carmel *et al.* (2001) who considered a structurally simpler landscape (i.e. fewer land-cover classes and interacting processes driving LUCC). As Pontius *et al.* (2004) have observed frequently in the past, null models were found to perform best here for landscape proportion for all time periods, and performed best on a pixel-by-pixel basis for 1984 – 1991 and 1991 – 1999, but not 1984 – 1999 (Table 3.3). Thus, while the study area landscape is dynamic (Romero-Calcerrada and Perry 2004) it is not so dynamic that over a 7 – 8 year period more landscape pixels change than remain in their original state. This again indicates the importance of the previous land-cover on future land-cover and the trajectory or pathway change might follow. However, it suggests that over longer time periods (i.e. for 1984 – 1999) this influence would appear to wane, and regression modelling becomes more useful compared to simply using the null model of pure persistence.

Table 3.3 Empirical models' prediction accuracy for individual pixel locations. All values are percentages. Differences are between the *full model* and the reduced models, full model differences are comparisons with the *null model*. Null models outperform all models for 1984 – 1991 and 1991 – 1999, but not 1984 – 1999 models.

Model	*Time Period*					
	1984 – 1991		*1991 – 1999*		*1984 – 1999*	
	Accuracy	*Difference*	*Accuracy*	*Difference*	*Accuracy*	*Difference*
Null Model	55.6		56.97		49.68	
Full Model	47.8	7.9	54.29	2.7	51.93	-2.2
SocMod	28.75	19.0	40.45	13.8	39.02	12.9
PhysMod	45.20	2.6	51.77	2.5	48.88	3.0
SpatMod	32.63	15.1	38.07	16.2	40.34	11.6
PhysMod2	35.17	12.6	38.21	16.1	38.21	13.7

Of the models that consider alike sets of variables, *PhysMod* was the only model to perform with any degree of accuracy (Table 3.3). However, when the original land-cover variable is removed from the variable set, performance drops to around the same level as the *SocMod* and *SpatMod* variables models. According to the LR stat, all variables are statistically significant across all time periods. This fact highlights the need for, and to some extent justifies, an approach that examines the contribution of individual variables in a model to predictive success by comparing the accuracy of the model with the accuracy of a reduced version (i.e. with models with variables systematically removed). If purely statistical techniques, such as the LR statistic, are used alone distinction between the variables importance (in terms of 'usefulness' to model future LUCC) is very difficult. As a further form of less quantitative analysis, the maps produced by the regression models were examined visually in terms of their configurational patterns and potential spatial artefacts produced by the models (Figure 3.1).

3.3.4 Discussion

The abject failure of the socio-economic variables (*SocMod*) to predict land-cover adequately is obvious, and the spatial aggregation of the data becomes apparent when the borders of the municipalities are overlaid (Figure 3.2a). The influence of the municipality aggregated data is also apparent if the full model map is studied more closely (Figure 3.2b). Boundary effects influencing the prediction of Holm Oak and Holm Oak with Pasture are distinct, predicted in one municipality but not the adjacent neighbouring municipality. These observations highlight the problems of using aggregated socio-economic variables (in this case at the municipality level) in regression techniques that examine LUCC over local to regional extents. The literature (e.g. Caballero 2001, Hoggart and Paniagua 2001, MAPA 2003, Mazzoleni *et al.* 2004, Romero-Calcerrada and Perry 2004) and local experts' understanding of the type of agricultural LUCC occurring in the landscape suggest that the level of agricultural employment and the ageing and diminishing of the agricultural population of the study area are driving the observed change (from a human-activity perspective). Furthermore, using the hierarchical partitioning approach based on regression models of SPA 56, Millington *et al.* (2007) found that a land-cover classification aggregating the study area land-cover into 10 classes proved more revealing in process terms than a four class land-cover classification. Selecting variables solely on the basis of this understanding failed to provide adequate predictive statistical models of change. Variables at finer scales, such as the age of individual farmers, the structure of individual family farm households, and the income of individual farms, drive changes in individual land parcels. Data at the farm level would be more suitable for a predictive regression model in this area as decision-making influences would be represented more explicitly. There has been some progress on incorporating this type of data into regression models recently (e.g. Pan *et al*. 2004, Overmars and Verburg 2005, Pan and Bilsborrow 2005). These findings suggest the need to represent farm households as decision-making agents driving land-use change in a simulation model of LUCC (see chapter five).

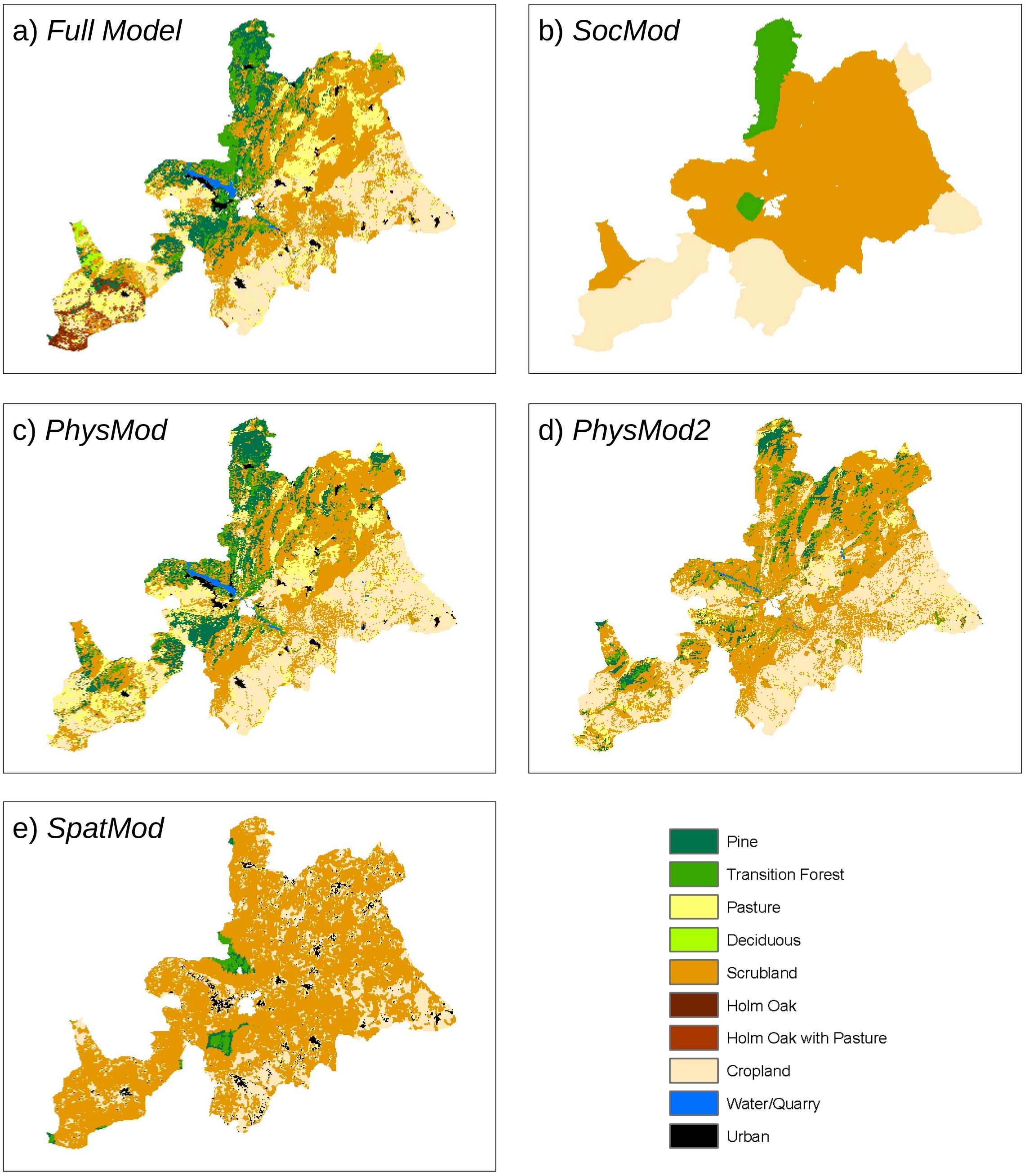

Figure 3.1 Multinomial model output maps. a) *Full Model*, b) SocMod, c) PhysMod, d) PhysMod2, and e) SpatMod. Maps are for 1991, calibrated over the interval 1984 – 1991. Definitions of each model are presented in Table 3.2.

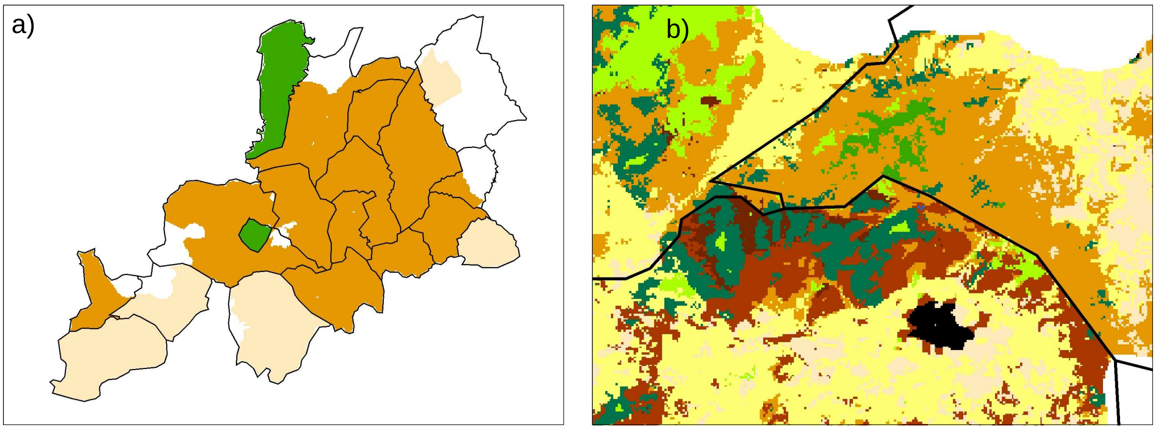

Figure 3.2 Example of the influence of municipality-aggregated data on model output. a) *SocMod* and b) *Full Model*. Output from the *SocMod* clearly follows the boundaries of the municipalities with no variation within municipalities. While the full model shows variation of land-cover within municipalities, boundary effects are evident; in b) Holm Oak and Holm Oak and Pasture are present in the south-western municipality but abruptly cease at the border with the north-eastern municipality.

It should be noted that while the models do not contain any explicitly 'economic' variables (e.g. total annual agricultural subsidy received etc.) they do implicitly assume economic decision-making via the inclusion of the variables 'distance to road', 'distance to urban' (i.e. the transportation costs to product and labour markets of the von Thünen model – see section 5.2) and 'land capability' (i.e. farmers will farm the land that provides greatest yields). Many farms in SPA 56 are small (77% ≤ 5 ha. in 1999, INE 2005), are run by farmers past retirement age (49% older than 65 in 1999, INE 2005) and operate at an economic loss. Often these farms are not the primary income for a family but instead provide supplementary income through seasonal crops (see section 5.4). In such traditional Mediterranean agricultural landscapes conventional accounting measures have been found to only partially account for total environmental goods and services (Campos and Caparros 2006). However, omitting explicitly economic variables, and the lack of farm-level data, is a drawback of the implementation of the methods presented here. Specifically, omitted variable bias – the omission of variables that have an influence on the dependent variable and are correlated with other independent variables, producing error in parameter estimates – is potentially present in the models presented here. Unfortunately, as one author has recently highlighted, this problem is logically unavoidable and cannot be solved simply by adding increasing numbers of predictor variables, but can be ameliorated by ensuring appropriate research design (Clarke 2005). The availability and analysis of data at scales that match causal processes is vital to the performance of the regression modelling techniques employed here and to ensure an appropriate research design (Millington *et al.* 2007).

Despite these data drawbacks, the predictive models presented here performed comparably with a previous regression modelling study of a Mediterranean landscape (Carmel *et al.* 2001). Compared to simpler binary regression models of LUCC (i.e. models of change/no change) multinomial models are able to project transitions between particular land-covers. This attribute allows them to be used to produce maps of potential land-cover by using values of the model coefficients estimated from an observed period with values of the predictor data from the end of that period. Projections for land-cover in 2014 (from a model for the observed period 1984 – 1999) suggest that the current shifts from agricultural land-uses to scrubland will continue (as models based on the assumption of stationarity would expect – see below). Such changes may be favourable for the endangered Iberian lynx (*Lynx pardinus*). Fernandez and Palomares (2000) suggest that increased scrubland will improve lynx habitat, and Fernandez *et al.* (2003) found that one of the lynx's key prey (the rabbit, *Oryctolagus cuniculus*) were four times more likely to be found in dense scrub than in other vegetation. The prospects for avifauna seem to vary by species, however. Preiss *et al.* (1997) found that following abandonment in southern France habitat for birds that prefer open landscapes (e.g. Wood lark, *Lulula arborea*) decreased, while forest-type habitat (preferred by species like the Great tit *Parus major*) increased. Corresponding changes in these bird populations were observed. Suarez-Seoane *et al.* (2002) confirmed that birds preferring scrubland may benefit initially following agricultural abandonment but may eventually suffer as land-cover progresses toward woodlands. The effects of wildfire on vegetation dynamics may mitigate this situation however, and Millington (2005) suggested the risk of wildfire in SPA 56 will increase if the LUCC projected here does occur. The statistical modelling approach taken here is unable to account for these potential landscape dynamics.

The quality of these models' projections is highly dependent upon both the performance of the model over the observed period, and the important assumption that the observed causal processes used to develop the model are stationary. As noted above (section 3.2), Romero-Calcerrada and Perry (2004) found that the stationarity assumption did not hold for 1984 – 1999, but the reasons for this were not examined. The stationarity assumption in this case is perhaps more likely to be invalid from a socio-economic than a perspective biophysical. While biophysical processes may be assumed to be relatively constant over decadal timescales (consideration of climatic change aside) this will not be the case for many socio-economic processes. With regard to SPA 56 for example, the recent expansion of the European Union to 25 countries, and the consequent likely restructuring of the Common Agricultural Policy (CAP), will lead to shifts in the political and economic forces driving LUCC in the region. Where socio-economic factors are important components of landscape change, and where these are likely to change during the projected period due to social, economic, political or technological innovation, regression models will be of limited use for future projections of LUCC and subsequent ecological interpretation.

3.3.5 Summary

The predictive performance of the empirical models examined here proved comparable with those of Carmel *et al.* (2001) who considered a structurally simpler (i.e. three land-use) Mediterranean landscape. Models generally did not perform as well as the null model of no change (i.e. if no change is assumed) over shorter time intervals, but performed better when a longer interval (i.e. 15 years) was examined. The results for the specific variable 'types' (i.e. *PhysMod*, *SocMod*, and *SpatMod*) suggest that the physical variables are most important for predicting future land-cover accurately. However, this is likely due to the disparity in resolution at which the data has been aggregated and any model that aims to represent spatial change in this study area will need to consider socio-economic processes at a sub-municipality level. Finally, it is suggested that the empirical approach presented is not adequate to represent the landscape dynamics in areas where processes are not stationary (for example, changing political and cultural norms, climatic change) and involve spatial feedbacks between pattern and process (for example, wildfire).

3.4 Simulation Modelling

3.4.1 Introduction

The primary contemporary alternative to the empirical modelling methods described above are simulation modelling techniques. When a problem is not analytically tractable (i.e. closed form equations cannot be written down and are non-integrable) simulation models may be used to represent a system by making certain approximations and idealisations (Winsberg 1999). When attempting to mimic a 'real world' system, simulation modelling has become the *sine qua non* (Peck 2004). This may have become the case since, as Winsberg (2001) highlights, simulation modelling can be used when data are sparse. Spatially-explicit simulation models of LUCC have been used since the 1970s and have dramatically increased in use recently with the growth in computing power available. These advances mean that simulation modelling is now one of the most powerful tools for environmental scientists investigating the interaction(s) between the environment, ecosystems and human activity (Mulligan and Wainwright 2004).

Simulation modelling overcomes many of the problems associated with the large time and space extents involved in landscapes studies that make empirical experimentation in the field virtually impossible because of logistic, political and financial constraints. Further, experimenting with simulation models allows experiments and scenarios to be run and tested that would not be possible in real environments and landscapes. Simulation models of potential LUCC have become a useful tool not only to improve understanding, but also to meet the demands of land managers by providing a means to examine and visualise the results of alternative management strategies (Veldkamp and Lambin 2001). More specifically, Zavala and Zea (2004) claim that the development of predictive and explanatory models of Mediterranean forest and vegetation dynamics are critical to provide scientifically-based tools to help land-use managers evaluate their options. However, simulation modelling is an iterative process in which learning takes place throughout and which produces models (and tools) that are open to continual improvement. The modelling process iterates through the stages of setting and refining model purpose, through model conceptualisation and construction to testing and analysis and is discussed further in chapters seven, eight and nine.

Current spatially-explicit simulation modelling techniques allow the spatial and temporal examination of the interaction of numerous variables, sensitivity analyses of specific variables, and projection of multiple different potential future landscapes (Baker 1989, Veldkamp and Lambin 2001). This functionality facilitates the evaluation of proposed alternative monitoring and management schemes, the identification of key drivers of change, and potentially improved understanding of the interaction(s) between different types of variable and processes both spatially and temporally. The recognition that landscapes are the historical outcome of multiple complex interactions between social and natural processes has given rise to recent spatially-explicit LUCC models that integrate both ecological and socio-economic processes to examine these interactions, (hopefully) producing more accurate projections (e.g. Costanza *et al.* 2002). However, the extent and structure of this integration is dependent both upon the aims and objectives of the research and the subject background and expertise of the researcher (or research group). Typically the former (aims and objectives) are influenced by the latter (expertise) and distinct differences in how models are structured remain as a consequence.

Broadly speaking, simulation modelling approaches to LUCC can be classed into those that take a 'top-down', aggregated approach to represent systems structure and feedbacks, and those that take a 'bottom-up' approach explicitly representing the function and interaction of individual system components. Subtle variation in both the labels and structure of the models of these two approaches are found between subject areas. For example, in ecology the distinction would be manifested as the distinction between Spatially-Explicit Lansdcape Models (SELMs) and Individually-Based Models (IBMs) (Perry and Enright 2006), and in social simulation between Systems Dynamics Models (SDMs) and Agent-Based Models (ABMs) (Scholl 2001). The distinctions between, and advantages and disadvantages of, these approaches are examined in the following chapters with specific regard to the development of a spatially-explicit simulation model of LUCC and wildfire for SPA 56. In the remainder of this chapter recent attempts to link ecological and socio-economic processes in spatially explicit models are reviewed as the basis for the consideration of the most appropriate way to structure the simulation model for SPA 56.

3.4.2 Integrated Ecological-Economic Simulation Models

Ecologists have come to realise that human activity is often more significant than ecological processes in dictating direct agricultural/forest LUCC (especially in the Mediterranean – see section 2.3). Integrated ecological-economic modelling has been discussed and employed over the last 20 years with many papers extolling their potential for accurately representing and predicting land-use change (e.g. Brouwer *et al.* 1985, Bockstael *et al.* 1995, Veldkamp and Fresco 1996b, Munier *et al.* 2004). In contrast to ecological process-based models that have focused on land-cover change (LCC), integrated ecological-economic models have concerned themselves with land-use change (LUC – and particularly agricultural LUC). Economists study the interactions of traders, changes in financial value of commodities, and the flows of money through the worlds' economies. Generally, this demands studying and predicting human activity. Specifically it means the study of *Homo economicus*, the theoretical human species whose decisions are all perfectly economically rational. However, decisions regarding LUC are not necessarily determined economically and are also influenced by the values and goals of the individual(s) making the decision (Daly and Cobb 1989, Janssen and Jager 2000, Siebenhuner 2000, van den Bergh *et al.* 2000).

The difficulties of integrating ecological and economic theory into a model or framework for study have been outlined by Svedin (1985) and Bockstael *et al.* (1995). These authors highlight some common points regarding time and space scales. First, the spatial boundaries on systems analysis may not coincide, as economists place their boundaries according to the extent of the market, whilst ecologists typically use physical features. Second, the temporal extents of study may differ vastly as economists do not believe they can predict too far into the future, but ecologists are often more ambitious. Potentially the biggest stumbling block for integrating economic and ecological approaches, however, is the difference in the disciplines' fundamental approach and philosophy. First, economists disregard things that they cannot value financially but ecologists believe that a theoretical framework must take account of the most important aspects of a problem (regardless of financial value – Bockstael *et al.* 1995). As ecosystem processes are very difficult (if not impossible) to value in financial terms, these two standpoints are hard to reconcile. These differences in approach, and the difference in the systems of study, result in different "units of measurement, populations of interest, handling of risk and uncertainty and paradigms of analysis" when modelling (Bockstael *et al.* 1995 p.146). Svedin (1985) discusses the potential of using energy or information as fundamental units that might be used in common by the two disciplines. However, Bockstael *et al.* (1995) point out that reducing systems to the lowest possible common denominator has often simply resulted in larger black box models, compromising individual model modules' integrity. Svedin (1985) possibly realised this when he concluded that integration should be context-dependent for the study at hand, and that the underlying philosophies of different disciplines must be remembered when attempting integration.

The Conversion of Land-Use and its Effects (CLUE) model (Veldkamp and Fresco 1996b) has been developed as a tool to project future land-use, driven by biophysical and human land-use (economic) drivers and their interactions within a region. Spatially explicit and grid-based, this model considers broad land-use/cover types (e.g. agricultural systems, natural vegetation cover, urban areas) and assumes that agriculture is the main source of employment and income in the region studied, with all food produced either consumed or traded. Crop yields, technology levels, population size, perceived land values and biophysical constraints are incorporated, LUC occurring when biophysical and human demands cannot be met by the current land-use (Veldkamp and Fresco 1996b). Land-use needs are determined at a regional level but final change occurs at a local grid level. While Veldkamp and Fresco (1996b) believe that this model design is suitable for use in many regions around the world where diverse processes are at work, the assumptions regarding agriculture as the main source of employment and income seems to disregard many economic and social processes and influences, both internally and externally, to the region of study. Thus far, CLUE has only been applied, sometimes in extended form, in developing countries or rural areas; Costa Rica (Veldkamp and Fresco 1996a), China (Verburg and Veldkamp 2001), and the Philippines (Verburg and Veldkamp 2004) and several others (see CLUE 2006). It has been suggested that CLUE (and other similar models) are limited from an economic standpoint as they impose the underlying decision-making behaviour, making it impossible to model behavioural responses to LUC internally, and that decisions are made at the resolution of pixels rather than individual agents or farms (Irwin and Geoghegan 2001).

To overcome the limitations of imposing decision-making rules exogenously, the integrated ecological-economic Patuxent Landscape Model (PLM), forecasts LUC decisions probabilistically, based upon the economic and ecological state of a land parcel (grid cell) and its neighbours (Bockstael *et al.* 1995, Bockstael 1996, Costanza *et al.* 2002). Ecological and economic modules are spatially explicit, allowing examination of both the spatial and temporal dynamics of LUCC in a non-aggregated fashion (Irwin and Geoghegan 2001). As LUC decisions are derived from internal conditions they are able to respond to simulated landscape changes as the model runs, rather than being stipulated at the outset. The model is composed of a General Ecosystem Model (GEM) and an Economic Land-Use Conversion (ELUC) model. The GEM links hydrologic, nutrient, plant and agricultural modules spatially, monitoring the fluxes of materials between both cells and modules. The ELUC model takes two steps to calculate the probability of LUC. First, land parcels are valued economically using maximum likelihood estimates from independent variables such as access to markets/employment etc. and proximity to desirable or undesirable land-uses. Second, 'qualitative dependent variables' were estimated using historical LUC decisions to determine other factors influencing decisions (Geoghegan *et al.* 1997, Costanza *et al.* 2002). Rather than existing as one 'supermodel' the GEM and ELUC run in parallel at their own levels of specificity with information exchanged between the two. Thus, human decisions (from the ELUC) are imposed upon natural processes (of the GEM), and *vice versa*. Costanza *et al.* (2002) used the PLM to examine 18 scenarios of varying approaches to, and policies for, managing future development in the Patuxent River watershed, Maryland, which is experiencing increases in both population and land-use area. They observed that the model allowed examination of both site-specific and regional effects of new developments on water quality and net primary production, noting that the PLM's real power came from the dynamic linkage of ecological and economic modules and its spatially explicit nature. The PLM has been shown to be a very powerful tool when considering future LUCC and policy, but this power comes at a price. Extensive resources have been employed on this project by multiple participants over a long development period (the GEM was first conceptualised by Fitz *et al.* 1996).

3.4.3. Summary

The 'would-be world' (Casti 1997) provided by a simulation model provides a tool for researchers to 'experiment' on landscapes and examine scenarios in ways that would otherwise not be possible. The particular approach taken to construct a simulation model is dependent upon the nature of the system under study and the objectives of the research but may be broadly classed into 'top-down' aggregated models and 'bottom-up' model explicitly representing individual system components. Two large LUCC modelling projects that attempt to integrate socio-economic and biophysical processes presented in this section (CLUE and PLM) took the 'top-down' approach but others (to be discussed in chapter five) have taken a more actor-oriented approach. With this background to the field of LUCC simulation modelling in mind, the most appropriate approach to modelling SPA 56 is now discussed.

3.5 MODEL APPROACH AND VALIDATION

3.5.1 Introduction

As with all modelling projects, selection of the appropriate modelling technique for this project is dependent upon the research objectives, the nature of the system under scrutiny, the intended uses of the model and the resources available. As a reminder, the initial thesis aims are to:

iii) examine the impacts of human land-use/cover change upon wildfire regimes in a Mediterranean landscape;

iv) and to explore and evaluate novel methods to 'validate' simulation models (and processes of modelling) of environmental change considering human activity

3.5.2 Model Approach

The empirical modelling presented above highlighted the spatial and scaling problems of integrating data representing different types of processes at different levels of aggregation. Whilst progress is being made on hierarchical approaches to overcome such problems (e.g. Overmars and Verburg 2006) empirical modelling of this type would still not adequately address the first main research aim. The extrapolations for 2014 based on observed data are a useful guide to project how the landscape may look if current trends continue. However, the available statistical techniques provide little scope for the consideration of spatially-explicit pattern-process feedbacks, such as between fire and vegetation dynamics and between human activity and land-cover patterns, which are believed to be important in such a highly spatially heterogeneous landscape as SPA 56. GIS has been used in conjunction with the empirical modelling presented above to examine wildfire risk suggesting wildfire risk has increased, and will continue to increase, across SPA 56 (Millington 2005). However, the ecologically heterogeneous nature of SPA 56, allied with large and extended human presence demands a simulation approach rather than an empirical one to adequately address research aim one. Specifically, a spatially-explicit simulation model of vegetation dynamics capable of directly considering human activities is proposed. The use of this model will provide a novel approach to examine the effects of human activity on wildfire regimes. Whilst many spatially-explicit models of landscape change have considered wildfire and wildfire regimes as a primary component of succession-disturbance dynamics (see section 4.2 for review) few, in any, have attempted to explicitly incorporate human activity into a simulation model.

In the following chapters the rationale behind the construction of the model is presented. By way of a prelude, the model presented and analysed in these next chapters has two distinct components; i) a biophysical model designed to represent ecological processes in the landscape (chapter four) and ii) a human activity model constructed to represent landscape stakeholder decision-making of land-use and management (chapter five). These modules are combined to produce an integrated socio-ecological simulation model of LUCC in the landscape of SPA 56. The initial version of this model, named SPASIMv1 (SPA SIMulator Version 1), is run for scenarios of future economic and demographic change (chapter six). The biophysical model takes a Spatially-Explicit Landscape Model (SELM) type approach to represent vegetation dynamics and the fire regime on a 30 m resolution grid. Rather than considering individual organisms or species, a Plant Functional Type (PFT) approach is taken that reduces computational demands whilst maintaining an adequate representation of the vegetation dynamics at a landscape level. The problems of integrating ecological and economic models have been highlighted by Bockstael *et al.* (1995) and the models of agricultural change presented above rely on perfectly economically rational choices as the basis for allocating land-use. In contrast, because agriculture in SPA 56 is largely extensive (as opposed to chemically or mechanistically intensive) and demographic changes are believed to be one of the main factors driving agricultural land abandonment (section 2.5.4), an agent-based approach is taken here to consider a combination of both profit maximising stakeholders and traditional, non-commercial stakeholders. The activities of these agents determine modifications in land-cover modelled by the biophysical model.

3.5.3 Model Validation

It is not the aim of this project to explicitly predict future LUCC for specific points in the future because of the recognition of the problems of equifinality and 'model closure' of 'open' systems (Oreskes *et al.* 1994, Lane 2001, Beven 2002, Brown 2004). Briefly, but discussed in detail in chapter seven, Oreskes *et al.* (1994) demonstrated that temporal prediction (prediction of events to occur at explicit points in time or geographical space) is not possible by numerical models of complex natural systems. Equifinality in the model construction process implies there are multiple theories or models that could reproduce empirically observed data (Beven 2002). Furthermore, the understanding of the interaction of vegetation dynamics, disturbance regimes and human activities in Mediterranean landscapes is currently insufficient to attempt such explicit, accurate temporal predictions (discussed further in chapters four and five). Nevertheless, this thesis will achieve its first research aim by constructing and using a spatially-explicit simulation model of vegetation dynamics, human activity and wildfire.

Simulation models have been highlighted as a useful *heuristic* tool that can be used to mimic real world systems such that their manipulation allows exploration of the functioning of the system (Peck 2004). Using a model that does not attempt to explicitly predict future states of a specific system does not remove the need to assess the reliability of the results and knowledge produced by the model – the model will still need to be legitimised. Model validation will not be achieved here by matching the model output with real world observed landscape composition and configuration, due the epistemological problems mentioned above. Other ways of validating this model and its output will be needed and are examined in detail in chapters seven and eight. The usefulness and credibility of the simulation model produced for scenarios of economic and demographic change will be explored in non-quantitative ways. Specifically, the possibility of utilising local stakeholders' 'experience' knowledge regarding SPA 56 to assess and improve the credibility of the model output (or otherwise) will be explored. This possibility will be explored in chapter eight by examining how local stakeholders view the utility of the model and its output, and by questioning their trust (and the reasons for it or lack thereof) in both the model and the modeller (the 'expert'). Such issues have close ties with current global climate change research and the relationships between the global circulation models and the scientists that use them, policy-makers and the general public at large. From a social science perspective, such an approach raises questions and issues regarding the interaction between 'lay' and 'expert' knowledge, and the public understanding of science and environmental models. For example, Yearley (1999) concludes that detailed analysis of the public's understanding of models designed to aid policy and planning is needed to understand why these models are perceived as accurate, trustworthy, legitimate, or otherwise. Furthermore, this analysis will be able to contribute to recent studies of expertise and experience (e.g. Turner 2001, Collins and Evans 2002).

3.5.4 Summary

It has been suggested that an empirical modelling approach would be inadequate to sufficiently represent the dynamics of the socio-ecological processes present in SPA 56 and that a simulation model approach is required. Issues of model validation to be addressed later in the thesis were introduced and their implications for model use briefly outlined.

3.6 SUMMARY

This chapter has presented and reviewed the many different modelling methods and techniques that have been used to examine and predict LUCC. The primary empirical modelling techniques were outlined and the results of regression-based models presented. These results highlighted the problems of stationarity and of using differing 'types' of data (i.e socio-economic and biophysical) when using empirical approaches to model LUCC in dynamic socio-ecological systems. After a review of recent simulation modelling approaches a broad simulation modelling approach to examine LUCC in SPA 56 was proposed. This methodology is one that combines a SELM of wildfire and vegetation dynamics (chapter four) with an agent-based model of agricultural decision-making (chapter five) to represent the integral nature of humans in Mediterranean landscapes. The following discussion then highlighted some interesting points regarding the validity of the knowledge that this methodology might produce and which will be examined in the later chapters (seven and eight).

CHAPTER FOUR

LANDSCAPE FIRE SUCCESSION MODEL

4.1 INTRODUCTION

The term ‘Landscape Fire Succession Models’ (LFSMs) has been coined for those models that simulate the dynamic interaction of fire, vegetation, and often climate, in a spatially-explicit manner (Keane *et al.* 2004). By definition the model developed here is an LFSM, with the added consideration of human activity. Keane *et al.* (2004) reviewed 44 LFSMs, identifying the common strategies taken to represent vegetation succession, fire ignition and fire spread, highlighting the many methods and model structures possible. This chapter provides a detailed description of the particular methods used to represent the biophysical processes of vegetation-dynamics and wildfire in a LFSM (the human component of the model is described in the following chapter). The rationale for choosing the particular LFSM construction described from the many possible is also presented. First, the broad classes of modelling approach available to investigate vegetation-dynamics spatially are briefly discussed, before a more detailed examination of the particular strategies taken in Mediterranean-type ecosystems. Details of the representation by the LFSM of vegetation-dynamics are then presented, followed by those for the wildfire component. Finally, results from initial sensitivity analyses are discussed.

4.2 SPATIAL ECOLOGICAL MODELLING OF VEGETATION-DYNAMICS

4.2.1 Introduction

A general distinction can be made between ecological landscape models that take a ‘bottom-up’ approach, attempting to model all processes explicitly with minimal aggregation, versus those that take a ‘top-down’ approach, modelling larger spatial extents and often aggregating in time and space (Perry and Enright 2006). In ecological parlance these are termed Individual-Based Models (IBMs) and Spatially-Explicit Landscape Models (SELMs) respectively. Whilst some authors suggest this distinction is becoming increasingly blurred in ecological simulation, as models represent multiple scales and cross-scale interactions (Mladenoff and Baker 1999b), this dichotomy is used here to structure an examination of the different landscape models that been developed and applied. This brief description will provide the context for the decisions made regarding the model structure developed here.

4.2.2 Individual-Based Models

Individual-Based Models (IBMs) simulate vegetation dynamics by taking individual organisms as the basic unit of representation. IBMs may also be used to model the spatial behaviour of animal populations (see Perry and Bond 2004). Simulating the behaviour of individual organisms allows higher level system properties (i.e. swarms, regional distribution patterns) emerging from their interactions to be investigated (Breckling *et al.* 2006). The first widely used IBM was the JABOWA model (Botkin *et al.* 1972), which simulates forest-succession dynamics and is the prime example of a 'forest gap' model. JABOWA considers small forests plots (0.01 ha – 0.001 km^2) and represents birth, death and growth to allow focus on interactions between trees and between trees and their environment. Results from the simulation of these small plots can be applied across whole landscapes by assuming homogeneity in process across large extents (Mladenoff and Baker 1999a). Following the seminal review of IBMs by Huston *et al.* (1988), a rapid increase in the number of publications using IBMs was observed (Grimm 1999 – N.B. Grimm examined studies regarding animal populations only). One such IBM is SORTIE (Pacala *et al.* 1993, Pacala *et al.* 1996, Deutschman *et al.* 1999). SORTIE uses a very detailed mechanistic representation to explicitly model spatial dynamics of succession and competition – trees have an exact position in space (i.e. they do not occupy grid cells) and seed dispersal is also represented spatially-explicitly.

IBMs also often represent disturbance, and are being developed to consider increasingly larger spatial extents. For example, Keane *et al.* (1996) represented processes across a hierarchy of five organisational scales, from individual trees to the whole landscape, in a spatially-explicit model of vegetation succession and fire-dynamics (named FIRE-BGC). Individual trees are not modelled in a spatially-explicit manner, and competition between trees is considered at the stand level. Cross-scale process interactions are modelled both upward and downward throughout the organisational hierarchy. FIRE-BCG has been used to examine the responses of net primary production, nitrogen levels, and vegetation-dynamics to variations in fire regime (Keane *et al.* 1996), and also to consider the potential impacts of future climate change on fire regimes (Keane *et al.* 1999). Miller and Urban (1999) also presented a model that successfully coupled fire simulation to both climate and forest pattern at the individual tree level. The previously existing forest gap model ZELIG (Urban and Shugart 1992) was expanded by the addition of a fire simulation sub-model and re-parameterised to suit the study area in the Sierra Nevada, California.

4.2.3 Spatially-Explicit Landscape Models

A spatially-explicit model is one in which the behaviour of a single model unit of landscape space (often a pixel or grid cell) cannot be predicted without reference to its relative location in the landscape and to neighbouring units (Mladenoff and Baker 1999a). They are therefore useful tools with which to examine processes that propagate or disperse across landscapes through time (e.g. fire propagation or seed dispersal). SELMs take this approach to examine ecological processes across much larger spatial and temporal extents compared with IBMs (e.g. LANDIS examines landscapes of extent $1 \times 10^2 - 1 \times 10^4$ km^2, potentially for centuries – Mladenoff and He 1999). Further, their spatially-explicit nature allows examination of landscape patterns and their reciprocal feedbacks with landscape processes, characteristic of the questions asked by landscape ecology (Perry and Enright 2006). In contrast to most IBMs, SELMs are well suited for examining the impacts of disturbance at the landscape level (and above), and many landscape fire succession models have been produced (see Keane *et al.* 2004).

LANDIS is a spatially-explicit model of forest landscape dynamics and processes, representing vegetation at the species-cohort level (Mladenoff *et al.* 1996, He and Mladenoff 1999). This approach is markedly different from that of IBMs in terms of scale and detail of representation, but generalising small-scale processes allows simulation of the larger, more spatially dependent, processes. As a result, the model developed is better suited to use in multiple, different landscapes. LANDIS requires life-history attributes for each vegetation species modelled (e.g. age of sexual maturity, shade tolerance and effective seed-dispersal distance), along with various other environmental data (e.g. climatic, topographical and lithographic data) to classify ‘land types’ within the landscape. LANDIS is freely available (University of Missouri 2004) and has been so widely used since the mid-1990s that recently an entire issue of *Ecological Modelling* was devoted to examining the lessons learned from its development and use (*Ecological Modelling* 2004 Vol. 180, Issue 1). LANDIS has been used to examine the interactions between vegetation-dynamics and disturbance regimes (He and Mladenoff 1999), the effects of climate change on landscape disturbance regimes (He *et al.* 1999), and to simulate impacts of forest management practices such as timber harvesting (Gustafson *et al.* 2000). Franklin *et al.* (2001) used LANDIS to examine the effects of wildfire regimes on landscape patterns of Southern Californian plant functional types (see section 4.2.4) and suggested that while the model was originally developed for use in temperate forests, it is useful for the simulation of disturbance regimes in Mediterranean-type ecosystems.

4.2.4 Modelling Tradeoffs in Mediterranean-type Ecosystems

The two approaches presented above (bottom-up/IBMs and top-down/SELMs) to modelling vegetation-dynamics and associated biological/landscape processes highlight the relationship between scale and complexity. IBMs examine processes acting on and by individuals within a population across generally small spatial and temporal extents. SELMs examine vegetation in an aggregated, generally less detailed manner (e.g. at the species level), but consider these processes across much larger space and time extents. This dualism is in part due to the trade-offs are made to offset available computer processing power with the complexity of models. The construction of models considering processes of both fire and vegetation-dynamics at high levels of detail is appealing, but implementation has been found to be difficult due to the high levels of parameterisation and computational power required (Perry and Enright 2006). However, this dualism has also been due to the need for ecologists to represent processes and phenomena at their inherent scales (extent and resolution – e.g. Wiens and Milne 1989, Levin 1992). In turn, the impacts of these processes upon landscape pattern cannot be appreciated if an inappropriate scale of representation or analysis is used. Trade-offs between processing power, model complexity and data available for model parameterisation in Mediterranean environments, along with the spatial and temporal extent of the proposed research, mean that a SELM-type approach utilising conceptual Plant Functional Types is the most appropriate here.

Plant Functional Types (PFTs) are a useful conceptual model to examine vegetation-dynamics as they enable systematic analysis of ecosystem function and sensitivity to environmental change (McIntyre and Hobbs 1999). PFTs classify vegetation according to common plant responses to their environment in terms of growth, reproduction strategies and resource competition, and therefore provide a simplified representation of the numerous plant species in an ecosystem (McIntyre and Hobbs 1999, Rusch *et al.* 2003). Compared with the representation of individual plants, simulating the behaviour of PFTs reduces complexity of models but allows realistic representation of plant competition, growth and response to disturbance at a broader phenomenological level. When modelling the response of vegetation-dynamics to changes in the environment in Mediterranean-type ecosystems (for example changes in disturbance regime), Pausas (1999a) suggests that taking a PFT approach overcomes many of the problems associated with attempting to model individual plants in regions with Mediterranean-type vegetation. These problems are numerous but primarily related to particular morphological and behavioural differences between Mediterranean-type species and those of temperate regions. For instance, Pausas (1999a) suggests that the use of the same allometric equations (describing differences in growth of various parts of the same plant, e.g. roots versus trunk) for all species is not acceptable in Mediterranean-type vegetation, whereas it may be for models representing temperate regions. Furthermore, it is often difficult to establish the growth rates or life span of Mediterranean-type vegetation species (Pausas 1999a, Mouillot *et al.* 2001). Growth rates often vary in time according to resource availability (especially due to water availability and temperature), it is hard to establish growth rates empirically from tree rings in Mediterranean-type species, and root networks of species that regenerate following disturbance (see section 4.3.2) may be hundreds of years old while the above-ground vegetation may appear in a juvenile state (Pausas 1999a, Grove and Rackham 2001). Underground structures further impede the use of IBMs in these regions, as it is difficult to estimate parameters quantitatively for underground growth and competition (Pausas 1999a). Finally, the early stages of Mediterranean-type vegetation succession are not currently understood in sufficient detail for consideration at the individual level (Mouillot *et al.* 2001). The model by Malanson *et al.* (1992) is the closest to an IBM (as defined above) that has been developed for a Mediterranean-type ecosystem (in their case Californian chaparral). However this model, while using well-developed vegetation climate response functions, considered cohorts of plants rather than individuals (as with SELMs like LANDIS – see section 4.2.3).

4.2.5 Summary

Of the two broad modelling approaches used in spatial ecological modelling (IBMs and SELMs), the combination of the scale of this investigation (1×10^3 km^2 extent) with constraints on processing power, model complexity and parameterisation data for Mediterranean-type environments make a spatially-explicit landscape model-type approach most appropriate for this research.

4.3 MODELLING MEDITERRANEAN BASIN VEGETATION-DYNAMICS

4.3.1 Introduction

As outlined in section 2.2, the traditional conceptualisation of succession in Mediterranean Basin landscapes is one where shade-intolerant pines are replaced by shade-tolerant oaks that establish in the pine understory (e.g. Barbero *et al.* 1990b, Zavala *et al.* 2000). However, this theory precludes the occurrence of disturbance and the spatial variation of resources and their gradients that often prevent the evergreen oak 'climax' being reached. Thus, vegetation establishment and succession in Mediterranean-type ecosystems, such as the Mediterranean Basin, is spatially-dependent upon resource gradients (mainly water and light), disturbance type and intensity, previous land-use/cover, and the vegetation of adjacent land areas (seed dispersal). The importance of life-history strategies and disturbance in Mediterranean Basin landscapes demands a consideration of both succession and disturbance at a variety of scales (Mladenoff and Baker 1999a). Conceptual models of these processes are now presented, followed by an examination of their implementation in spatially-explicit simulation models.

4.3.2 Conceptual Models

'Resprouters' and 'seeders' are the two PFTs frequently used to describe the life-history strategies adopted by Mediterranean-type vegetation to survive in the face of frequent disturbance (e.g. Keeley and Zedler 1978, Barbero *et al.* 1990b, Pausas 1999a, Verdu 2000, Zavala *et al.* 2000, Pausas 2001). The evergreen Holm oak (*Quercus ilex*), widespread across Spain, is a good example of a 'resprouter' species (e.g. see Figure 2.7a and Save *et al.* 1999). *Q. ilex* is widespread across the western Mediterranean thanks to its ability to withstand wide variations in thermic, hydric and substrate conditions (Barbero *et al.* 1992). The Aleppo pine (*Pinus halepensis*) is the most common and widely-distributed pine in the Mediterranean Basin (Barbero *et al.* 1998), and is the prime example of a 'seeder' species (see Figure 2.7b).

Resprouters rely on large underground biomass stores (lignotubers) and root systems or protection of above ground biomass to survive disturbance and resprout vegetatively. Regeneration may occur soon after being burned (Lopez-Soria and Castell 1992) or cut for management purposes (e.g. Gracia and Retana 2004). Following fire, *Q. ilex* generally resprouts from underground lignotubers, while the thick corky bark of *Q. suber* protects above-ground biomass that then resprouts (Pausas 1997). Following cutting, *Q. ilex* will resprout direct from the remaining aboveground stool (Espelta *et al.* 2003). Seeders on the other hand generally die in the event of disturbance, but populations are maintained by rapid recolonisation of the disturbed area from seeds stored in fruits and cones in the canopy (Enright *et al.* 1998a). Seed release in serotinous species (species that store seeds for release following disturbance) is often induced by fire (pyriscence). Seedling recruitment is very high immediately following a fire, and *P. halepensis* allocates much of its resources when young to seed development in order to reduce reproductive failure in the event of a short interval between fires (Ne'eman *et al.* 2004).

The trade-offs associated with these two life-history strategies have been widely examined (e.g. Keeley and Zedler 1978, Enright *et al.* 1998a, 1998b). Once established, the resprouter that survived fire (with its root network intact) would have little competition from seeders that (generally) would not have survived. Mortality during fire demands that seeders allocate a large amount of resources to seed production rather than to plant growth (i.e. to a future generation rather than the current generation) to attempt to ensure persistence following fire. However, resprouters are characterised by lower levels of seed production and storage (e.g. Cowling and Lamont 1987) as resources have been diverted to ensure fire survival (e.g. the thick corky bark of *Q. suber* – Pausas 1997). Therefore the ability to survive fire has a cost in that colonisation is likely to be slow or weakened when opportunities for recruitment do present themselves (Enright *et al.* 1998b). For example, seed (acorn) survival of Mediterranean oak species is unlikely following fire because they are sensitive to both dehydration and heat, and are vulnerable to predation (Santos and Telleria 1997, Retana *et al.* 1999, Leiva and Fernandez-Ales 2003). Bellingham and Sparrow (2000) presented a model to account for the relative allocation of resprouting versus seeding for a variety of disturbance regimes. Their model suggested the dominance of seeders at low disturbance frequencies (as allocation of storage organs for regrowth would reduce competitive ability as resprouting is needed infrequently), while resprouters would be dominant at intermediate disturbance frequencies before seeders again become dominant at very high frequencies (because resprouters are unable to maintain storage organs against such a high frequency of damage).

The deterministic, equilibrium-based Clementsian view of succession (progressing toward a stable climax community) is currently widely rejected, and the use of the term ‘succession’ is now understood to describe vegetation distribution and change due to competition across spatio-temporal resource gradients (section 2.2). For the two Mediterranean PFTs considered here, and their associated life-history traits, two ‘pathways’ of vegetation succession and change have been suggested – secondary (‘old-field’) succession on abandoned fields or following extreme intensities of disturbance, and regeneration succession following moderate disturbance (van der Maarel 1988, Tatoni and Roche 1994). Depending upon available resources, resprouters would be expected to (re-)establish themselves faster and dominate seeders in regeneration succession, whilst seeder species would be expected to have a stronger presence in secondary succession as all resprouter material is likely to have been removed by humans for agricultural purposes (or, less commonly, destroyed by extreme intensity fire events – Tatoni and Roche 1994). These conceptual succession pathways are discussed in more detail in sections 4.4.3 and 4.4.4.

Competition for water and light in the face of disturbance (by fire and animal browsing) are the main drivers of vegetation-dynamics in Mediterranean regions. The nutrient content of soils is generally poor throughout the Mediterranean Basin and therefore soil quality does not often act as a resource for competition between species (Vila and Sardans 1999), but soil depth may act as a competitive resource in some instances (via water availability – Lopez-Soria and Castell 1992). Predominantly however, it is competition for water and light following disturbance (due to canopy removal), and along gradients of these resources that leads to characteristic Mediterranean community structures (Vila and Sardans 1999). Zavala *et al.* (2000) reviewed the most important factors required in a conceptual model of Mediterranean forest vegetation-dynamics. They too highlighted that the dynamics of mixed *Q. ilex* – *P. halepensis* forests may primarily be described by gradients of water availability and light intensity (i.e. time since disturbance). Furthermore, they emphasise the need to consider (at levels of aggregation appropriate to the species’ responses) the processes generating and produced by disturbance and spatial vegetation patterns.

4.3.3 Spatial Simulation Models

The feasibility of the spatially-explicit physiologically-based forest dynamics modelling that has been undertaken in better-studied ecosystems (e.g. temperate) has been doubted in Mediterranean landscapes because of their characteristic complexity and heterogeneity (Zavala *et al.* 2000). The problems of representing individuals in

Mediterranean-type ecosystems have already been highlighted (see section 4.2.4 – e.g. Pausas 1999a). Because of this, the development of conceptual models of vegetation-dynamics (i.e. PFTs) has been the standard approach. Zavala *et al.* (2000) highlighted that until their review no attempt had been made at modelling landscape vegetation and fire-dynamics in a spatially-explicit physiologically-based fashion in Mediterranean ecosystems. Prior to this Pausas (1999b) presented two non-spatial models of Mediterranean-type vegetation-dynamics using PFTs. The FATE model (Moore and Noble 1990) is a qualitative rule-based model considering vegetation-dynamics at a cohort level as plants proceed through four discrete life stages – propagules, seedlings, immature and mature plants (Pausas 1999b). Output is a qualitative description of the vegetation in each cohort (abundance defined as absent, low, medium and high). The second model, BROLLA (Pausas 1998), is a simple stochastic, quantitative, mechanistic gap model that simulates establishment, growth and death of individual plants on a 200 m^2 plot. Pausas (1999b) used these models to simulate abundances of different plant (functional) types under scenarios of varying fire regime and seed dispersal, finding maximum abundance of seeder species at intermediate disturbance frequencies (in contrast to the conceptual model suggested by Bellingham and Sparrow 2000).

A spatial understanding of the many factors and processes influencing Mediterranean vegetation-dynamics is vital, however. For example, Lookingbill and Zavala (2000) found that oak seedlings were positively spatially associated with pine trees, as the latter afforded the saplings safe sites for recruitment. Thus, several attempts have been made to model Mediterranean-type vegetation-dynamics in a spatially-explicit manner, with varying degrees of mechanistic representation. A sole SELM has been built for the Mediterranean since Zavala *et al.* (2000) suggested the possibility of integrating physiological stand-level models with landscape-extent models. The process-based SImulator for meditERRnean landscApes (SIERRA Mouillot *et al.* 2001, Mouillot *et al.* 2002) has been designed to examine the interaction between vegetation-dynamics and fire regimes for landscapes with Mediterranean-type vegetation communities that experience large recurrent fires. A PFT-type approach, with stands of vegetation on a 30 m resolution grid, represents spatial heterogeneity in landscape patterns and processes – carbon and water budgets are used to drive competition and estimate fire risk (via vegetation water status). Seed dispersal, surface water flow and fire spread are also simulated spatially, but again the assumption is that water availability and solar radiation are critical constraints on vegetation productivity and competition. A large number of parameters are needed to drive this physiological, mechanistic model that uses numerous equations to simulate processes such as infiltration, root water uptake and net primary production.

Other spatially-explicit but less mechanically mimetic, and therefore less data-demanding, models recently constructed and implemented to consider Mediterranean-type vegetation-dynamics are those of Pausas (1999a, 2003, 2006) and Zavala and Zea (2004). MELCA (Pausas 2003) is a spatially-explicit model considering two plant functional types – resprouters and seeders. Three causes of mortality (age for both PFTs, fire for non-resprouters, and resprouting failure following fire for resprouters), growth, seed dispersal, and fire ignition and spread are all represented. The use of the model to simulate a 100-year extent suggests that the long(er)-term dynamics of Mediterranean-type vegetation is spatially dependent, and that differences in basic plant traits (i.e. the difference between resprouters and seeders) are important and will have long-term consequences (Pausas 2003). Using a similar modelling approach Zavala and Zea (2004) examined the spatial dynamics of the same two PFTs, varying soil-moisture and disturbance occurrence across a hypothetical landscape. They found that the spatial distribution of soil-moisture and the presence/absence of disturbance influenced both the spatial distribution of species and the temporal variation in the modelled populations. Initialising simulation runs with a random spatial distribution of plant seeds they found several pathways led to pine-oak interactions and shifting mosaic behaviour of the plant types in agreement with Holocene records for the Mediterranean Basin. Thus, the type of succession followed (regeneration or secondary) is also spatially dependent, determined at a specific location by the type of previous land-use or disturbance.

4.3.4 Summary

The traditional conceptualisation of succession in Mediterranean landscapes is one where shade-intolerant pines are replaced by shade-tolerant oaks that establish themselves in the pine understory (e.g. Barbero *et al.* 1990b, Zavala *et al.* 2000). However, this theory precludes the occurrence of disturbance and the spatial variation of resources and their gradients that often prevent this climax being reached. Thus, vegetation establishment and succession in Mediterranean-type ecosystems is dependent, in both time and space, upon resource gradients (moisture and light), disturbance type and intensity, previous land-use/cover, and the vegetation of adjacent land areas (seed dispersal). The recent history of modelling Mediterranean-type vegetation-dynamics is therefore one that has begun to consider processes in a spatially-explicit manner. Recent models have remained largely theoretical and independent of specific study area (e.g. Pausas 2003, Zavala and Zea 2004), and the problems of representing individual species because of high morphological and behavioural heterogeneity mean that conceptual models of vegetation life-history traits (i.e. PFTs) are still generally the most pertinent option for modelling at the landscape scale. Particularly, the plant functional types 'seeder' and 'resprouter' are especially useful to model the strategies of Mediterranean species – even the most mechanically representative spatially-explicit landscape model (SIERRA) aggregates stands of vegetation to consider PFTs. Whilst SIERRA is still the current 'state-of-the-art' for modelling Mediterranean-type vegetation-dynamics across landscapes, and does consider disturbance by fire, it lacks any representation of the human activity is ubiquitous in Mediterranean landscapes (as discussed in chapter two). With this review of the most recent literature in mind, the structure of the biophysical module developed in this thesis is now described in two sections – vegetation-dynamics first and the wildfire regime second.

4.4 LANDSCAPE FIRE SUCCESSION MODEL I: VEGETATION-DYNAMICS

4.4.1 Introduction

When ecological knowledge regarding the ecosystem(s) of a landscape is imprecise or incomplete, and when only roughly quantitative and qualitative data are available, time-driven modelling simulations have been suggested as useful for examining system dynamics (McIntosh 2003, McIntosh *et al.* 2003, Oxley *et al.* 2004). Consequently, McIntosh (2003) and McIntosh *et al.* (2003) developed the Rule-Based Community-Level Modelling (RBCLM) system for vegetation modelling with qualitative knowledge, in circumstances where quantitative data for model parameterisation are sparse. The RBCLM approach provides a means to represent changes in qualitative state variables (e.g. land-use categories) through time via IF…THEN statements (rules) and qualitative understanding linking state and environmental variables. The two key attributes of vegetation change addressed by the RBCLM approach are (McIntosh 2003):

1. *direction of transition* between discrete land-cover (vegetation) classes
2. *rate of transition* between these discrete land-cover classes.

Considering vegetation change at a categorical, PFT level allows qualitative understanding of vegetation-dynamics to be translated into a formal, spatial model at the landscape scale. McIntosh *et al.* (2003) demonstrated the utility of the RBCLM approach for a Mediterranean ecosystem, considering several land-cover types (including grassland, scrub, deciduous forest and *Quercus ilex* forest), environmental variables (including aspect, precipitation and temperature) and disturbance (i.e. fire and grazing). The model developed here exploits this methodology, basing rules for change on the behaviour of broad land-cover classes (defined in Table 4.1) and their interaction with key environmental resource constraints (water and light availability) and disturbance (fire and agriculture). A schematic overview of the model procedure is presented in Figure 4.1.

Table 4.1 Land-cover classes represented in the model. Potential transitions between land-cover classes are shown with the duration over which they occur. Transitions and durations are derived from previous literature and 'expert' knowledge as described in detail in section 4.4.4, and specified in Appendix I.

State	*Land-cover*	*Description*	*State Change*
1	Pine	Primarily *Pinus pinea* and *P. pinaster*	→ 2; 15 – 40 years* → 3; 20 years
2	Transition Forest	Mixed *Pinus, Quercus* and *Juniperus* species	→ 1; 20 – 30 years → 3; 20 – 25 years → 4; 20 – 50 years*
3	Deciduous	Primarily chestnut (*Castanea sativa*) and alder (*Alnus glutinosa*) but also *Populus* species	→ 1; 20 – 30 years → 2; 30 – 40 years
4	Holm Oak	*Quercus ilex*	→ 2; 30 years
5	Pasture	Land exclusively reserved for livestock grazing	→ 8; 3 years
6	Holm Oak with Pasture	Representative of traditional *dehesa* woodland	→ 8; 3 years
7	Cropland	Cereals, vines, olives, almonds and figs	→ 8; 3 years
8	Scrubland	*Cistus, Lavandula* and *Genista* species with *Pinus* and *Quercus* species	→ 1; 10 – 15 years → 2; 15 years → 3; 15 – 20 years → 4; 30 – 50 years
9	Water/Quarry	River, reservoir or open stone quarry	
10	Urban	Built-up area	
11	Burnt	Post-fire condition of states 1 – 8	→ 8; 3 years

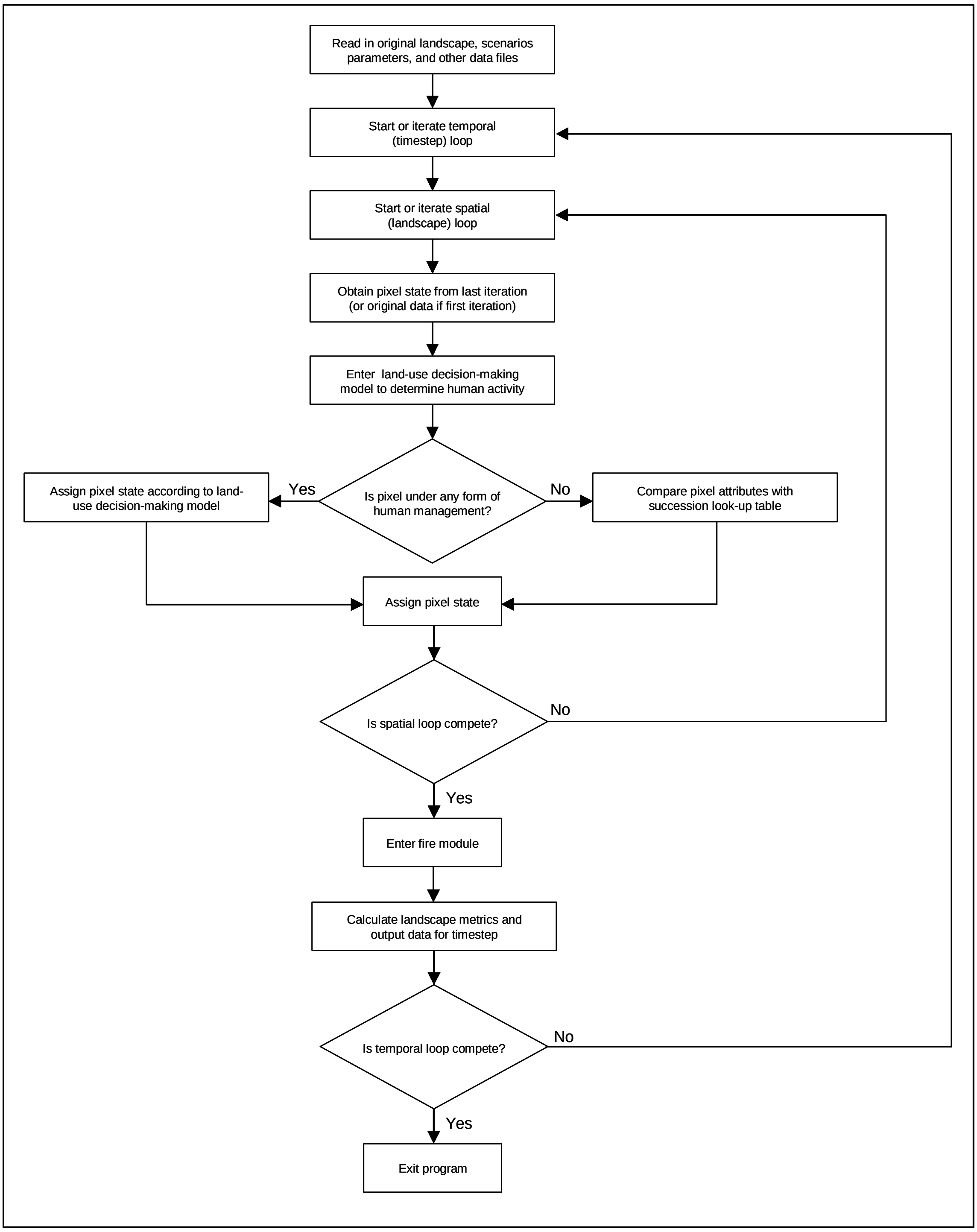

Figure 4.1 Schematic diagram of the LFSM. Starting from the top of the diagram, this process is iterated for as many annual steps as required. The land-use decision-making model is described in detail in chapter five.

The 11 broad land-cover classes are derived with both ‘natural’ vegetation-dynamics and human activity in mind. Pine, ‘Transition Forest’, Deciduous, Holm Oak and Scrub land-covers are considered as ‘natural’ vegetation types (i.e. human activity does little to influence dynamics) with distinct life history traits and reproductive strategies. The Pine and Holm Oak classes are considered as directly analogous to the PFTs ‘seeder’ and ‘resprouter’ (respectively) discussed above and as such the landscape is conceptualised in a similar vein to the archetypal models of Mediterranean pine-oak ecosystems (e.g. Pausas 2003). However, here other important vegetation types and assemblages are also considered. The 'Transition Forest' land-cover does not represent a single species, rather it is the manifestation of the mixed state between an idealised Pine-Oak transition and other mixed forest conditions (i.e. mixed Pine-Deciduous forest). The Deciduous forest found in the study area is also composed of mixed species (see Table 4.1), and is found in the relatively cooler, moister areas of the landscape. The deciduous species in SPA 56 show both resprouter and seeder responses to fire and are represented as a combination of these PTFs. Finally, Scrub is the key land-cover type linking the 'natural' vegetation types with the 'human activity' land- covers. All land-cover transitions from the human influenced covers (i.e. Pasture, Crops and Holm Oak with Pasture) toward the 'natural' vegetation types will first pass through this cover (see Table 4.1). Further, if burnt, *all* land-cover classes will become Scrub soon after burning. Unlike the other 'natural' land-covers, Scrub can undergo transition directly to all of the others. Composed of pioneer grass and shrub species (e.g. *Cistus, Lavandula* and *Genista* species), emergent *Pinus* saplings, and/or resprouting *Quercus* species (either in shrub form or stools/stumps of former trees following fire), the Scrub land-cover is the root of the possible land-cover transitions in this model of Mediterranean landscape dynamics.

Such broad classes of vegetation cover are also appropriate for, and consistent with, the 30 m spatial resolution of the lattice (grid of pixels) that represents the landscape. For each 30 m square pixel at each model timestep a rule-set – in combination with a look-up table for pixel physical attributes (Appendix I and section 4.4.4) – defines a direction of transition (from the current state to a future state) and a duration over which this transition occurs. Specifically, four pixel variables are considered in this process (McIntosh *et al.* 2003):

1. *State (S)*: current land-cover class (as defined in Table 4.1)
2. *Time in state* (T_{in}): length of time pixel has been in current state *S(t)* (i.e. *S* at current time *t*)
3. *Direction of transition* ($D\Delta$): the resulting state, *S(r)*, of pixel on completion of its current transition trajectory, as a function of *S(t)* and environmental conditions
4. *Total time required to complete transition* ($T\Delta$): a function of $D\Delta$ and pixel physical attributes.

Values for T_{in}, are derived from a logical rule base:

- IF $S(t) \neq S(t-1)$ THEN $T_{in}(t) = 1$ S 4.1
- IF $S(t) = S(t-1)$ AND $D\Delta(t) = D\Delta(t-1)$ THEN $T_{in}(t) = T_{in}(t-1)+1$ S 4.2
- IF $S(t) = S(t-1)$ AND $D\Delta(t) \neq D\Delta(t-1)$ THEN $T_{in}(t) = 1$ S 4.3

To derive $D\Delta$ and $T\Delta$, the set of pixel physical attributes are compared to the look-up table (see Appendix I). Pixel physical attributes are composed of environmental conditions (water and light availability) and successional attributes (successional pathway and seed sources), which are discussed in turn in the following sections. $T\Delta$ is dependent on both $D\Delta$ and pixel physical attributes, however, and thus a rule is required for the situation in which $D\Delta$ changes before a transition has successfully been completed:

- IF $D\Delta(t) \neq D\Delta(t-1)$ AND $S(t) = S(t-1)$ THEN $T\Delta = [T\Delta(t-1) + T\Delta(t)]/2$ S 4.4

where $T\Delta(t)$ is newly established for time t from the look-up table. Finally, at each timestep rules are checked to establish whether a state transition occurs:

- IF $T_{in}(t) \geq T\Delta(t)$ THEN $S(t+1) = S(D\Delta_X)$ S 4.5
- IF $T_{in}(t) < T\Delta(t)$ THEN $S(t+1) = S(t)$ S 4.6

It should be made clear that corresponding with the consideration of coarse vegetation types and assemblages, and the 30 m spatial resolution of these land-covers, the interaction between vegetation and environmental conditions and the accuracy of temporal estimates of change are also considered in a coarse manner. Environmental conditions are classified into broad groups as described in sections 4.4.2 and 4.4.3. Values for $T\Delta$ represent the mean duration for a transition of type $D\Delta$ to occur given the classes into which the pixel physical attributes fall. Transitions consider all aspects of vegetation-dynamics – growth, reproduction, mortality, invasion, replacement, succession – at an annual timestep within a 30 m square pixel. Whilst seasonal environmental fluctuations may influence vegetation in the short term, they do little to cause species (let alone PFT) replacement (Glenn-Lewin and Van der Maarel 1992). Further, the problems regarding representation of these dynamics at a more detailed taxonomic level than broad PFTs (e.g. at the individual species level), have already been highlighted (section 4.4.2). Given that this model is constructed at the landscape scale (1×10^3 km^2) for analysis of change over decades, the temporal generalisations (averaging) made here are deemed justified in terms of both the best possible representational accuracy (given the available data) and computational resources. The influences of environmental conditions and successional attributes on pixel physical attributes are now described.

4.4.2 Environmental Conditions

4.4.2.1 Climatic Variability and Change

Precipitation and temperature are the two key climatic variables that are considered in the model, precipitation in both vegetation and fire-dynamics components, temperature in the latter only. Wind is also considered. Maps of each variable were derived as described in section 2.5.5. The initial maps are used as a baseline from which inherent annual climatic variability and trends of climatic change are considered throughout model runs. Interannual climate variability is generally measured using the temporal standard deviation (SD) of the variable in question (e.g. temperature). However, the SD for annual precipitation is affected by its mean and therefore the coefficient of variation (CV) is a more independent measure (Giorgi *et al.* 2004b). Giorgi *et al.* (2004b) found that for the Iberian peninsula over the interval 1961 – 1990 temperature SD ranged over 0.5 – 1.0°C throughout the year, and precipitation CV over 0.25 – 0.50. An examination of climate records for the study area shows that the temperature SD averaged 1.0°C (across the study area for the interval 1974 – 1996) and precipitation CV averaged 0.29 (across the study area for the interval 1956 – 1996). Climatic variability was simulated by generating a normally-distributed random deviate in a given interval (±1°C for temperature and ±0.3 for precipitation). For temperature this value was simply added to the current annual mean – for precipitation the value was multiplied by the current annual mean and the result added to the mean. If required, scenarios of climatic change for a model replicate (run) are provided by the user, specifying the change (since the previous timestep) in the mean annual value of each variable at each timestep. Interannual variability is then calculated from this new mean value. Interannual climatic variability and climatic change is assumed to be spatially homogenous across the study area.

Data regarding wind direction and strength are lacking for SPA 56. Data from nearby weather stations, and anecdotal evidence, suggest that wind across the region as a whole blows predominantly from the west or southwest and is generally light in strength. Because of poor data availability, and because wind is generally weak, wind direction and strength are assumed in the model to be distributed equally around the compass and across three classes of strength (arbitrarily 'Strong', 'Medium' and 'Light'). Thus, seed dispersal is assumed to be evenly distributed in all directions and wind direction and strength are chosen at random (with equal probability) for each fire simulated.

4.4.2.2 Water Availability

Soil-moisture availability (*SM*) in a pixel is calculated as a function of incoming precipitation and overland flow and outgoing overland flow:

$$SM = P + IR - OR \qquad \text{Eq. 4.1}$$

where *P* is precipitation, *IR* is incoming overland flow and *OR* is outgoing overland flow (all in units of mm). The Soil Conservation Service Curve Number (SCS-CN) method (SCS 1985) is a commonly used method for calculating runoff in agricultural, forest and urban watersheds due to its relatively low data and parameterisation requirements. It has been found to provide satisfactory estimates of annual runoff values during long-term hydrological simulations with coarse temporal resolution (i.e. daily or longer – Mishra and Singh 2004, Mishra *et al.* 2005). Because of these attributes the SCS-CN method is used here in preference to other more mechanistically detailed (and therefore data-demanding) approaches. The SCS-CN method calculates overland flow (*Q*) as:

$$Q = \frac{(P - 0.02S)^2}{(P + 0.08S)} \qquad \text{Eq. 4.2}$$

where *S* is given by:

$$S = 2.54\left(\frac{1000}{CN} - 10\right) \qquad \text{Eq. 4.3}$$

and where *CN* is a curve number. The curve number is a function of vegetation, slope and soil type. Curve numbers have been calculated for numerous vegetation and soil types and the values used here are presented in Appendix II.

To apply this method to a grid-based model a flow-routing algorithm is needed to distribute moisture around the landscape due to topography. The RUNOFF function in IDRISI {Clark Labs, 2004 #1623} was used to produce a map detailing into which adjacent pixel water would flow in the landscape. As soil erosion and other geomorphological processes are not considered in the model, this map is assumed not to change during model runs. Moisture availability is classified into three increasingly moist classes (xeric, mesic and hydric) as shown in Table 4.2.

Table 4.2 Soil-moisture class definitions. Soil-moisture is estimated via a combination of soil properties and topographic location (i.e. due to overland runoff)

Class	*Soil-Moisture (mm)*
Xeric	≤ 500
Mesic	500 < M ≤ 1000
Hydric	> 1000

Whilst recent literature provides a qualitative representation of the effects of soil-moisture on vegetation types (e.g. Zavala *et al.* 2000), specific quantitative relationships at these landscape scales are currently unavailable. Therefore, these parameter values are largely arbitrary and tuned to ensure model performance (i.e. proportions of representative vegetation-type classes) are within realistic bounds. These values are similar to those used in previous Mediterranean vegetation-dynamics modelling (e.g. Zavala and Zea 2004, Zavala and Bravo de la Parra 2005).

4.4.2.3 Light Availability

The availability of solar radiation is modelled as a function of the aspect of a pixel, with southerly facing slopes receiving greater insolation than northerly slopes annually. This situation is reflected in the look-up table (Appendix I and section 4.4.4), with deciduous vegetation favouring northerly slopes, and pine vegetation favouring southerly slopes. The shade-tolerance of evergreen (Holm) oak and preference to establish in the understory of other species is reflected in the conceptualisation of the landscape successional dynamics (as described in section 4.3.2).

4.4.3 Succession Attributes

4.4.3.1 Successional Pathway

At each timestep pixels are classified in the model as either being on a ‘secondary’ (‘old-field’) or ‘regeneration’ succession pathway. Following the work of van der Maarel (1988) and Tatoni and Philip (1994), secondary succession is assumed to occur in areas of agricultural abandonment (or very frequent/intense disturbance) where all resprouter material (e.g. lignotubers) are likely to have been removed. In these areas, seeder species will be dominant initially as their seed dispersal mechanisms from adjacent areas are more efficient than those of resprouters. Regeneration succession processes are assumed to be dominant in areas in which resprouter species have been present and disturbance has been moderate (i.e. cutting for firewood, infrequent/low-intensity fire, grazing). Thus, the succession pathway which a pixel is following is dependent upon previous land use and the intensity of any disturbance. The rules used to determine which succession type a pixel is following are;

- IF Pixel is Holm Oak or Holm Oak w/ Pasture, or S 4.7a
- IF Pixel is scrub, Transition forest, pasture or burnt and on regeneration pathway or contains mature oak, or S 4.7b
- IF Pixel is Pine or Deciduous and direction is toward Transition Forest on regeneration pathway S 4.7c
- THEN set succession regeneration pathway
- ELSE set succession to secondary pathway

The age of vegetation in a pixel is important to determine if maturity has been reached, allowing regeneration following fire (in the case of resprouters) and the production and dispersal of seeds (in the case of seeders). The ages at which vegetation is assumed to reach reproductive maturity are presented in Table 4.3. The model monitors the age of each key vegetation type (i.e. Pine, Oak, and Deciduous) in each pixel at each timestep regardless of land-cover. This monitoring is important to ensure situations where two or more mature vegetation types co-exist in a pixel represented by a single land-cover class. These situations may arise because of the coarse land-cover types used (e.g. ‘Transition Forest’ and Scrub) and the way in which disturbance is represented. Disturbance of a pixel by fire does not consider the intensity of burning and simply assumes that all burnt pixels are re-set to the ‘burnt’ state. However upon burning, and unless burn frequency is high (see below), *mature* resprouter vegetation is assumed to survive burning, whilst seeder species of all ages die (their seeds surviving if they were mature however). For instance, if an ‘Oak’ pixel is burnt land-cover is not classed as ‘Oak’ subsequent to burning, but mature oak genets will have survived the fire. Thus, ‘Oak Age’ is not re-set to zero during burning, allowing regeneration succession to be initiated subsequently. Furthermore, this allows the model to record the vegetation types present in the mixed land-cover types ‘Transition Forest’ and ‘Scrub’. Once an Oak land-cover is reached, all other species are assumed to have been out-competed and their ages are set to zero.

Table 4.3 Maturity age for three main forest land-cover types. At this age vegetation is assumed to be mature and capable of reproduction (i.e. producing and dispersing seeds and/or resprouting following disturbance)

Land-cover Type	*Maturity Age (years)*	*Source*
Oak	15	Pausas (1999)
Pine	12	Pausas (1999)
Deciduous	12	-

In reality, if burning is particularly intense or frequent resprouter organisms will not survive (Zavala *et al.* 2000). Fire intensity is not considered in this model, primarily because intensity is related to the volume of biomass available to burn. Vegetation biomass levels are not currently modelled but this is an area in which the model can be developed in the future. Fire return times to each pixel can be monitored easily, however, and are used to assess the survival of resprouter vegetation following disturbance. Mortality following fire is related to the organism biomass or trunk diameter (Moreno and Oechel 1993, Pausas 1997, Hodgkinson 1998). Again, as vegetation biomass is not considered here, age will be used as a proxy for biomass, with biomass increasing with age until a maximum. Thus, mortality occurs if:

$$f > A/200, \text{ when } A < A_{max} \quad \text{S 4.8a}$$

$$f > A_{max}/200, \text{ when } A \geq A_{max} \quad \text{S 4.8b}$$

where f is fire frequency (fires per year), A is age (years) and $A_{max} = 100$. The value 200 is largely arbitrary as insufficient data has prevented empirical studies from establishing the disturbance frequency causing mortality (e.g. Trabaud and Galtie 1996). The parameter A_{max} denotes that age at which the relationship between vegetation biomass and age becomes so weak as to be irrelevant. However, this value is also unfounded as an empirical fact and its influence on the dynamics of the model is explored in the sensitivity analysis (section 4.6).

4.4.3.2 Seed Sources

Seed sources for each vegetation type are assumed to be available in a pixel if mature vegetation is present either within that pixel or in an adjacent pixel. Furthermore, at each timestep, and at model initialisation, a number of seeds (a proportion of the number of cells containing mature vegetation for each of Pine, Oak and Deciduous land-covers) are dispersed to pixels randomly across the landscape. Such an approach has been used in previous models of Mediterranean-type ecosystems (e.g. Pausas 1999b) and is used here for two reasons. First, whilst the majority of seed dispersal will be local, dispersal can also occur over large distances. For example, *P. halepensis* can disperse its seeds by wind transport across distances of up to several kilometres (although most seeds do not reach distances greater than 30 m from their source – Nathan *et al.* 2000, Nathan and Ne'eman 2004). Whilst primarily regarded as a resprouter here, the model also considers oak seed (i.e. acorn) dispersal over longer distances driven primarily by acorn predators (such as birds and small mammals). If pixel physical attributes (including land-cover) are not favourable for germination seeds are assumed to die after one timestep. Second, the initial state of the landscape at the start of a model replicate, in terms of vegetation age and location of seed sources, is not accurately known. From the three land-cover maps available (i.e. for 1984, 1991 and 1999) seed locations and initial vegetation ages (using the rules specified in Table 4.4) are assigned. Because the original land-cover maps do not specify the mature vegetation types present in each Transition Forest pixel, Transition Forest in the initial land-cover map are assumed to contain *all* seed sources (as Transition Forest must contain at least one mature species by definition). On the other hand, Scrub pixels are assumed to contain no seed sources, as original land-cover maps do not specify vegetation types present and scrub does not necessarily contain any mature vegetation. Random dispersal of seeds of each vegetation type (to 30% of the number of pixels of each vegetation type) at initialisation of the model is an attempt to overcome this.

Table 4.4 Rules to establish initial age of three main forest land-cover types. LC denotes any one of Oak, Pine or Deciduous land-covers. If the rule is satisfied the corresponding initial age of the given land-cover is assigned as model initialisation.

Rule	*Age of LC (years)*
IF 1984 AND 1991 AND 1999 = *LC*	16
IF 1984 AND 1991 = *LC* AND 1999 ≠ *LC*	8
IF 1999 AND 1991 ≠ *LC*	1
IF 1999 = Transition Forest	1
IF 1999 ≠ *LC* AND 1999 ≠ Transition Forest	0

4.4.4 Modelling State Transitions

To use the RBCLM approach described in section 4.4.1 for modelling land-cover-state change, a look-up table detailing the direction of transition ($D\Delta$) and time required to complete that transition ($T\Delta$) is used. For every possible combination of pixel physical attributes, a value for $D\Delta$ and $T\Delta$ are listed in the look-up table (Appendix I). The predominant routes of change considered by the model (via the look-up table) are summarised in Figures 4.2a and 4.2b and Table 4.1.

Literature and data regarding vegetation change in the Mediterranean under ‘natural’ conditions (i.e. where there is very little or no anthropogenic influence) over large time (decadal/centenary) and space (1×10^3 km^2) scales are relatively sparse. The values in the look-up table are derived from three main sources. First, the excellent review of vegetation change in forest ecosystems in the western Mediterranean Basin by Barbero *et al.* (1990b); second, anecdotal evidence collected from other sources (e.g. Grove and Rackham 2001); and third, knowledge of the study region and its dynamics gathered from ‘experts’ (i.e. scientists, forestry managers etc.).

Barbero *et al.* (1990b) discuss the important changes in patterns of disturbance and vegetation-dynamics in the north and south of the western Mediterranean over the previous 50 years, caused by social, economic and, importantly, agricultural change. They highlight three models of the influence of vegetation strategies on spatial pattern in the Mediterranean – the expansion model (rapidly colonising, mainly *Pinus,* species); the resistance model (long-lived, disturbance resistant species, including *Quercus ilex*); and the stabilization model (late maturity with limited seed dispersal, mainly deciduous species). Not surprisingly, the authors suggest vegetation characterising the stabilisation model is not widely spread across Mediterranean landscapes, and that the general pattern following disturbance is initial colonisation by expansion model species later replaced by resistance model species. Crucially, the authors provide a quantitative guide (Figure 4.3) to the duration over which these changes occur (rather than the skeletal, qualitative description of pine-oak replacement by many other authors) based on observations of spatial patterns of stand ages (primarily in southern France but also across the northern Mediterranean Basin).

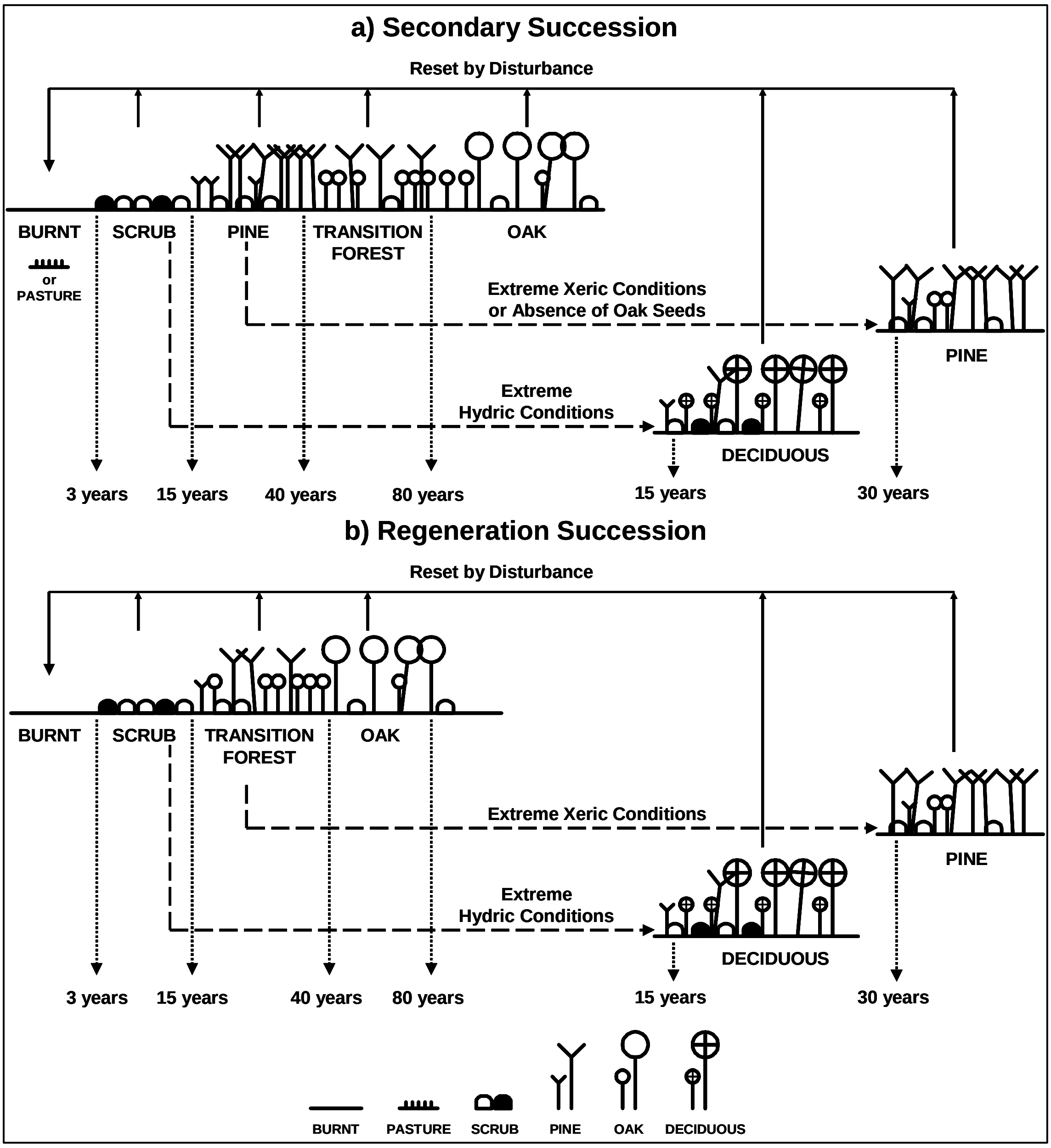

Figure 4.2 Succession pathways used in the LFSM. a) secondary succession and b) regeneration succession. Directions and rates of transition are shown for various environmental conditions.

The routes and durations specified in the look-up table are expected directions and times to transition for pixels under conditions in which pixel physical attributes do not change. Values for *DΔ* and *TΔ* will vary if pixel attributes change in time. Specifically, these values will vary depending upon seed, light and water availability. For example, if a seed source becomes available that was not present previously, the transition may change toward the vegetation represented by the seed source (e.g. transition from Scrub → Pine may change to Scrub → Deciduous if seeds are available for the later and conditions are hydric). A second example is that deciduous species will out-compete pine and oak in extreme hydric conditions – a shift toward these conditions may alter *DΔ*.

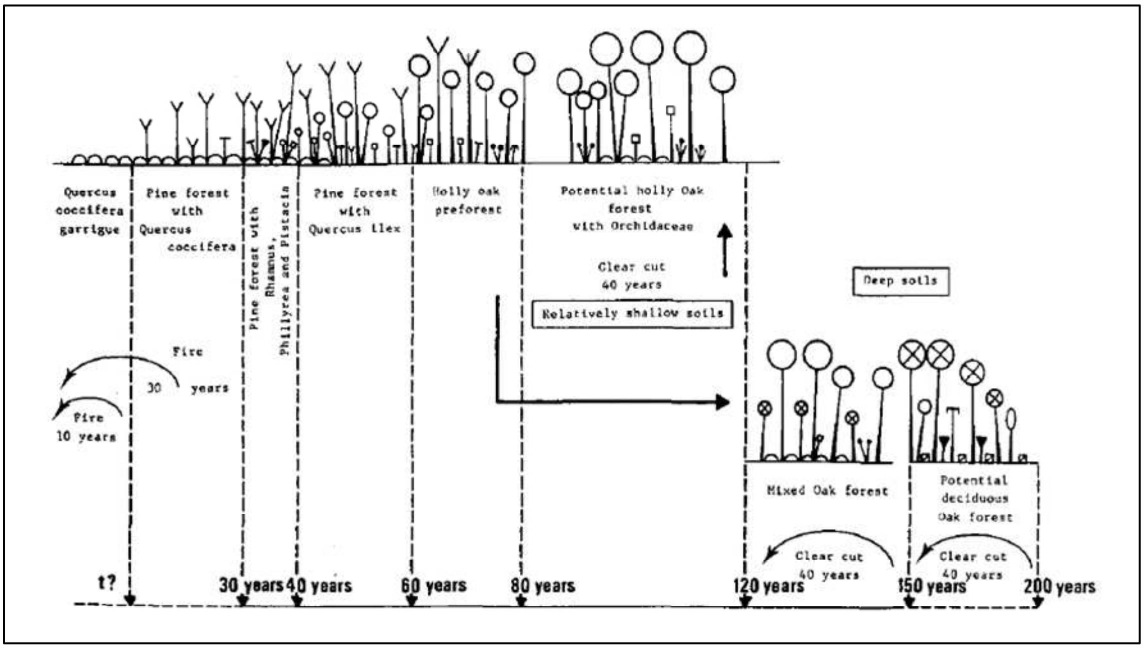

Figure 4.3 Example of the traditional conceptualisation of succession in Mediterranean-type vegetation. Source: Barbero *et al.* (1990b). Legend as for Figure 4.2.

The resulting vegetation dynamics of this model structure can be observed as a series of maps (Figure 4.4) and in time-series plots of land-cover abundance (Figure 4.5). For this 'no disturbance' model run starting from a 1999 land-cover map, all pixels originally in a 'human' land use (i.e. crops, pasture and HOP) are converted to bare ground (Burnt). After the initial dramatic shift from this bare ground to Scrub (not shown) an orderly succession toward the evergreen oak climax is observed. Pine comes to dominate the landscape by succeeding Scrub. Holm Oak establishes in the Pine understory leading to an increase in Transition Forest, before finally becoming dominant.

4.4.5 Summary

Vegetation-dynamics are represented in the LFSM using the Rule-Based Community-Level Modelling (RBCLM) system for vegetation modelling with qualitative knowledge, developed by McIntosh *et al.* (2003, 2003) for circumstances in which quantitative data for model parameterisation are sparse. Successional attributes (succession pathway, seed sources) and environmental conditions (water and light availability) are considered within a traditional conceptualisation of the nature of 'seeder' and 'resprouter' vegetation (e.g. Figures 4.2a and 4.2b). Output from initial LFSM runs (in an absence of disturbance) show model behaviour as expected from the basic model construction (Figures 4.4 and 4.5).

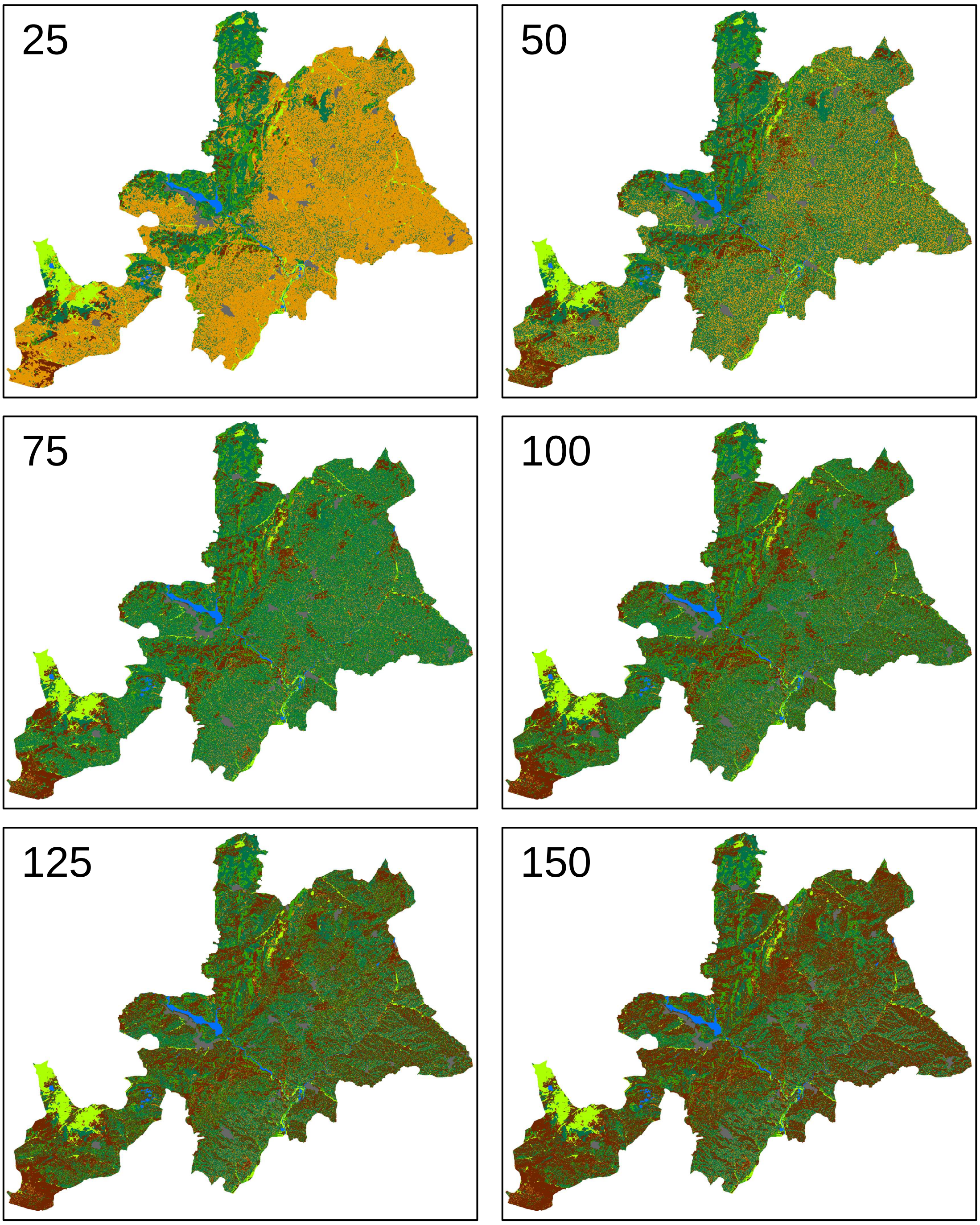

Figure 4.4 Spatial representation of landscape change from the LFSM. Land-cover for SPA56 for intervals of 25 years. The conceptual model of succession from Scrub through Pine and Transition Forest to Holm Oak is observed by variation in relative abundance of each land-cover though time. Legend as for Figure 2.10 (and Figure 4.5 over page).

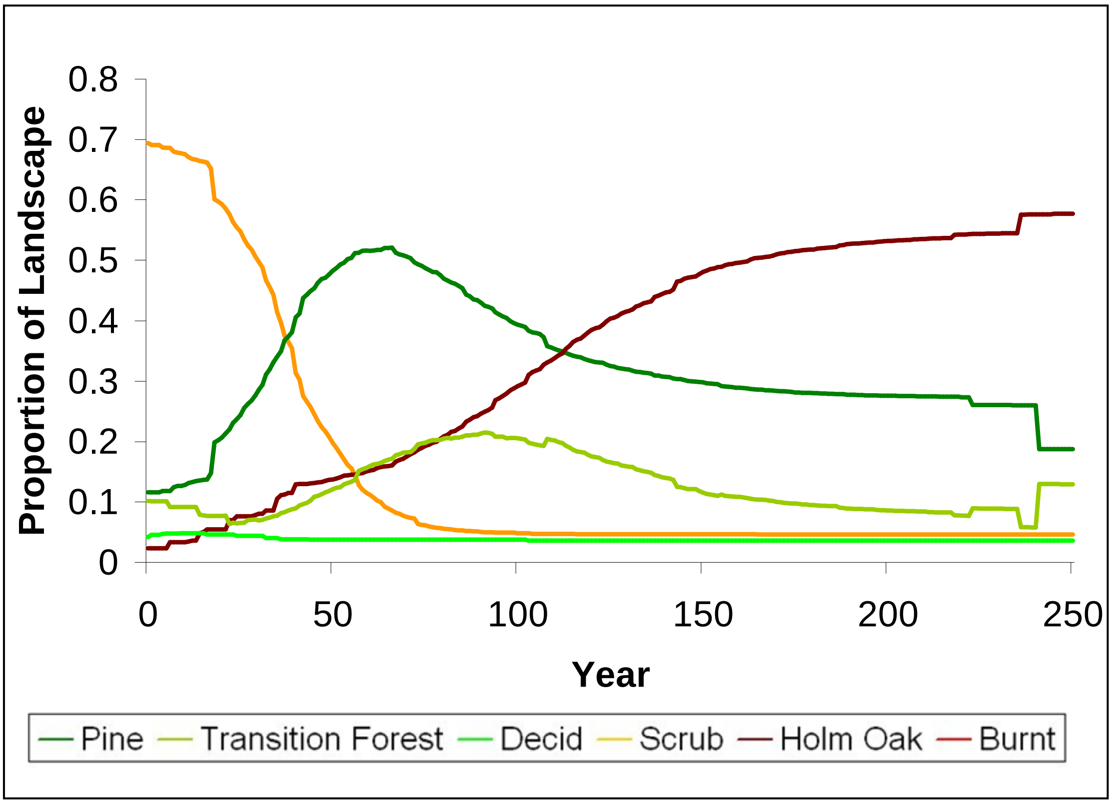

Figure 4.5 Time-series of landscape land-cover composition from the LFSM. This result is for a 'no disturbance' model run and demonstrates the conceptual model of succession from Scrub through Pine and Transition Forest to Holm Oak.

4.5 Landscape Fire Succession Model II: Wildfire Regime

4.5.1 Introduction

The wildfire 'regime' is the frequency, timing, and magnitude of all fires in a region (Whelan 1995). The frequency and timing of individual fires that contribute to the wildfire regime are related to the ignition of these events. Once alight, the magnitude of a fire is related to the subsequent behaviour (i.e. spread) of that fire. The spread of an individual fire is dependent upon many factors including microclimatic conditions, topography and fuel conditions. In a heterogeneous landscape these factors make the location (spatially) of an ignition an important determinant of the magnitude of the event. These aspects are now inspected in turn.

The vast majority of wildfires in mid-latitude ecosystems occur each year during a small number of months known as the 'fire season' (Whelan 1995). Regardless of the source of ignition (i.e. human or lightning) this seasonal effect is closely related to climatic variables (Whelan 1995, Westerling *et al.* 2003). Much more attention, however, has been directed at establishing the frequency of wildfire events in a given region (often of a given size – see below), primarily because of the ecological consequences on the reproductive strategies of vegetation species (Millington *et al.* 2006). Empirical analysis of wildfire records has suggested that the Weibull and negative exponential distributions are good models of fire return times (i.e. the mean interval between the burning of a point in a region Johnson and van Wagner 1985). Fire frequency and fire size are related (Li *et al.* 1999) with many studies suggesting 'heavy-tailed' frequency-size distributions (i.e. the upper tail decreases slowly – Millington *et al.* 2006).

The majority of the studies of fire frequency, including those cited above, have paid little regard to the factors driving the observed relationships. Millington *et al.* (2006 p.156) suggest that "considerable debate remains over the drivers of spatio-temporal variability in wildfire frequency (in particular the relative roles of weather v. fuels), and unravelling these patterns is a focus of current work". In the Mediterranean Basin, climate has frequently been cited as an important factor influencing ignition, with some studies suggesting that changes in wildfire frequency and hazard are due to changing climatic conditions (Vazquez and Moreno 1993, Pinol *et al.* 1998). Whilst GCMs are known to predict changes in intra-annual rainfall poorly (Albritton and Meira Filho 2001), predictions suggest increases in the number of extreme rainfall and drought periods for the Mediterranean in the future (Giorgi *et al.* 2004a, Gao *et al.* 2006). In these conditions fire risk and the number of fires would be expected to increase. The impacts of potential climate change (specifically warming) on lightning frequency remain unclear however (Williams 2005). Diaz-Delgado *et al.* (2004) suggest that the link between climate and fire return time is a relationship that has not yet been established with any confidence, and other studies have suggested human influence is just as important as climate in influencing changes in wildfire activity (De Luis *et al.* 2001). With human activity being the predominant cause of fire in the Mediterranean Basin (e.g. more than 97% of fires in the interval 1974 – 1994 in Spain were caused by humans – Moreno *et al.* 1998) considering the human element (alongside climate) is clearly required in any model of Mediterranean wildfire regimes. Spatial patterns of wildfire ignition are non-random (Vazquez and Moreno 2001), and related primarily to human activity, topography and vegetation. Salas and Chuvieco (1994) claim the clear evidence for the human influence on ignition is shown in their analysis of Spanish fire reports in the interval 1968 – 1988, which showed the majority of fires were started at weekends and during summer holidays, and near roads and trails. Mouillot *et al.* (2003, 2005) found the spatial location of ignition in rural Corsica to be a function of topography and vegetation, leading to high spatial contrasts in the frequency of burning (some areas burnt up to seven times in 40 years, others never). Lightning fires in Mediterranean regions have been found to be located non-randomly, occurring with greater frequency in mountainous areas (Vazquez and Moreno 1998). Whilst the specific details of spatial-temporal patterns of wildfire ignition in the Mediterranean remain to be clarified, a general summary of the literature is that ignition frequency is a function of human activity and climate, whilst ignition location is a function of human activity, topography and vegetation.

Several different modelling approaches have been developed to simulate individual wildfire behaviour (e.g. spatial nature of spread and rates of spread) ranging in sophistication and representation of the processes controlling wildfire spread. Perry (1998) distinguishes three types of fire behaviour model – physical, semi-physical and empirical. Physical models are based on the first principles of thermodynamics and known physical and chemical relationships (e.g. radiative transfer models – Albinet *et al.* 1986, Weber 1989). However, these models are very difficult to parameterise in the field and therefore their use has been limited. Semi-physical models consider heat fluxes and radiative transfer using empirical relationships to represent physical processes, enabling parameterisation for field conditions. The best known example is Rothermel's (1972) model, on which the United States National Fire Danger Rating System (NFDRS, Deeming *et al.* 1977, Burgan 1988), the BEHAVE fire prediction system (Burgan and Rothermel 1984, Andrews 1986) and the FARSITE fire growth simulation model (Finney 1998) are based. Examples of parameters required to implement these models include fuel particle properties (density etc.), fuel arrangement (fuel-bed depth etc.) and other environmental parameters (wind speed etc.). Perry (1998) distinguishes between two types of empirical model – statistical models establishing relationships between physical variables developed in test fires (similar to the 'black-box' models of LUCC described in section 3.2), and models that are specifically concerned with the simulation of the spatial behaviour of fire.

'Fire-shape' models are a form of empirical model (*sensu* Perry 1998). Knowing the ignition location, wind direction and speed it is possible to estimate to a reasonable degree of accuracy the size and shape of a wildfire from a set of fire-shape templates (Green 1983), the most commonly used being the ellipse (e.g. Anderson *et al.* 1982, Catchpole *et al.* 1992). However, fire-shape (elliptical) models often assume fire spread across uniform fuel, topography and microclimatic conditions, variation in any of which will cause variation in rate and direction of spread. Another form of empirical model, grid-based fire simulation models overcome these problems by dividing a landscape into finite cells and then assuming the internal state of a cell is uniform. Spread within and between cells is simulated on an individual basis. Such models belong to the class of models known as 'cellular automata models' (CAMs), of which possibly the most famous is the Forest Fire Cellular Automata (FFCA) that was used amongst a suite of similar models by Per Bak and others to propound the theory of 'Self-Organised Criticality' (SOC, e.g. Bak *et al.* 1987, Bak *et al.* 1988). In the FFCA, at each timestep during a model run, a 'tree' is dropped into a random cell of a grid. At a specified frequency 'matches' are dropped into the grid. If the match falls on a cell with a tree in it, the tree burns – any trees in neighbouring cells also burn and the fire spreads across the grid (if the match falls into an empty cell no fire occurs – Figure 4.6).

Although it inspired much research into the frequency-size scaling of empirical wildfires (see section 6.2), the 'forest fire' term in the FFCA was intended as a metaphor rather than a claim of direct representation (Millington *et al.* 2006). Such simplification does little to overcome the representational drawbacks of the elliptical fire-shape models. The metaphor fitted however because the basic underlying dynamics of a wildfire are exhibited. Vegetation growth (randomly dropping 'trees' into the grid) leads to potential ignition (randomly dropping 'matches' at a given frequency) and a fire then spreads dependent upon the spatial distribution of fuel (trees). The grid-based fire -dynamics model developed here builds upon this basic idea, producing a more realistic representation of the wildfire regime, by integrating with the vegetation-dynamics model described above, and by considering how environmental variables such as topography and climate influence vegetation flammability. As previously discussed (section 4.2.4), the representation of vegetation in the vegetation-dynamics model is coarse, due to the large space and time scales involved and the relative difficulty of representing Mediterranean vegetation in more detail. This representation therefore precludes the use of a more detailed fire behaviour model such as BEHAVE. Furthermore, as discussed at the outset of this chapter the integration of process-rich vegetation- and fire-dynamics models has proven to be troublesome because of data and computational constraints (Perry and Enright 2006). Constructing the model in the way described in this section allows straight-forward integration with the vegetation-dynamics model described above (section 4.4). The structure of the wildfire-dynamics model is now presented.

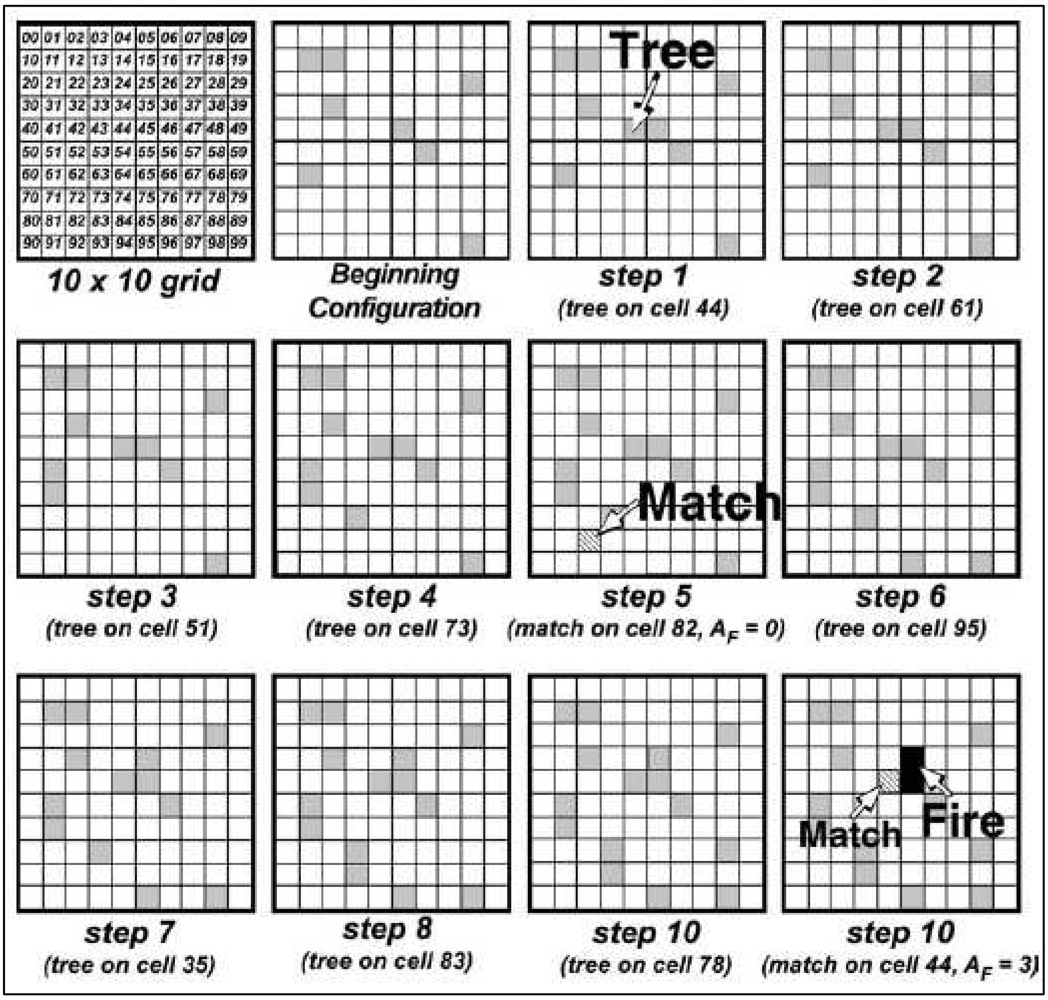

Figure 4.6 Illustration of the Forest Fire Cellular Automata Model. At each step 'trees' are dropped into the grid. At a given frequency 'matches' are dropped in to the grid (e.g. step 5). If a match lands on a cell occupied by a tree it (e.g. step 10), and all neighbouring trees are burned. Source: Millington *et al.* (2006)

4.5.2 Fire Ignition

4.5.2.1 Frequency

Whilst the Weibull distribution has been used widely in wildfire modelling to simulate fire return periods (i.e. the probability of a fire occurring at a point each year), it does so empirically, based solely upon the observed length of the disturbance cycle. For example, the models by Miller and Urban (1999) and Perry and Enright (2002) used a fire-interval method to establish whether a fire might be ignited in each pixel in a particular year. In this method, ignition-probability distributions are specified prior to running the model (i.e. the shape parameter of the Weibull distribution is set prior to running the model) and the state of the landscape (and its environment) is not considered at each timestep when evaluating ignition frequency. To represent the interactions and feedbacks between the ignition factor of primary interest (i.e. human activity) and the dynamics of the wider wildfire regime throughout model runs, an alternative method is desirable.

In an empirical study Diaz-Delgado *et al.* (2004) found that the number of fires per year in northeast Spain during the interval 1975 – 1998 followed a Poisson process. The Poisson distribution, in conjunction with the causal variables of human activity, climate conditions and vegetation, is used here to calculate the number of individual wildfire events during each timestep. The Poisson distribution allows the expression of the number of events occurring in a given time interval based on a known average occurrence rate (λ) and assuming the events are independent. Using the Poisson distribution, the probability *p* of the occurrence of exactly *x* events during a given time interval is given by:

$$p(x) = \frac{e^{-\lambda}\lambda^{x}}{x!} \qquad \text{Eq. 4.4}$$

where λ is the shape parameter (i.e. mean occurrence for the given time interval). The model uses climatic conditions and, depending upon the scenario being run, human activity to set the shape parameter at each timestep. Thus, while the mean number of events in a timestep is determined by the factors driving ignition, the actual number of events that occur is determined stochastically by comparing a random number in the interval [0,1] to the derived cumulative probability of each number of events. This method allows general trends to be modelled whilst maintaining a stochastic representation of inter-interval variation.

In model runs where climate alone is considered as a driver of the number of wildfire events per timestep, λ is calculated by:

$$\lambda = m * \frac{MAT}{MAP} \qquad \text{Eq. 4.5}$$

where *MAT* is mean annual temperature (°C), *MAP* is mean annual precipitation (mm) and *m* = 50. The value for parameter *m* was derived from observed data for the interval 1989 – 2000. This equation is used to represent vegetation moisture, and therefore differs from the soil-moisture classification presented in Table 4.4. Short-term (i.e. daily, weekly) climate conditions are known to influence wildfire ignition risk more than mean annual conditions, and thus are used by several major wildfire risk models (e.g. the US NFDRS, Deeming *et al.* 1977, Burgan 1988). However, the model here assumes that decreases in mean annual precipitation correspond with increases in the number/intensity of intra-annual drought periods, in turn increasing wildfire frequency. De Luis *et al.* (2000, 2001) observed such a link between annual and monthly precipitation patterns in Spain, and suggested it was one factor causing observed increases in frequency.

When human activity is also considered as a driver of the number of wildfire events per timestep, λ is calculated by:

$$\lambda = \lambda_c + HF + 0.1(AG - RB) \quad \text{Eq. 4.6}$$

where λ_c is λ from Eq. 4.5, *HF* is a baseline number of human fires to be ignited at each timestep, *AG* is the percentage difference between the initial number of farmers (model agents) at the start of the model run and that of the current landscape, and *RB* is the percentage of the landscape burnt in the last *RBY* years (explained below). This equation implicitly assumes a relationship between the number of fires ignited and the ‘types’ of people present in the landscape. A simple dichotomy of people ‘types’ is considered – ‘land-owners’ (farmers, hunting enterprises) and ‘non-land owners’ (commuters, tourists). The assumption is that there is a positive relationship between the number of non-land owners and the number of (unwanted) fire ignitions, and an inverse relationship between the numbers of people types (hence *AG*). Further, this equation assumes (via *RB*) that the number of non-land owners visiting the landscape decreases as the percentage of land that has been burnt in recent years increases (and therefore so too does the potential for ignition) as people will be unwilling to visit burnt areas for recreational purposes. These values are scaled by a factor of ten such that the values are upon the correct order of magnitude for the annual number of fires, and the value of *RBY* is set by default to 10 years. This value is based upon the subjective, aesthetic value that visitors place on the landscape and is difficult to quantify. Investigation into recreational landscape users’ preferences following fire would be useful for modelling purposes similar to this. The inclusion of the *HF* term allows a baseline number of fires to be ignited at each timestep. This can be used to simulate scenarios of the relative number of ‘non-land owners’ to ‘land owners’.

4.5.2.2 Spatial Location

The model can be set to ignite a specified number of fires per timestep distributed randomly across the landscape (in the 'non-flammable' land-covers 'Urban', 'Water' and 'Burnt'). However, fires in the Mediterranean Basin do not ignite or spread through the landscape randomly, with topography being a primary determinant of spatial patterns (Vazquez and Moreno 2001, Mouillot *et al.* 2003). In a statistical study Vazquez and Moreno (1998) found that the spatial patterns of lightning- and people-caused fires in Spain were significantly different, with the vast majority of lightning-caused fires igniting in mountainous areas and at upper elevations. The model can therefore be run for 'lightning fires only' scenarios, in which case fires ignite randomly in pixels above 1000 m ASL and with flammable land-cover. Currently, neither the 'random' nor 'lightning fires only' methods of ignition are particularly realistic in the study area or across the Mediterranean Basin as a whole. These modes of ignition are useful for testing the biophysical module or for considering landscapes without human activity (which are non-existent in contemporary times).

As noted above (section 4.5.1), the high proportion of fires ignited by humans demands the consideration of human activity in a model of Mediterranean fire-dynamics. The ability of Geographical Information Systems (GIS) to integrate multiple (maps of) spatial variables, wildfire-related data has led to their frequent use for mapping and modelling wildfire risk (e.g. Chuvieco and Salas 1996, Camia *et al.* 1999, Chuvieco *et al.* 1999, Millington 2005). The 'GIS approach' has generally been a qualitative-objective method, whereby specific values are assigned to risk-related variables, such as vegetation (fuel) condition and human presence, according to their perceived importance to determine ignition location (Millington 2005). However, the final risk maps produced using this approach consider the risk of a point in the landscape *burning*, whether because the fire originated at that point or spread into it from elsewhere. In the model developed here, risk of fire ignition and spread are considered separately. Thus, once it has been determined that a fire will ignite somewhere in the landscape (the number of fires for any timestep having been calculated as described in the previous section), the probability of ignition in each pixel is estimated as a direct relationship with human the level of human activity in or near it (Table 4.5). An analysis of fire reports for Spain (1968 – 1988) showed that fires were more likely to be ignited near roads and trails than elsewhere in the landscape (Salas and Chuvieco 1994). Probabilities of ignition are therefore estimated according to this evidence, and also consider the location of recreation areas (i.e. picnic sites) where high numbers of people gather (relative to the rest of the landscape, Table 4.5). A small probability (0.1) of ignition remains in all pixels in the landscape however. A uniform random deviate is compared with the probabilities specified in Table 4.5 to establish in which area of the landscape a fire ignites, and a pixel chosen at random from all those that fall within those areas. Probability of ignition is immediately set to zero if a pixel is in the burnt land-cover class, or within 90 m of a pixel that has burnt within the last *RBY* years (see Eq. 4.6 above). This procedure is repeated in turn for the specified number of fires that occur in each timestep (once the preceding fire has extinguished).

Table 4.5 Human fire ignition probabilities. These probabilities are estimated according to an analysis of Spanish fire reports that found fire are more likely to ignite near roads and trails (Salas and Chuvieco 1994). Areas near recreation sites are also considered areas at increased risk. All other points in the landscape have a minimal probability of ignition.

Pixel Location	*Human Ignition Probability*
None of the below	0.100
Within 45m of a trail	0.125
Within 15m of a trail	0.250
Within 45m of a road	0.175
Within 15m of a road	0.350
Within 90m of a recreation area	0.300

4.5.3 Fire Spread

Fire spread is modelled using a cellular automata approach that builds on previous, more abstract, models of this kind (i.e. Bak *et al.* 1987, Bak *et al.* 1988). Specifically, variation in vegetation flammability, slope and climatic conditions are considered as influences on spread. Management activities that might restrict fire size, such as the construction of fire-breaks and fire-fighting efforts, are also considered. The fire module procedure is represented by flow chart in Figure 4.7. The cellular automata approach essentially considers spread as a series of ignitions from cells to their neighbours. Probability of ignition in this context (i.e. probability of spread) is different to the probability of initial fire ignition (i.e. that process considered in section 4.5.2).

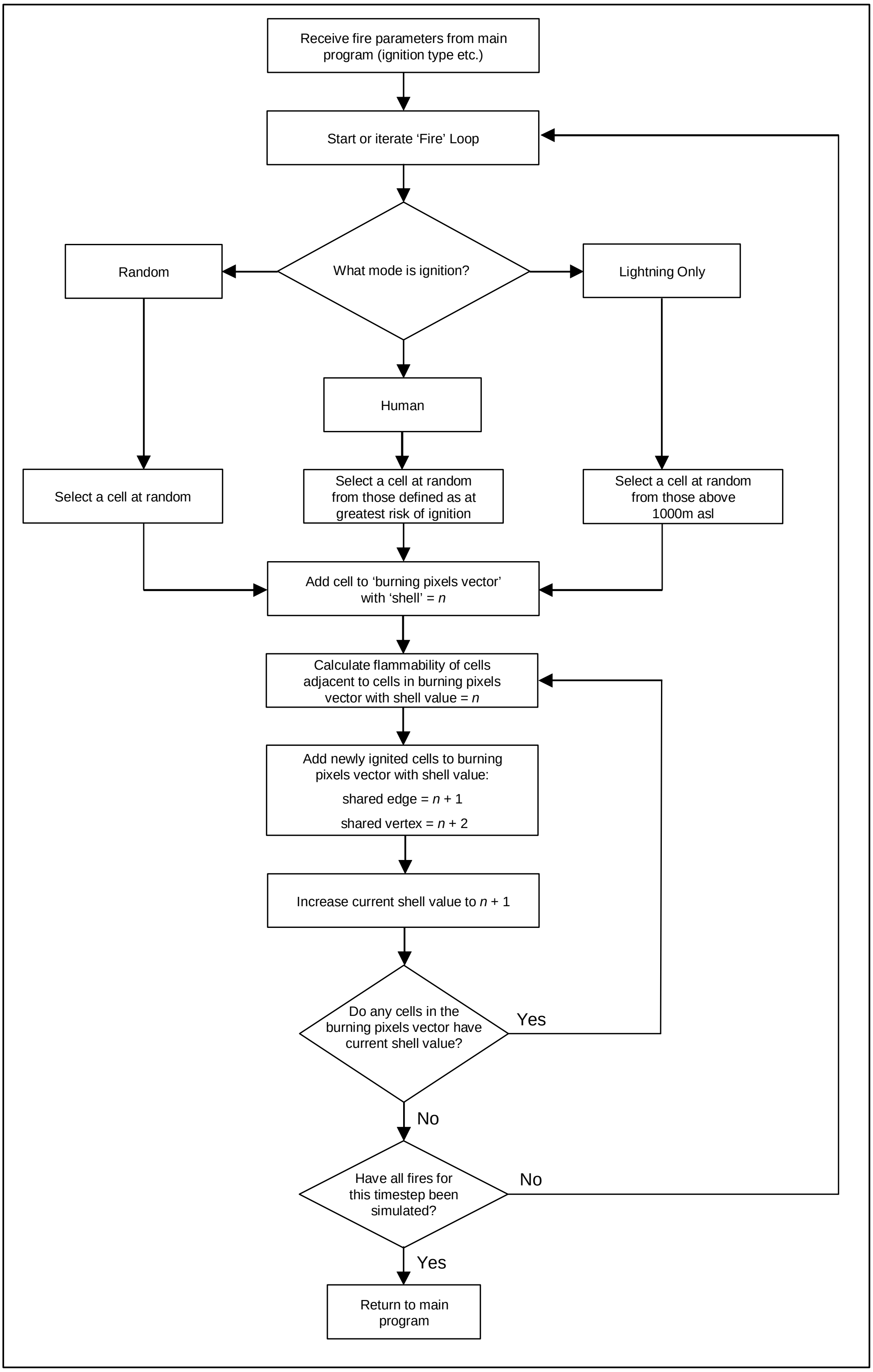

Figure 4.7 Schematic diagram of the cellular automata-based fire model. Starting from the top of the diagram, this process is iterated for as many fires as required for the timestep.

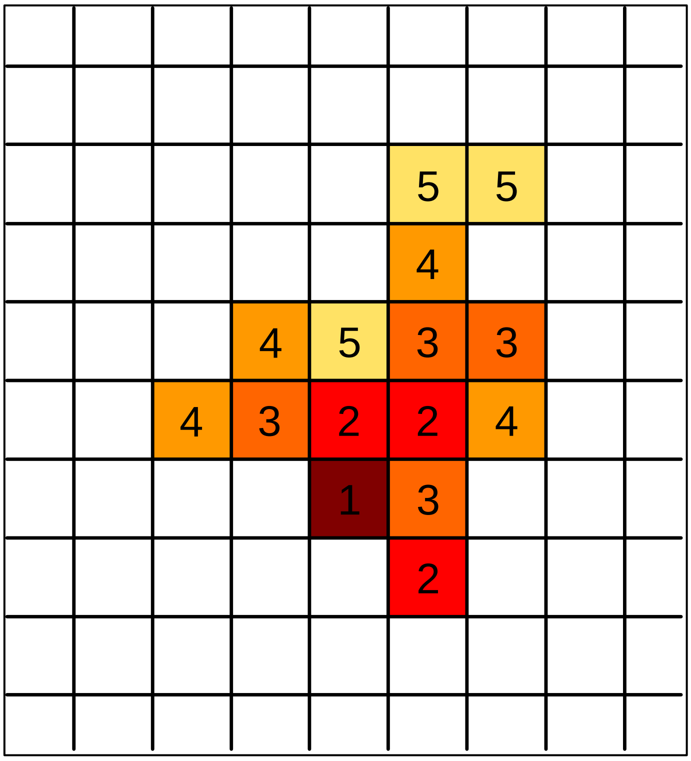

Figure 4.8 Schematic diagram of fire shells as used in the fire-behaviour model. Shells are used to keep track of which cells are alight, and as a means to restrict fire size (for management scenarios).

Fire 'shells' are used to track which cells are currently alight; all cells that ignite during the same iteration of the fire procedure belong to the same shell (see Figure 4.8). Cells are assumed to burn for one shell before extinguishing, at which time the probability of spread to all adjacent cells is calculated and the presence/absence of spread tested (see below). If the current shell is n, an adjacent cell with a shared edge will have shell $n+1$ if spread occurs into it – if the cell shares a vertex it is added to shell $n+2$. In this way a time lag due is simulated for spread moving diagonally between cells (as the distance of spread is further). Fire may spread into any of the adjacent eight cells that contain land-covers that are deemed flammable (i.e. all land-covers except 'Urban', 'Water' and 'Burnt'). Fires are all assumed to burn at the same intensity such that land-cover of a cell is completely burnt. A test is made for fire spread by comparing a uniform random deviate in the interval [0,1] with the probability of ignition (spread) of the adjacent cell.

The probability of ignition of the adjacent cell (spread) is calculated as a function of the vegetation flammability probability, modified by slope and climate conditions (i.e. abiotic factors). The flammability probabilities of the different land-cover types (Table 4.6) should be interpreted as the probability of a cell with the given land-cover being ignited by an alight adjacent cell given flat ground (between -5% and 5% slope), vegetation moisture in the interval 0.5 – 0.6, and no wind.

Whilst studies have examined the flammability of Mediterranean vegetation (individual species) according to calorific value (Dimitrakopoulos and Panov 2001), time-to-ignition (Dimitrakopoulos and Mateeva 1998), and classified flammability more generally (Dimitrakopoulos and Papaioannou 2001), no studies are known to have considered the explicit probability of ignition (i.e. spread between cells) at the scale considered here (i.e. broad land-cover vegetation classes at the landscape level) for the cellular automata approach. Thus, land-covers were ranked in order of flammability according to these previous studies (Dimitrakopoulos and Mateeva 1998, Dimitrakopoulos and Panov 2001, Dimitrakopoulos and Papaioannou 2001) and flammability probabilities assigned (Table 4.6) and calibrated to ensure realistic wildfire behaviour (i.e. similar frequency-area distributions). Land-cover flammability probabilities are modified by weighted multipliers for categorised slope and local climate conditions. Wildfire is known to spread preferentially upslope due to flame height and vertical heat convection effects (Viegas 1998). Categories of slope (%) are classified and weighted following Perry and Enright (2002b, Table 4.6).

Table 4.6 Slope classes and fire ignition weights. Slope is calculated between neighbouring pixels. A weight is then multiplied to the existing ignition probability dependent upon the slope classes.

Slope (%)	*Weight*
-25 and below	0.80
-25 to -15	0.90
-15 to -5	0.95
-5 to 5	1.00
5 to 15	1.05
15 to 25	1.10
25 and above	1.20

Alongside vegetation type, vegetation moisture content (especially in small branches and leaves – the fine fuel moisture content) is known to be an important determinant of flammability, and is an important consideration in many fire danger rating systems (e.g. NFDRS) and fire simulation models (e.g. BEHAVE Burgan and Rothermel 1984). Vegetation with lower moisture contents is generally more flammable and harder to prevent spread into (Castro *et al.* 2003). Vegetation moisture is considered here by classifying cell values derived from Eq. 4.5 into five classes and assigning appropriate multipliers (Table 4.7).

Table 4.7 Moisture classes and fire ignition weights. A weight is multiplied to the existing ignition probability dependent upon the vegetation moisture class.

Moisture Class	Weight
0.2 and below	0.80
0.2 to 0.3	0.90
0.3 to 0.5	0.95
0.5 to 0.6	1.00
0.6 and above	1.05

As noted above (section 4.4.3), wind data for the study area are not available and so conditions (direction and strength) are generated at random for each fire simulated. Multipliers for direction of spread are shown for each of three possible wind strength classes in Figure 4.9. These multipliers are based on the results of previous, similar, cellular automata-based simulations of wildfire (Karafyllidis *et al.* 1997, Perry and Enright 2002b).

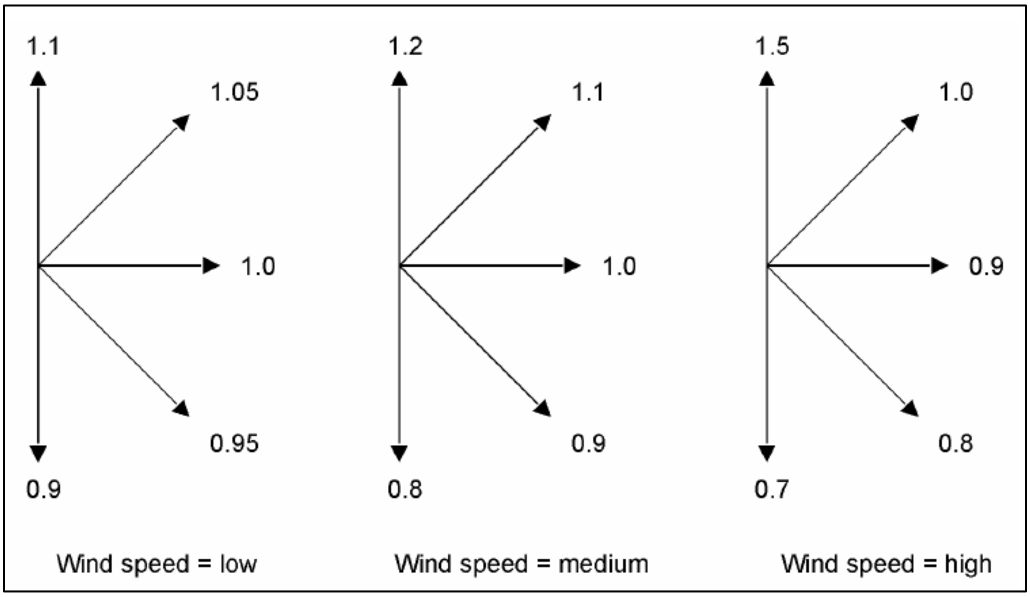

Figure 4.9 Fire ignition weights as a function of wind speed and direction. The weight is a multiplier of the existing ignition probability.

Fire size can be restricted in the model in two ways. First, 'fire breaks' may be defined in the landscape. Fire-breaks, lines of land cleared of vegetation, are often used by fire managers in an attempt to restrict the growth of fires by removing fuel available to burn. Fires in the model will not spread into (and therefore across) any cell defined as containing a fire-break. Roads also act as fire breaks in the landscape. Second, fire sizes may be restricted dynamically at model run-time by assigning a fire size beyond which fire spread is restricted. This option may be used to represent fire-fighting activities, or to investigate how changes in the maximum fire size affect the fire regime more broadly. The maximum burn size is defined in the model in terms of shell number (see Figure 4.8) – upon reaching this shell an additional multiplier (arbitrarily set at 0.8) reduces total flammability at each subsequent shell, leading fires to extinguish rapidly.

4.5.4 Summary

The model developed and presented here uses a combination of the Poisson distribution and consideration of climatic (i.e. temperature and precipitation) and human influences to simulate fire ignition. A cellular automata approach is used to simulate spread, considering vegetation flammability, slope, vegetation moisture and wind as factors influencing fire behaviour. The relative importance of these factors on land-cover composition, and the parameters used to represent vegetation-dynamics (presented in section 4.4), are now explored

4.6 SENSITIVITY ANALYSIS

4.6.1 Introduction

Sensitivity analysis is undertaken at the stage in the modelling process when initial model construction is deemed complete. This analysis helps the modeller to verify that the implemented model structure produces expected behaviour given the underlying conceptual model, and to examine the importance of parameters used in the model. A parameter that causes large variation in a model's state variable for only a small change in its value is understood to be 'sensitive'. Emphasis should be placed on ensuring that an accurate estimate of sensitive parameter values is found. The majority of the parameters used in the model presented here are not based on empirical data and cannot be formally verified in the sense of finding their precise 'real world value'. The sensitivity analysis performed is therefore used here to verify that the model behaves as expected and to assess which aspects of the model contain most uncertainty. This analysis may also help to point towards aspects of Mediterranean vegetation-dynamics that require most attention at the landscape scale.

The sensitivity analysis undertaken here is a simple univariate approach. Each input parameter is varied by ±10% of its 'base' value in turn, maintaining all other parameters at their base value. The output value of the LFSM's state variable is then compared with its value when all parameters are at their base value. This method assumes that i) there is a linear relationship between the state variable and each parameter, and ii) there is no interaction between parameters as each value is systematically altered (Drechsler 1998). Drechsler (1998) highlights that this latter problem, when interactions between uncertain model parameters produce model output that varies in a non-linear fashion, is a particular problem for complex models of (complex) systems. Other more sophisticated methods, such as Monte Carlo analysis (e.g. Xu *et al.* 2004) and latin hypercube sampling (e.g. Xu *et al.* 2005), have been found to be useful for analysis of spatial landscape models. These methods are not used here because of data (e.g. lack of probability density functions for each parameter) and computational requirements. In the following chapters (seven and eight) alternative, more qualitative, methods of examining model performance will be assessed.

4.6.2 Parameter Testing

To examine the influence of parameters the proportion of landscape in the Holm Oak land-cover class after 500 timesteps was examined. This variable was chosen as the state variable for comparison of model performance as it represents (at least in the model) the 'climax' vegetation species in the region and therefore gives an indication of the 'maturity' of the landscape. Other landscape measures (i.e. the number of vegetation patches and Shannon's index of diversity for vegetation classes) were also considered but found to be correlated with the abundance of Holm Oak. Three model repetitions were made for each parameter, and the mean proportion of the landscape occupied by Holm Oak was compared with the proportion for the base values. Parameter values used for each run are shown in Table 4.8. Human ignition probabilities are not considered as these are a relative measure of human activity which is assumed not to change through time in the biophysical model.

Sensitivity indices have been derived for use in sensitivity analyses (e.g. Burgman *et al.* 1993, Hamby 1994). However, the most interpretable method for understanding the influence of each parameter is simply to look at the proportional change in the state variable for the given (i.e. ±10%) change in the parameter in question (Table 4.9).

Table 4.8. Parameter values used in LFSM sensitivity analyses. Values were varied by ±10% from their original values (tuned or specified from the literature. In the case of the parameter sets (e.g. Land-cover Flammability) all parameter values were varied equally. Values for Mean Annual Fires were varied to the nearest integer as required by the model specification.

Parameter	*Base Value*	*+10% Value*	*-10% Value*
Xeric Moisture Class	500	550	450
Hydric Moisture Class	1000	1100	900
Mean Annual Fires	5	6	4
Climate Ignition Risk	50	55	45
Land-cover Flammability	0.23, 0.23, 0.18, 0.22, 0.24, 0.22, 0.18, 0.15	0.253, 0.253, 0.198, 0.242, 0.264, 0.242, 0.198, 0.165	0.207, 0.207, 0.162, 0.198, 0.216, 0.198, 0.162, 0.135
Slope Risk	0.80, 0.90, 0.95, 1.00, 1.05, 1.10, 1.20	0.880, 0.990, 1.045, 1.100, 1.155, 1.210, 1.320	0.720, 0.810, 0.855, 0.900, 0.945, 0.990, 1.080
Vegetation Moisture Risk	0.8, 0.9, 1.0, 1.1, 1.2	0.88, 0.99, 1.10, 1.21, 1.32	0.72, 0.81, 0.90, 0.99, 1.08
Random Seed Dispersal	0.3	0.33	0.27
Oak Mortality Burn Frequency	200	220	180

Results (Table 4.9) show that i) increases in parameter values cause more change in the state variable than decreases, and ii) that the most sensitive parameters are fire-related. With the exception of ‘Xeric Moisture Class’, large changes in the state variable are a result of the increased contribution of ‘large’ fires to total burned area due to fire spread-related parameters. An increase in the boundary of what constitutes a ‘Xeric Moisture Class’ is seemingly important for successional processes specified by the conceptual time-to-transition model.

The relatively low importance of ‘Mean Annual Fires’ (controlling number of fires per year and ranked equal seventh) shows the proportion of different land-cover types is much more sensitive to parameters controlling fire spread (i.e. the size of fires) than fire ignition. Parameters controlling processes of succession are also consistently ranked lower than fire spread parameters. The high sensitivity of fire spread-related parameters in models that take such a percolation-type (cellular automata) approach has been observed previously, particularly with regard to vegetation flammability values (ranked as most sensitive here – Ratz 1995, McCarthy and Gill 1997, Perry and Enright 2002b). The importance of land-cover flammability probabilities is an indication of the presence of a critical range of values which produce system behaviour somewhere between events that consistently span the entire grid and events that consistently burn very few grid cells.

Table 4.9 LFSM sensitivity analysis results for land-cover composition. Resulting land-cover proportions for Holm Oak are presented with percentage change due to each treatment (PD) – parameters for which a ±10% change in value causes a change of greater than ±15% in the state variable are shown in bold. Parameters are ranked according to their sensitivity (total change in state variable), in descending order of sensitivity.

Parameter	*Mean (Base)*	*Mean (+10%)*	*Mean (-10%)*	*PD (+10%)*	*PD (-10%)*	*Sensitivity Ranking*
Xeric Moisture Class	0.662	**0.516**	0.789	**-0.22**	0.12	5
Hydric Moisture Class	0.662	0.709	0.618	0.07	-0.07	6
Mean Annual Fires	0.662	0.682	0.635	0.03	-0.04	7
Climate Ignition Risk	0.662	**0.644**	-0.027	**0.65**	-0.02	4
Land-cover Flammability‡	0.662	**0.120**	0.626	**-0.82**	-0.06	1
Slope Risk‡	0.662	**0.154**	0.654	**-0.77**	-0.01	2
Vegetation Moisture Risk‡	0.662	**0.189**	0.627	**-0.72**	-0.05	3
Random Seed Dispersal	0.662	0.643	0.670	-0.03	0.01	9
Oak Mortality Burn Frequency	0.662	0.678	0.695	0.02	0.05	7

‡Final land-cover proportion is the result of taking a running mean with lag = 2 due to highly variable nature of annual land-cover proportions (many very large fires)

The sensitivity of the four main fire parameters highlights the importance of ensuring that the most appropriate values for these parameters are found and used. Ensuring the accurate values for these parameterisations is troublesome however, because of the difficulty of translating data collected for the parameterisation of models such as the Rothermel (1972) semi-empirical model (e.g. fuel surface-area-to-volume ratio, daily humidity) for use in cellular automata. The values presented in Tables 4.6, 4.7 and 4.8 above were chosen via a combination of parameter tuning (to mimic observed wildfire regime characteristics – discussed further in chapter six), mimicking previous modelling approaches in the literature, and so that mechanistic representation is consistent with other literature.

4.6.3 Summary

The sensitivity analyses presented here were used predominantly to check the functioning of the LFSM was as expected given the nature of the model structure, but also to identify the most sensitive (i.e. influential) parameters. Results suggest the fire spread parameters are the most sensitive, with greatest influence on land-cover composition and wildfire regime behaviour. The high sensitivity of this type of parameter has been found in previous studies (e.g. Ratz 1995, McCarthy and Gill 1997, Perry and Enright 2002b).

4.7 SUMMARY

This chapter has presented previous conceptual and simulation approaches to model Mediterranean-type vegetation-dynamics. Based on this review the structure of the Landscape Fire Succession Model was detailed and justified. Specifically, the vegetation-dynamics module uses a rule-based modelling approach at the community level that considers successional attributes (succession pathway, seed sources) and environmental conditions (water and light availability) within the traditional conceptualisation of competition for resources between 'seeder' and 'resprouter' vegetation (e.g. Figures 4.2a and 4.2b). The review of Mediterranean-type vegetation-dynamics highlighted the importance of disturbance – these are predominantly fire and human activity. The modelling of human activity is considered in the next chapter. To establish an appropriate representation of fire, previous approaches to modelling individual fires and their regimes was presented. Of the several approaches available, a modified version of the basic 'forest fire' cellular automata represents fire spread. Land-cover flammability probability, slope, vegetation moisture and wind are considered as factors influencing individual fire behaviour. Fire ignition is modelled considering both climatic and human influences.

A sensitivity analysis of the parameters used in both the vegetation-dynamics and wildfire modules showed the fire parameters to be most sensitive and influential on land-cover composition and wildfire regime behaviour. Although difficult to quantify accurately, these model parameters were tuned to effectively mimic observed wildfire regime behaviour. The sensitivity analysis also verified the correct functioning of the LFSM (relative to that intended by its construction).

CHAPTER FIVE

AGENT-BASED MODEL OF LAND-USE/COVER CHANGE

5.1 INTRODUCTION

In similar fashion to the ecological modelling approaches outlined in chapter four, it is generally recognised that the problem of modelling spatial location and allocation of rural land-uses (i.e. finding the optimal location for land-uses) may be approached from two perspectives. The first addresses broad-scale regional land-use patterns resulting from unknown decisions made by individual actors within the region of interest, the second by considering the decision-making process of individual decision makers (Kellerman 1989). These two distinct perspectives result from opposing philosophical attitudes toward modelling the processes of change. The former assumes that general theories of agricultural location can be developed ‘from the top-down’ according to knowledge about the state of a whole region. The latter perspective considers understanding individual actors’ actions ‘from the bottom-up’ to be necessary, as global patterns emerge from these local decisions which are made based on knowledge about the state of a limited part of a larger region. Thus, the ‘top-down’ approach attempts to produce general laws for the optimal spatial orientation of land-uses in a region regardless of the context of individual circumstances, whilst the ‘bottom-up’ approach examines processes within the context specific to the region and locales within it.

Simulation modellers and social-scientists have recently begun to approach the task of understanding and forecasting the human shaping of landscapes, particularly the nature of agricultural land-use and decision-making, by using ‘bottom-up’ actor-oriented approaches at detailed levels of spatial, temporal and decision-making resolution. The contextualisation of local decision-making about land-use is likely to be an important consideration for any LUCC modelling in the traditional Mediterranean landscapes that SPA 56 characterises. Agricultural planning has generally not been explicitly dictated from central planning officials in the Mediterranean Basin (although EU initiatives such as the Common Agricultural Policy have been used as incentives to influence decision-making) and the highly spatially heterogeneous nature of the landscapes produces a high diversity of local decision-making contexts.

The 'bottom-up' approaches discussed here all consider the basic unit of organisation to be some form of 'agent' which may interact with other agents and their environment. As a relatively new form of modelling framework, the methods required to develop 'agent-based' decision-making models are still under development and, as with much simulation modelling, a 'motley' assortment of approaches is often assembled (Winsberg 2001). Becu *et al.* (2003) suggest that there are two broad approaches available to develop representations of individuals' behaviour in agent-based models – a) by using previously established theoreticl frameworks from the social or computer sciences, or b) by eliciting behavioural rules directly from stakeholders' understanding of their own behaviour and social system within which they act. A combination of these approaches is taken here; established 'top-down' Agricultural Location Theory (ALT) is complemented by elicitation of agent behaviour specific to the study area to produce a coherent agent-based model of agricultural decision-making. Recent developments in agent-based approaches are now discussed, preceded by a brief outline of the origins of ALT on which much of these models are based. The model structure is presented before testing and sensitivity analysis.

5.2 Agricultural Location Theory

Classical Agricultural Location Theory (ALT) aims to explain the spatial location of agricultural land-uses and practices. The birth of classical ALT is attributed to von Thünen's 1826 study (see Hall 1966) conceptualising an isolated agricultural state (region). The theory developing and extending from this simple conceptual model has formed the basis of much later work and theory (e.g. Dunn 1954, Chisholm 1962). Many authors have since described, summarised and evaluated the assumptions, utility and importance of von Thünen's model (e.g. Chisholm 1962, Kellerman 1989, O'Kelly and Bryan 1996) which, in essence, examines the spatial outcomes of the relationship between crop value and crop-to-market transport costs. Assuming perfect knowledge of prices and costs, perfect economic rationality on the part of producers (i.e. the product that provides greatest profit is always chosen), and holding all other environmental factors constant (including soil fertility, labour availability etc.), von Thünen showed that generalised land-uses with varying economic yields and transport costs form concentric rings around a single central market (Figure 5.1). By conceptualising the competition for land-use in this manner von Thünen introduced the idea of *economic rent* – "the return that can be obtained [from land] above that which can be got from the land which is at the margin of economic cultivation" (Chisholm 1962). Chisholm (1962) goes on to highlight the importance of economic rent as the concept that resolves land-use competition and produces the observed spatial concentric distribution of land-uses. It is important to highlight here that the emphasis of von Thünen's model is on spatial location (and its corollary, distance to market), and that it assumes perfect economic rationality and environmental homogeneity.

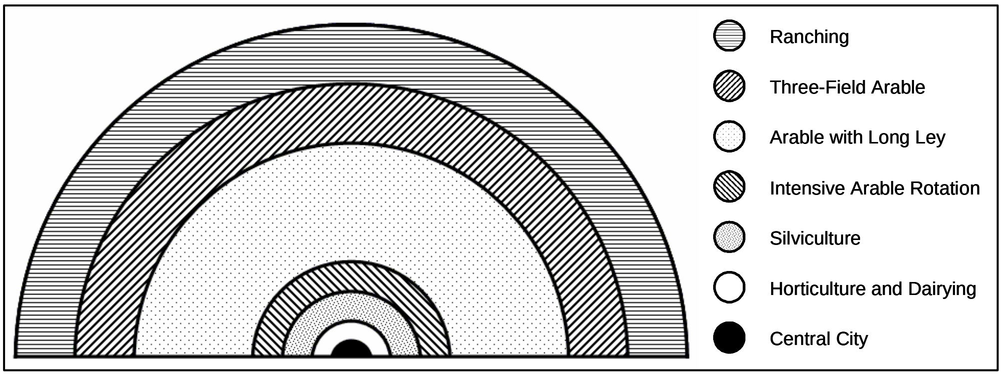

Figure 5.1 von Thünen's classical model of agricultural location. With all other environmental factors held constant, generalised land-uses with varying economic yields and transport costs produce a concentric ring pattern around a central market.

Chisholm's (1962) book *Rural Settlement and Land-use: An Essay in Location,* is widely recognised as the seminal theoretical starting point for the study of the spatial distribution of contemporary land-use. Chisholm introduced von Thünen's theory and method into the analysis of rural geography to explain spatial patterns of agricultural land-use by examining relative and absolute locations and distances of different production entities at multiple organisational levels and spatial scales. Applying the concept of economic rent (outlined above) and considering the importance of location and distance, he examined the importance of the spatial relations between the farmstead (i.e. the farm house and stores) and the village, between the (entire) farm and the village, and between agricultural regions and nation states. In this analysis the relative importance of transport costs versus crop yields is the prominent determinant of the allocation of land-uses.

Many critics have noted the "enduring methodological significance" of Chisholm's work (Cliff *et al.* 1997, p.206), but also highlight the incompleteness of such a spatially-dependent theoretical framework (e.g. Moran 1994, Munton 1994). Munton (1994) highlights that there are many other important factors alongside transport costs associated with the distance of a location to a market in determining the spatial allocation of agricultural land-uses at both global and local scales. Geopolitical and international trade agreements, and the structure and functioning of global food markets will be important at broader scales, whilst financial support of the local rural economy (e.g. via agricultural subsidies) and historically contingent determinants of local agricultural production (e.g. land-tenure history, traditional agricultural practices) will be important at local scales. Thus, Munton (1994) suggests that the universal importance of location (and distance to market) in Chisholm's work must be contextualised by pertinent aspects of local production in the region under study. This is a statement that should not be underestimated when considering new, and increasingly favoured, 'bottom-up' approaches to modelling spatial land-use distribution and its change.

This need for increased contextualisation in theories and models explaining the spatial distribution of rural land-uses is not a recent observation, however, and this, amongst other criticisms of this type of 'top-down' approach, was voiced by Harvey (1966) soon after the publication of *Rural Settlement and Land-use*. Chisholm's approach was not 'top-down' in the sense that it could be related to centrally unplanned situations (Chisholm 1994). However, it did not consider the behaviour of individual actors in their own context and, further, considered the behaviour of all actors in the region to be uniformly, perfectly economically rational. Making these points Harvey (1966, p. 370) summarised, "the only way we can understand regional variations in agriculture will thus be through an understanding of decision-making processes; and decisions are never simply economic ones". It is from these criticisms, and this perspective, that recent 'bottom-up' (i.e. agent-based models) approaches have emerged to consider individual agents' behaviour.

One Agent-Based Model (ABM) was recently developed in an attempt to reproduce the predictions of von Thünen's theory (Sasaki and Box 2003). Whilst Sasaki and Box (2003) did successfully reproduce the concentric ring pattern expected from von Thünen's 'top-down' conceptualisation, their model still deals with idealised agents acting in a decontextualised environment and so does not address many of the criticisms raised by Harvey (1966). Namely, agents are influenced exclusively by economics, are perfectly rational, and have perfect knowledge regarding the state of the market. Advances in resolving these idealisations so that they are more realistic have been made, and will be discussed in the next section. However, such theoretical advances are proving demanding in terms of the data needed to parameterise the models produced. In turn, this demand for data is raising questions about how the modelling process should proceed, particularly regarding the representation of agents, 'participatory' approaches to modelling and model validation. These questions are addressed in the following sections.

5.3 AGENT-BASED MODELLING

5.3.1 Introduction

Modelling techniques taking a bottom-up approach, considering some form of agent as the basic unit of organisation, are known in LUCC literature interchangeably as Agent-Based Models (ABMs, and thus ABM/LUCC, e.g. Parker *et al.* 2002) or Multi-Agent Systems (MAS, and thus MAS/LUCC, e.g. Parker *et al.* 2003). However, on closer inspection, Hare and Deadman (2004) suggest that there are subtle differences between the terms ABM and MAS due to differences in their origin – ABMs arose from artificial life research and have been primarily used to simulate macro-scale behaviour through the simple interaction of micro-scale components, whilst MAS refers to systems of many agents that are autonomous, can communicate and interact with other agents, may respond to their environment and are goal-driven. These agents are not necessarily individual human actors, but might also represent an organisational decision-making unit at another level, for example a household or village. Note that the definition of ABMs does not deny that agents can interact with one another or their environment. Relying on the above definitions suggests that the term used will depend on the specific focus of the study at hand – if broad patterns of LUCC are to be examined an ABM is used, but if the system of agents and how it produces patterns of LUCC are themselves the subject of study then a MAS is used. This terminology is used here, but as its use in the LUCC literature has been loose, some liberties will be taken to follow the terminology used by the authors cited.

Several in-depth reviews of the use of ABM/LUCC have recently been published (Parker *et al.* 2002, Parker *et al.* 2003, Bousquet and Le Page 2004), highlighting the increasing interest in their use. Parker *et al.* (2003 p.314) describe MAS/LUCC models as,

> *"combin[ing] two key components into an integrated system. The first component is a cellular model that represents the landscape over which actors make decisions. The second component is an agent-based model that describes the decision-making architecture of the key actors in the system under study. These two components are integrated through specification of interdependencies and feedbacks between the agents and their environment."*

With this model conceptualisation several representational improvements over the 'top-down' approaches described in the previous section can be highlighted. First, the approach is process-based and considers the behaviour of the actors making the decisions that will influence land-use distributions. Furthermore, by considering agents acting within a cellular (grid) landscape this approach allows the consideration of linkages and interaction between socio-economic and physical environment processes and phenomena. Consequently agents' behaviour, and in turn the distribution of land-uses, is interpreted in a spatially-explicit manner across a range of scales (depending on what actor the agent represents and the size of the grid cells representing the landscape). This spatially-explicit and agent-based nature of ABM/LUCC means the biophysical and socio-economic processes and organisations being represented can be examined at appropriate scales. All three points contribute to improving the 'contextualisation' of the model with respect to the region under study and are now explored in more detail. This contextualisation is particularly useful as it allows the representation of impacts of heterogeneous spatial decision-making conditions on individual land holders' decisions, a consideration likely to be important in highly fragmented Mediterranean-type landscapes. However, as discussed below, while offering many perceived improvements over 'top-down' ALT approaches, many questions and issues raised by the use of such models remain open.

5.3.2 Recent Agent-Based Modelling of Human-Environment Interaction

Many of the recently published studies exploiting these benefits of the ABM/LUCC approach have addressed tropical regions of the world (Evans *et al.* 2001, Deadman *et al.* 2004, Huigen 2004, Manson 2005). Whether this is because LUCC in tropical forest landscapes is usually a direct shift from forest to agriculture and therefore relatively straight-forward to model, or due to other issues, is not clear. Evans *et al.* (2001) and Deadman *et al.* (2004) both modelled land-use change in the Amazon with agents representing households. Evans *et al.* (2001) make limited use of the potential to integrate the biophysical environment into agents' decision-making, focusing instead on household composition (via human fertility/mortality rates) and market conditions (e.g. crop prices). Deadman *et al.* (2004) consider soil and burn (for land clearance) quality to be key factors in the decision-making process. However, it seems that in these environments (where LUCC is mainly due to large areas of untouched rainforest that are being converted to agricultural land-use) biophysical factors (only) weakly constrain human decision-making. Huigen (2004) presents a MAS/LUCC modelling framework with reference to a study area in the Philippines. He puts much greater emphasis than has been previously discussed here or attempted elsewhere on the social context of the actors being represented, attempting to "transform real-life stories [from locals] into a computerised model via behavioural and decisional conceptual theories, empirical data and intuitions" and focussing on ethnicity as a key factor in land-use dynamics (Huigen 2004 p.6). Other modelling efforts have utilised an MAS approach to consider how agricultural decisions are made in Germany (Balmann 1997), Chile (Berger 2001) and Senegal (Barreteau *et al.* 2004). Rather than focusing specifically on how land-use changes, these modelling exercises have concentrated efforts on examining the interactions between actors, institutions and other organisational structures and how they contribute to decision-making in the agricultural sector.

Manson (2005) however, places greater emphasis on environmental interactions with actors in his actor-institution-environment modelling framework to consider the Southern Yucatán Peninsular Region (SYPR) of Mexico. The model considers soil fertility, elevation, slope, aspect and precipitation as factors influencing land-use decisions. Specifically, soil fertility is modelled as being dependent upon the type of and duration of land-use and therefore subject to actors' decisions. Reciprocally, these decisions are assumed to be dependent upon soil fertility. Manson (2005) also utilises another potential of ABM/LUCC approaches by considering the interaction of agents at two organisational levels – smallholder households and institutions (administration units, financial markets and conservation organisations). An *et al.* (2005) also consider human-environment interactions more explicitly than previous models by tracking the life-histories of individual actors in their model to explore the impacts of household dynamics on Panda habitat in China. The interaction between forest growth and harvest is represented in a spatially-explicit manner based upon household locations and sizes. The authors found several non-linear and counter-intuitive landscape responses to the conservation scenarios considered with potential policy implications. The Framework for Evaluation and Assessment of Regional Land-use Scenarios (FEARLUS) developed at the Macaulay Institute (Macaulay Institute 2004) also allows for the explicit consideration of agent interaction with the biophysical environment. However, to date this modelling framework has not been applied to a real world landscape.

5.3.3 Current ABM/MAS Issues

5.3.3.1 Introduction

Computer simulation models representing agents and their interaction have arisen in the past two decades primarily due to increases in computer-processing power making the approach possible. The implementation of these models to applied (spatially-explicit) situations has also only been possible because of the availability of high-resolution spatial data (often collected by satellite sensors). Model development has been further eased, leading to a greater number of these models being developed, by the recent availability of specialised program libraries (e.g. SWARM 2005), modelling environments (e.g. NetLogo – Wilensky 1999) and object-oriented programming languages (e.g. C++). However, whilst these improvements often make for conceptually and visually impressive modelling capabilities, many questions remain to be addressed for agent-based model development. The three main issues relate to data, agent representation, and model validation, and are considered now in turn.

5.3.3.2 Model Data

Although the availability of high-resolution spatial data has increased dramatically in past decades with the introduction of global satellite remote sensing coverage, data issues still remain. For example, Baulies and Szejwach (1998) highlighted that at the time of writing the level of data concerning the extent, rate and direction of LUCC dynamics provided by these sensors was adequate, but that functional understanding and adequate parameterisation was lacking. Data collected by satellite remote sensors are primarily related to biophysical properties of the Earth's surface and not socio-economic properties and the availability of socio-economic data at a sufficient spatial resolution is conspicuously sparse. Such a paucity of data has led some authors to suggest a need for "socialising the pixel" and "pixelising the social" of remotely-sensed data (Geoghegan *et al.* 1998). By this the authors mean that remotely sensed data need to become more amenable to interpretation in the context of socio-economic processes and phenomena relevant to LUCC. Much contemporary socio-economic data are aggregated at levels related to organisational units of data collection (e.g. municipalities, census tracts, etc.) and frequently do not match the spatial resolution of available biophysical data, as shown by the empirical modelling in section 3.3. Improvements in availability of high-resolution spatial data for the socio-economic aspects of LUCC modelling are needed to allow adequate parameterisation of future models.

5.3.3.3 Agent Representation

Availability of parameterisation data is an issue that also hampers the development of conceptual architectures for the appropriate representation of agents and their behaviour. When the lowest unit of model representation is the individual actor, understanding of individual human behaviour is required. Many formal models of human behaviour in social science have been developed by economists, which have (as with ALT discussed above, section 5.2) conventionally been based on the (fictional) perfectly economically rational human species *Homo economicus*. *Homo economicus* acts to optimise wealth above all else (Janssen and Jager 2000). Goodrich *et al.* (2000 p.83) however, highlight that “[r]ationality is not tantamount to optimality”, and that in situations where information, memory or computing resources are not complete (as is usually the case in the real world) the principle of *bounded rationality* is a more worthwhile approach. Simon (1957) recognised that rarely do actors in the real world optimise their behaviour, but rather they merely try to do ‘well enough’ to satisfy their goal(s). Simon (1957) termed this non-optimal behaviour ‘satisficing’, the basis for much of bounded rationality theory since. Thus, satisficing is essentially a cost-benefit tradeoff, establishing when the utility of an option exceeds an aspiration level (Goodrich *et al.* 2000). The formal logic of how such a decision-making processes has been examined with much work on the development of ‘intelligent machines’ following from it. In the context of LUCC, Gotts *et al.* (2003) examined the importance of the aspiration level on land-use choices, finding that the optimal level is influenced by environmental heterogeneity (in this case represented as an abstract parameter influencing land productivity). Most recently Evans *et al.* (2006) performed laboratory experiments to examine resource allocation decisions in abstract landscapes. These authors’ results highlight the importance of considering heterogeneous and non-maximising agents in agent-based LUCC models.

Whilst this idea of bounded rationality seems to fit more comfortably with how decisions in the real world are made, its application to representative models still poses many questions. As Gotts *et al.* (2003) examined in an abstract manner, the first set of considerations a modeller needs to make is related to what actors in the system consider to be their aspiration threshold and to what degree of variation it is open to. This decision is likely to depend on a number of factors in turn dependent upon the values and characteristics of the individual actor. Does the actor put financial gain above all else? Is the actor a risk-taker or a conservative decision-maker? Are there other traditional, ethical or other values that will affect the individual actor? These questions all relate to the goal(s) of the actor. The second consideration is what options are open to the actor? Further, will these options be constant in time? What about innovation (options not available now that might be 'created' by the actors)? Will all options be open to all actors? Finally, the modeller needs to consider how actors measure the utility provided by each option. This final consideration is related to what the actor considers to be their goal(s). Thus, there is clearly a large amount of data needed to implement a multi-agent model of boundedly rational decision-making, much of which will not be readily available and will need to be collected specifically for the modelling at hand. Furthermore, the more complicated the decision-making process, the greater the demand for data to parameterise agents' behaviour will be.

The main source of this required information and data is likely to come from interviews and surveys of the actors being represented in the model, methods which have been utilised in previous modelling exercises (e.g. Castella *et al.* 2005a). Castella *et al.* (2005b) developed a Role-Playing Game (RPG) to develop knowledge and understanding about the interactions between actors and their decision-making process that were then used to construct an ABM/LUCC in the northern mountains of Vietnam. RPGs have also been suggested and used elsewhere (Barreteau 2003, Barreteau *et al.* 2003, D'Aquino *et al.* 2003) as a tool to put stakeholders in realistic decision-making situations that can then be observed, and also as a communication tool. However, Barreteau *et al.* (2003) note the high resource demands of such using such a tool, and also the difficulties of developing an RPG. Such a high-resource demand prevented the development of such a process for data collection in this thesis. Instead, interviews with key local actors were undertaken to derive an appropriate model structure and time was invested in examining the potential use of participatory approaches for model ‘validation’ or assessment subsequent to model development (see below and chapter eight).

Bousquet and LePage (2004) suggest that interesting questions regarding the theoretical development of modelling the LUCC decision-making process are still emerging. For example, should research proceed by developing theoretical models and testing their applicability to different situations (as advocated by Parker *et al.* 2003)? Or, should a more inductive approach be taken with a wide variety of models designed for specific case studies examined for common elements (as advocated by D'Aquino *et al.* 2003)? Questions of actor learning and agency are still open and an area in which Bousquet and LePage (2004) suggest LUCC modellers could learn much from computer science. Finally, as discussed in the next section, the theoretical issue of how to validate MAS/LUCC models continues to command attention.

5.3.3.4 Model Validation

Parker *et al.* (2003) distinguish between model verification and validation as ‘building the system right’ and ‘building the right system’, respectively. Thus, validation is a procedure (or set of procedures) for assessing the appropriateness of model structure, assumptions and parameters values for representing the real-world, of which comparison of model output with outside data and expectations is only one option. Acquiring the data by which to parameterise an agent-based model, or for comparison with model outcomes, may be troublesome. The difficulties of putting a number on an inherently qualitative concept, for example trying to quantify attitudes toward drugs, are well known to modellers attempting to represent social behaviour (Agar 2003). In general, the consensus amongst the ABM/LUCC community seems to be that all possible validation techniques for a model should be examined and not only more traditional methods (Parker *et al.* 2003, Bousquet and Le Page 2004, Manson 2005). Such a perspective matches Winsberg’s (2001) epistemology of simulation, one component of which states that simulation models are autonomous. That is, their architectures must be internally justified by evaluation on a variety of fronts, from theory through empirical generalisations to personal experience of the modeller and other interested parties. Other simulation modellers have suggested the comparison of different model of the same process (model alignment or ‘docking’, Axtell *et al.* 1996) or of multiple implementations of the same model (model replication, Edmonds and Hales 2003), to improve understanding of model (and system) behaviour and, potentially, reveal 'hidden' parameters in the model structure. For MAS/LUCC models the range of evaluation methods often demands the comparison of model results and assumptions with results of other types of model and with data collected by surveys, interviews and censuses (Bousquet and Le Page 2004, Manson 2005). As mentioned in the previous section, and as will be explored again in more detail in chapter seven, much of this validation in MAS/LUCC models will come through 'participatory' approaches (e.g. Barreteau *et al.* 2001).

5.3.3.5 Summary

The growth and increased use of distributed agent-based modelling systems to represent human-environment interactions has brought three main issues to the fore; data, agent representation and model validation. As increases in the types and volumes of data available enable the diversity of approaches to agent representation to grow (via new model architectures), new techniques and methods by which to assess model validity will be required. The inclusion of new sources of knowledge to model construction and assessment procedures will enable this.

5.3.4 Implications for Modelling Mediterranean Human-Environment Interactions

The conclusion reached at the end of any human decision-making process will inherently (whether via explicit consideration or implicit influence) be shaped by the environmental setting and context of the decision-maker. In the case where the biophysical environment and land-tenure structure are highly spatially heterogeneous, such as in the Mediterranean Basin, local decision-making contexts are likely to be highly variable within a single region. Thus, even if actors are assumed to be perfectly economically rational, they are unlikely to act uniformly or make the same decisions across the whole study area. If actors are not assumed to make decisions solely (or perfectly) on economic grounds, decisions and actions will be increasingly varied spatially, dependent upon the objectives of the actor. Further, as an actor's decision-making context changes in time, the actions taken and decisions made will likely vary accordingly.

For example, in areas of highly fragmented and dynamic land-tenure, actors' decisions are likely to be influenced at any point in time by the configuration of their land. As this changes so too will decisions and actions taken regarding the use of land, which in turn will influence future biophysical properties and land-tenure orientations. In regions such as SPA 56 where the characteristic land-holding size is not large (Figure 5.2), spatial configuration is likely to become increasingly important as individual land owners' land-use decisions will affect small, localised patches within the entire landscape mosaic and LUCC will occur at a fine spatial scale (grain). This fine scale change makes understanding the impacts of changes on spatial processes, such as seed dispersal and wildfire, more difficult because of the higher number of potential landscape (spatial) states that may be produced. In areas with small land-holding sizes, composition also becomes important – as the number of different types of land owners increases, so too does the potential number of different decisions that may be made. If these decisions are context-dependent, (i.e. dependent upon both the composition of neighbouring owner types and the configuration of land holdings), the number of unique decisions possible rises exponentially.

Figure 5.2 The fragmented and heterogeneous nature of SPA 56. The aerial photograph spans approximately 1.6 km (1 mile) and contains numerous land-use and cover types including pasture, crops and urban areas.

William Odum (1982) highlighted the importance of these "small decision effects" on wider environmental issues and management. The "tyranny of small decisions" as it has also been termed results when broad-scale 'decisions' arise *post hoc* from the accretion of small decisions, often resulting in solutions and results that are not necessarily optimal for society or the environment. In the case of SPA 56, an important issue ('big decision') is the sustainable maintenance of the relationship between fire, vegetation, and human activity in the landscape as a result of land-use decisions made on an individual basis ('small decisions'). To ignore the potential effects of these small decisions on the wider environment could prove costly. Equally, when studying effects of these small decisions, the reciprocal effects of changes in the wider environment (e.g. the wildfire regime) upon them should not be omitted. The importance of recognising these feedbacks and relationships in models of land-use change is widely understood (Parker *et al.* 2002, Parker *et al.* 2003). In this thesis, appropriate investigation of the reciprocal nature of 'big' and 'small' decisions will only be possible with a spatially-explicit model, both in terms of the cellular representation of the landscape and the spatial 'awareness' of the agents.

5.3.5 Summary

As the land-use decision-making process is largely a function of environmental constraints and actor contexts, regions with spatially heterogeneous and temporally dynamic biophysical and socio-economic environments demand consideration of the varying context in which decisions are made. SPA 56 exhibits both a highly spatially heterogeneous and a temporally-dynamic nature in terms of its biophysical resources (e.g. vegetation, soil type) and land-tenure structure. Recognising the relationships and feedbacks between the nature of these environments and the decision-making process is important when modelling LUCC (Parker *et al.* 2002, Parker *et al.* 2003). However, of the agent-based models of LUCC (ABM/LUCC) examined here, very few explicitly represent interactions with the biophysical environment, and none seem to explicitly consider the spatial nature of land-tenure on decision-making. This representational approach is likely to exist largely because of the broadly homogeneous nature of the environments and type of LUCC examined (e.g. tropical deforestation to agricultural land-use). Such an omission should not be made in Mediterranean landscapes, where stronger and more varied cultural and biophysical legacies have produced more complex contemporary land-use scenarios.

However, whilst bottom-up, agent-oriented modelling offers greater representational power, it comes at a cost. Representing individual agents generally requires more data, and presents new model validation issues regarding agent behaviour. Specifically, qualitative data regarding actor intentions must be transformed into computer code – these data must be collected and the method of transformation justified. More detailed representation of agents' behaviour demands more data and a more complex model structure, leading to decreased tractability of results. As with all modelling exercises, the following (inter-related) requirements must be met: a level of model complexity sufficient to represent the system adequately; a sampling strategy capable of providing sufficient parameterisation data to allow the application of the proposed model structure; and production of model results that are interpretable and understandable relative to the implemented structure. Efforts to achieve these requirements are currently generating new and innovative methods to data collection and model analysis and validation.

5.4 ABM/LUCC STRUCTURE

5.4.1 Introduction

The broad aim of this ABM/LUCC is to represent agricultural land-use changes that are driven by farmer choices and decisions spatially. Interviews were undertaken with local agricultural actors (i.e. farmers, agricultural organisation officials) in the study area in November 2005 [N.B. these interviews were undertaken during model construction and are not the same as those presented in chapter eight which were made upon completion of the first iteration of the model-building process]. As highlighted above, interviewing examples of the actors represented in an ABM is a useful way of developing system understanding to ensure appropriate representation. Semi-structured interviews were used to elucidate understanding regarding individuals' land-use goals, land-use options, and land tenure markets. Interviews were conducted in Spanish and were audio recorded allowing the relevant sections to be transcribed and presented below. Arising from these interviews, a primary distinction is made in the model between two different 'types' of farmer that are representative of two different worldviews – commercially-minded ('commercial') and traditionally-minded ('traditional') farmers. These two types of agent (i.e. model representations of real world actors) take distinct land-use decision-making approaches to establish whether an area of land will be crops, pasture or non-agricultural land. Agents consider the landscape as a grid of finite land units (i.e. pixels) on a seasonal basis (i.e. four timesteps per year). Pixels with orthogonal neighbours either in the same land state or owned by the same agent are considered to be pixel 'clusters' (i.e. farmers' fields). The status of each agent (e.g. age, wealth, etc.) is monitored at each timestep. A summary of the model procedure is presented as a flow diagram in Figure 5.3.

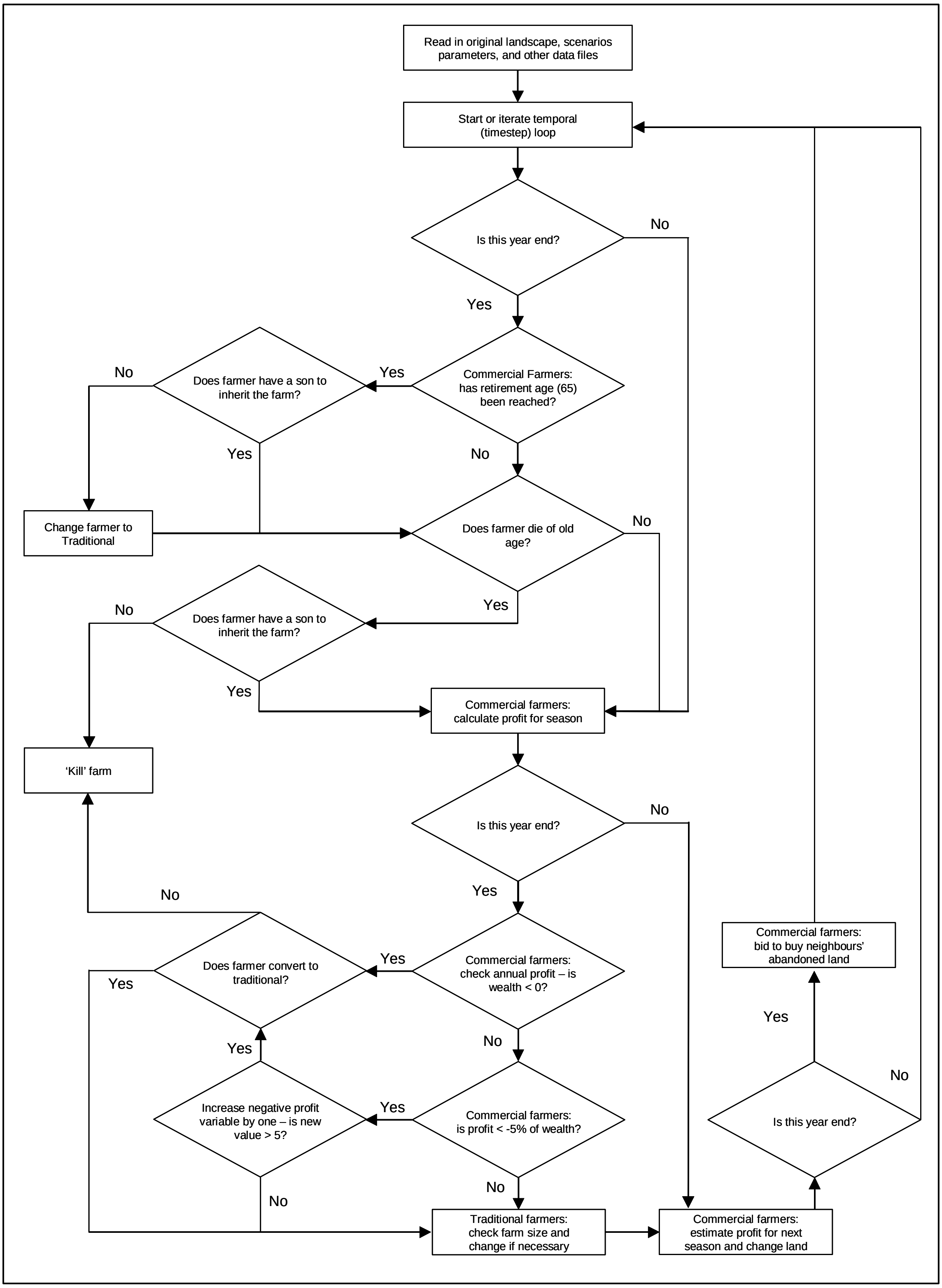

Figure 5.3 Procedure of the agent-based model of land-use/cover change. During each timestep seasonal functions are executed; annual functions are executed every fourth timestep (i.e. four seasons are simulated per year). Commercial agents demand more functions, resulting in increased computational demands as their number increases.

5.4.2 Rationale for Farmer Types

Criticisms of using the *Homo economicus* assumption when modelling human decision making have already been discussed above (section 3.5) and it became clear during interviews with local farmers and farming officials that the use of a single model of agent behaviour would not be sufficient to represent all farmers in the landscape adequately. There is a clear distinction between commercial farms that operate in a rationally economic, profit-maximising manner, and those that operate on a part-time basis or merely to maintain traditional agricultural practices and landscape aesthetics. One local vintner in SPA 56 suggested:

> *"Whoever has a vineyard nowadays is like a gardener... they like to keep it, even if they lose money. They maintain vineyards because they have done it all their life and they like it, even having to pay for it. If owners were looking for profitability there would be not a singe vineyard... People here grow wine because of a matter of feeling, love for the land..."*

Thus, for many land owners in the landscape their farm is a hobby kept for traditional cultural reasons with very little regard for its financial rewards. The economically rational agent is not appropriate for these 'traditional' farmers. Furthermore, there are many farms across the study area that are run to provide *supplementary* household income. For example, in one area (Santa Maria) one farmer stated that of the 80 livestock farmers in the area only 20 (one quarter) made their living solely from farming with the rest taking their primary income from alternate sources (light industry or building services nearby) but still keeping some land and livestock active. The farmer termed this supplementary activity 'extensive farming':

> *"Extensive [pastoral] farming is less time demanding... once a day, 1 or 2 hours can be enough... I see in the future extensive farming will be the most important though - I don't know if it is more profitable or not - but because it needs less time."*

In San Martin del Valdeglesias, a farming official described a similar situation:

> *"In general, [pastoral] farming is part-time, with only a few exceptions. The main problem is that it is not economically profitable - it is difficult to sell the product."*

And again in Villa del Prado, a spokesperson for a Farming Association:

> *"[Arable] agriculture here is part time... there are a few farmers that live on horticulture, and have vineyards as an extra."*

Finally, in an area traditionally known for its vineyards:

> *"Part-time workers? Yes, most of them... there are no full-time farmers. Here there are only retired people and their children, who work somewhere else, and help their retired parents with the labouring. There is no way to live on wine production."*

This part-time nature must be acknowledged in the behaviour of agents of the model. In general, we might say that these traditional and part-time farmers are less concerned with the economic state of the market, and their activities will not be sensitive to changes in it. In contrast, those farmers for whom their farm is their sole source of income treat their land as a commercial enterprise:

> *"There are some young farmers, 5 or 6 of whom are less than 25 years old, that are making important investments. If someone wants to live from livestock farming, they need to have an entrepreneurial vision, a business mentality, like in any company."*

These farmers will adopt land-use and farming practices to maximise their income and behaviour very much in the *Homo economicus* model. Different behaviours (i.e. model rules) are therefore appropriate for each type of farmer; 'commercial' agent or 'traditional' agent (Table 5.1). Before the attributes and decision-making rules unique to each agent type are discussed in detail, attributes common to both are now described.

5.4.3 Agent Attributes

Each agent owns a farm, composed of a finite number of pixels which may be in any one of nine of the eleven total land-covers – agents cannot own water or urban land-

Table 5.1 Summary comparison of 'traditional' and 'commercial' agents' attributes. Attributes were derived after interviews with local actors within the study area.

Attribute	*Traditional Agent*	*Commercial Agent*
Commitment	'Part-time' or 'Hobby' farmer	'Full-time' businessman
Age	Often older	Retires at 65
Land Exchange	Will not exchange land	Will buy/sell land to achieve profit
Land-uses	Maintain land in 'traditional' uses	Whatever land-use maximises profit
Financial Attitude	No interested in profit-making	Aims to maximise profit

cover pixels. Agents have an explicit age (measured in years) which is increased at each year end. The probability that an agent dies in any given year is based upon published human 'life-tables' that specify the probability of mortality of an individual given their age and country of residence (HLTD 2002). Using the standard practice, a uniform random deviate in the interval [0, 1] is generated and if less than the probability specified for the agent's age, the agent is deemed to have died during this year. If an agent does die, there is a probability that he will have a son to inherit the farm and continue its maintenance – the calculation of this probability is dependent upon the type of agent and explained in more detail below.

5.4.3.1 Commercial Agents

If a commercial agent dies there is a son to inherit the farm when the following rule is true:

$$\text{IF } (\text{R}{\scriptsize\text{AND}}[0, 1] < (\text{propC} + \textit{personal\_choice})\ ) \qquad \text{S 5.1}$$

Thus, if a uniform random deviate in the interval [0,1] (RAND[0,1]) is less than the proportion of agents that are commercial (propC) plus a *personal_choice* parameter, the agent has a son to inherit the farm as a commercial agent. The probability of inheritance is based on the proportion of commercial agents in the neighbourhood not because this is the mechanism that dictates whether an agent has a son but because this is likely to be an important factor in determining whether a son *wants* to continue his father's business. The *personal_choice* value is added to the probability to ensure that when there are no commercial agents in the landscape, there is still a chance that a son will want to continue the business – this accounts simply for personal choice (of the son) and the father's individual influence over his son's attitudes (which are likely to be just as, if not more, important than the proportion of the local community that has the 'commercial worldview'). This value may be positive (the son is inclined to continue the business) or negative (the son is disinclined to continue the business). An agent's initial *personal_choice* parameter is set in one of two ways. First, a mean *personal_choice* value can be set at model initialisation, around which all agents' *personal_choice* variables will vary. Second, if a mean is not set at initialisation each agent's *personal_choice* value will be set randomly in the interval [-0.5, 0.5]. If a son inherits the farm, a new *personal_choice* value is set as ±10% of his father's. The son's age is randomly set to a value between 20 and 40, ensuring that the value is less than the dying agent's age minus 20 (this assumes that farmers do not have children before the age of 20). However, if the agent is younger than 40 it is assumed that either he does not have a son, or if he does that the son is not old enough to take over the running of the farm. If a son is not present the farm 'dies' – ownership of all pixels is released (i.e. enter an un-owned state) and the farm is assumed to be abandoned.

As well as passing their farm onto their sons at death, commercial agents may also retire. If the commercial agent has reached retirement age (65 years) a check is made to establish if the agent has a son in the same manner as above (S 5.1). If there is no son, the agent becomes a traditional agent. This transition is made because it is assumed that having farmed his land all his life, a farmer is unlikely to want simply to give up his land for nothing (a sentiment that interviews suggested to be strong). Commercial agents' land-use decisions are based on several factors related to profitability – market conditions, land fragmentation, transport costs, and land productivity. Crop and pasture yields are not represented explicitly in the model and thus the use of real-world data for market conditions (i.e. profits and costs of production) is not possible. Furthermore, economic market fluctuations (i.e. responses of prices to supplies and demand) are not modelled explicitly. Rather, hypothetical scenarios of crops and pasture 'values' and 'costs' (or production) are used to simulate landscapes situated in buoyant, depressed or other economic situations. Market values and costs, along with other parameters influencing agent behaviour, are tuned to represent 'business as usual' market conditions (Table 5.2).

As noted in section 5.3.4, the spatial biophysical heterogeneity and land-tenure history of SPA 56, has left farmland in a fragmented configuration. A farm in which land parcels are spatially contiguous with large land agglomeration will provide greater economies of scale than land owned by a agent that is composed of smaller, fragmented and spatial distributed parcels of land. Commercial agents in the model consider land fragmentation as they calculate their estimated and actual profit:

$$\text{Fragmentation Value} = 1 - (\text{prop\_farm} / \text{max\_dist}) \qquad \text{Eq. 5.1}$$

where prop_farm is the proportion of the total farm area composed by the pixel cluster (i.e. field) in which the pixel under consideration lies, and max_dist is the maximum distance between the pixel under consideration and another pixel owned (and in use) by the same agent. Thus, when prop_farm is large and max_dist is small, the fragmentation value of the pixel is low. This index penalises pixels in small clusters at greater distances from other pixels owned by the agent.

Table 5.2 Parameters for baseline model configuration. These parameter values specify the 'business as usual' scenario, and are explained in more detail below. Results from this parameter set are used as the standard by which to compare model tests and sensitivity analysis.

Parameter	*Value*
Agent Age	Random normal distribution ± 10 years about 50 year mean
Bad Years	5 years
Conversion Costs	Pine, 0.3; Transition Forest, 0.3; Pasture, 0.2; Deciduous, 0.3; Scrub, 0.1; Holm Oak, 0.3; HOP, 0.4; Crops, 0.1;
Land-cover	SPA 56 1999 Land-cover (8 covers, 109 patches)
Loss Resilience	-0.1
Land-tenure	SPA 56 2005 Land-tenure (519 agents, 1213 patches)
Market Values	ValueC = 5.0 ValueP = 2.5, costC = 1.0, and costP=1.0
Personal Choice	Normally distributed range about mean of 0.0
Perspective	Randomly assigned with equal probability

Distance to the nearest road or track is considered as an acknowledgement to incurred transport costs as specified by von Thünen's model of agricultural location theory. Direct distance to market is not considered as in the von Thünen model, as there are multiple market locations in this study area. The cost of distance to the nearest road for each pixel is normalised by the maximum distance possible across the whole study area (giving a range for this value of [1/max_dist] to 1). Pixel productivity is considered in the model by the land capability index used in the biophysical model as previously discussed above (section 2.5). Pixels with greater land-capability values are assumed to produce higher yields and are thus more profitable.

Considering these factors, commercial agents calculate profit, at each timestep and for each pixel, for the three potential land-uses as follows:

$$\text{Crops profit} = (\text{valueC} \times \text{lcap}) - (\text{frag\_value} \times 2 \times \text{costC}) - (\text{road\_dist} / \text{lsp\_max}) \qquad \text{Eq. 5.2}$$

$$\text{Pasture profit} = (\text{valueP} \times \text{lcap}) - \text{costP} - (\text{road\_dist} / \text{lsp\_max})) \qquad \text{Eq. 5.3}$$

$$\text{Abandoned profit} = -0.1 \qquad \text{Eq. 5.4}$$

In terms of crop-land, greatest profit is earned by pixels with a high land capability, low fragmentation value, and low distance to the nearest road when the value for crops is high and the cost of production is low. Pasture profit is calculated in a very similar manner to crops, the difference being that the fragmentation value of the pixel is not considered. The rationale behind this position is that land for grazing does not afford much advantage by being clustered in large patches. Small areas of land may be used for grazing just as easily as large. However, the distance to the nearest road or track is important, as this will facilitate movement of livestock between areas of pasture and to the market. There is no immediate value provided by owning land in an abandoned state. However, the costs of doing so are also minimal (all land is assumed to be owned rather than rented, as is largely the case in SPA 56). As the land may become profitable in the future, and long term planning or forecasting of the state of the market is not represented in the model, the cost per abandoned pixel per season is minimal compared to the costs of active agricultural land-uses (i.e. non-abandoned pixels).

Profit at each timestep is calculated for each individual pixel and the sum of these values is calculated for each commercial agent's entire farm. If the total farmed area of the farm exceeds a 'maximum single farmer area' (max_farm_size), for each pixel exceeding this area a further cost is subtracted from the total farm profit. This cost is designed to reflect the infrastructure and labour required to farm an area greater than that possible by a single farmer with no hired labour. This maximum area that can be maintained by a single farmer (and his family) not employing hired labour is set at 40,000 m^2 (0.04 km^2, 44 pixels) in accordance with values suggested by interviewees.

During each season commercial agents estimate their profit for the next season with the land they currently own. Each possible configuration of their land (i.e. with each pixel they own in each of the three possible land-use states) is checked and one change made accordingly to maximise profit for the next season. Commercial agents' estimates of the values and costs for crops and pasture pixels for that next season are based on the values and costs of the current and previous seasons and the accuracy of the agents' estimate in the previous season. For example, an agent estimates the value of crop-land for the next season by:

$$\mathit{Est\_ValueC} = \mathrm{valueC} + \mathit{actual\_value\_diffcC} + \mathrm{R{\scriptsize AND}}[0, 0.05] \times (\mathrm{valueC} - \mathit{prev\_est\_valueC}) \qquad \text{Eq. 5.5}$$

where *actual_value_diffcC* is the difference between the previous value of crops and the current value of crops, *prev_est_valueC* is the previous estimated value of crops, and RAND[0, 0.05] is a uniform random deviate in the interval [0, 0.05]. This method ensures agents can estimate future prices reasonably well when values and costs change slowly, but perform worse when changes are rapid.

If the land-use configuration of the land currently owned can be modified to improve profit, land-use conversions are made. A hierarchy of land-uses is considered to restrict some land-use conversions. In this hierarchy crop-land is above pasture, which in turn is above abandoned land. An unlimited number of pixel conversions down the hierarchy may be made in any one season. Only one conversion up the hierarchy may be made in any one season. Conversion up the hierarchy (e.g. from abandoned to crop-land) will require resources of both time and money and thus the rate at which these changes can be made is restricted in the model. Conversions down the hierarchy require considerably fewer resources and are simply achieved by reducing maintenance levels.

At each year's end, commercial agents assess the profitability of their farms. If annual profit is equal to, or less than, a specified proportion of their wealth (specified for all agents by the loss_resilience parameter), the 'negative profit' year counter is increased by one (this counter is re-set to zero if annual profit is not less than the specified proportion of wealth). If this counter becomes greater than a specified number of years (specified by the bad_years parameter), a check is made on the commercial agents' transition to a traditional agent. A transition occurs when the following rule is true:

$$\text{IF } (\text{R}{\small\text{AND}}[0, 1] < (\text{propT} + \textit{personal\_choice}) \text{ OR } \textit{age} > 50) \qquad \text{S 5.2}$$

If either statement in this block is true, the commercial agent becomes a traditional agent. The first half of the block (before 'OR') is similar to the check made when an agent dies or reaches retirement age. However, in this case the proportion of traditional agents in the local neighbourhood is considered as this will be a prime determinant on whether the commercial agent is susceptible to the 'traditional worldview' and wants to continue to farm, despite it not being his primary income. The second statement in the block checks the age of the agent. If the agent is 50 or older he automatically becomes a traditional agent. This switch in perspective is based on the assumption that a younger farmer will want to move onto another job elsewhere because they still have 'time on their side' to start a new career. If older than 50, it is assumed that the farmer will be less inclined (or skilled) to endeavour to find a new full-time career and will therefore maintain the farm as a supplementary income. However, if both statements are false, the farm is abandoned.

At each year's end commercial agents also bid to buy abandoned pixels that neighbour their own land. All neighbouring abandoned pixels are examined to assess if their ownership and conversion, to either crops or pasture, will increase the agent's profit in the next season. The neighbouring abandoned pixel that will increase profit by the largest margin is then bid for. A 'conversion cost' is factored into the cost of purchasing that is dependent upon the land-cover being converted and the duration the pixel has remained in either a non-agricultural land-use (for non-agricultural land-use) or in the current use (for agricultural land-uses). A conversion factor for each land-cover is multiplied by the time in the given state, to produce the conversion cost. These conversion factors are weighted to penalise land-covers that will require large amounts of biomass to be removed and will be costly to convert to agricultural land-use (e.g. the pasture conversion factor is greater than for crops as it will be more costly to go from pasture to crop-land than *vice-versa*). The duration a pixel has been in its current state is introduced as a factor as this is both an indicator of, again, biomass levels but also of how 'established' an area of land is from a historical human land-use perspective. The maximum bid an agent will offer is given by:

$$max\_bid = 4 \times pixel\_profit \times (65 - age) \qquad \text{Eq. 5.6}$$

where *pixel_profit* is the estimated increased profit it will afford (multiplied by four seasons to give profit for a year) and *age* is the age of agent. The constant is included to account for the age of the farmer, as this gives a rough guide to the number of years of profit the pixel (if bought) will provide to the farmer until retirement. If the bid being made is larger than the asking price of the current owner, ownership passes to the bidding agent and land-use changed to the most profitable state. The buying agent's wealth is decreased by the asking price (not the maximum bid), and the seller's increased commensurately. If two agents bid for the same pixel, the highest bid wins (assuming it is greater than the asking price) and the maximum bid is the value that changes hands (i.e. seller's wealth increases by the maximum bid value, and the buyer's decreases commensurately). The asking price of an agent is set as the current wealth of that agent divided by the total number of pixels owned by that agent. If the pixel is abandoned but un-owned the asking price is set to the 'current market price' for pixels. This price is set to:

$$\text{Current Market Price} = 40 \times \text{mean\_tot\_pixel\_profit} \qquad \text{Eq. 5.7}$$

where mean_tot_pixel_profit is the mean pixel profit for the season across all pixels owned by commercial agents in the landscape. The constant gives an estimate of potential profit to be made by that pixel (in the current market state) over the next decade (i.e. 40 seasons). If a bid is not as large as the asking price, ownership stays with the current owner and the pixel remains in the abandoned state. Whatever the result of a bid, once all agents' bids have been considered, the program continues to the next season.

5.4.3.2 Traditional Agents

Traditional agents follow similar rules regarding their succession following death, but are not assumed to retire, do not consider any profit-making activities, and do not seek to buy land from neighbours. If a dying traditional agent is older than 40, checks are made to establish if there is a son to inherit the farm (as either type of agent). The son inherits the farm as a commercial agent when the following check is true:

$$\text{IF (mean\_tot\_pixel\_profit + propC - (}\textit{age} \text{ / 100)) > 0} \qquad \text{S 5.3}$$

This statement assumes that the son will be willing to become a commercial farmer when the profit in the landscape is generally high, when there are other commercial farmers in the landscape (i.e. he sees that others are finding it possible to make a living from their land), and when his age is low (and therefore he is assumed to be more willing to take a risk and 'give it a go'). Age is scaled to the order of mean_tot_pixel_profit and propC (i.e. 1×10^0).

If this statement is false a check is made to establish if there is a son to inherit the farm as a traditional agent in a similar manner to that above (S 5.1). There is a son to inherit the farm when the following rule is true:

$$\text{IF (R}\textsc{and}\text{[0, 1] < (propT + }\textit{personal\_choice}\text{))} \qquad \text{S 5.4}$$

As above, the probability that a traditional agent has a son is based on the proportion of traditional agents in the neighbourhood not because this is the mechanism that dictates whether there is a son but because this is likely to be an important factor in determining whether a son *wants* to continue in the steps of his father and the *personal_choice* parameter reflects the son's personal attitude. Again, the son's age is randomly set to a value between 20 and 40, ensuring that the value is less than the dying agent's age minus 20. If both checks (S 5.3 and S 5.5) are false the farm is abandoned.

Just as commercial agents consider a 'maximum single farmer area' (*max_farm_size*) greater than which they must pay for extra maintenance costs (for hiring labour etc.), traditional agents consider a maximum farm size beyond which they cannot maintain land. If their total farm size is larger than this maximum size, the appropriate number of pixels (with the lowest land capability values of pixels at the edges of clusters in the farm) is abandoned. This maximum farm size decreases with age once the agent reaches retirement age (65 years), representing his decreasing ability to maintain land (despite help from relatives). This rate is given by the function:

$$mf\_size = max\_farm\_size * \exp((retirement\_age - age)/wt) \qquad \text{Eq. 5.8}$$

where *mf_size* is the maximum farm size of a the retired agent, *retirement_age* is the retirement age of agents in the landscape, *age* is the age of the agent in question, and *wt* is a shape parameter (default value = 8, chosen in rough accordance with interviewees' understanding). Thus, the area of land a farmer is able to maintain is assumed to decrease exponentially with age after retirement.

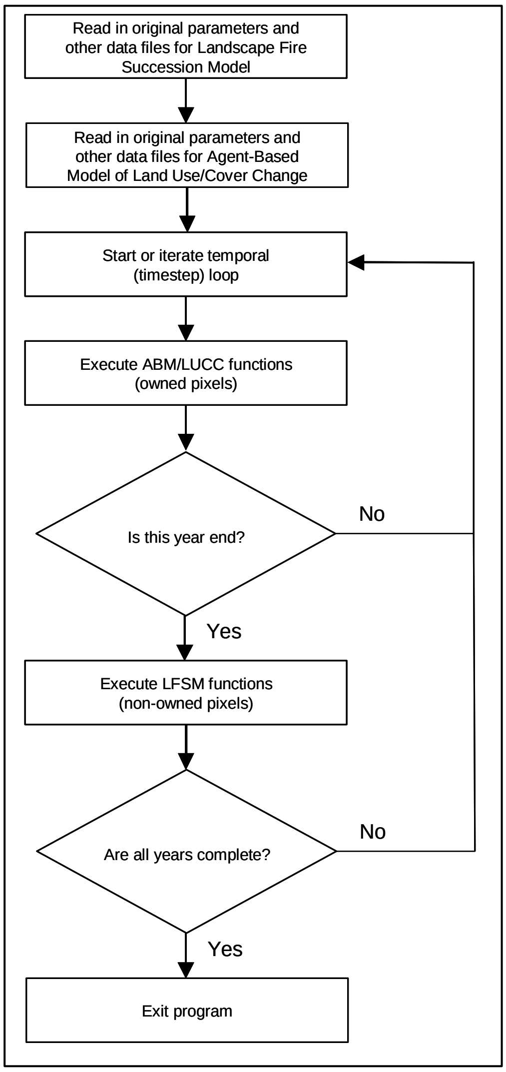

Figure 5.4 Procedure of the integrated socio-ecological simulation model. Agents' activities are simulated at each timestep - vegetation succession and wildfire at each year end. ABM functions are executed prior to LFSM.

5.4.4 Integration with Landscape Fire Succession Model

The agent-based and landscape fire succession (described in the chapter four) models are directly integrated (Figure 5.4). Agents' activities are simulated in each timestep – vegetation succession and wildfire are simulated at each year's end. ABM/LUCC functions are executed prior to LFSM functions in a given timestep, with vegetation succession only simulated for non-owned pixels. Thus, the model assumes that if an area of land is under human management, vegetation succession is not sufficiently active to cause a land-cover change. Figure 5.5 presents an example time series of land-use and land-cover proportions for baseline parameters (Table 5.2).

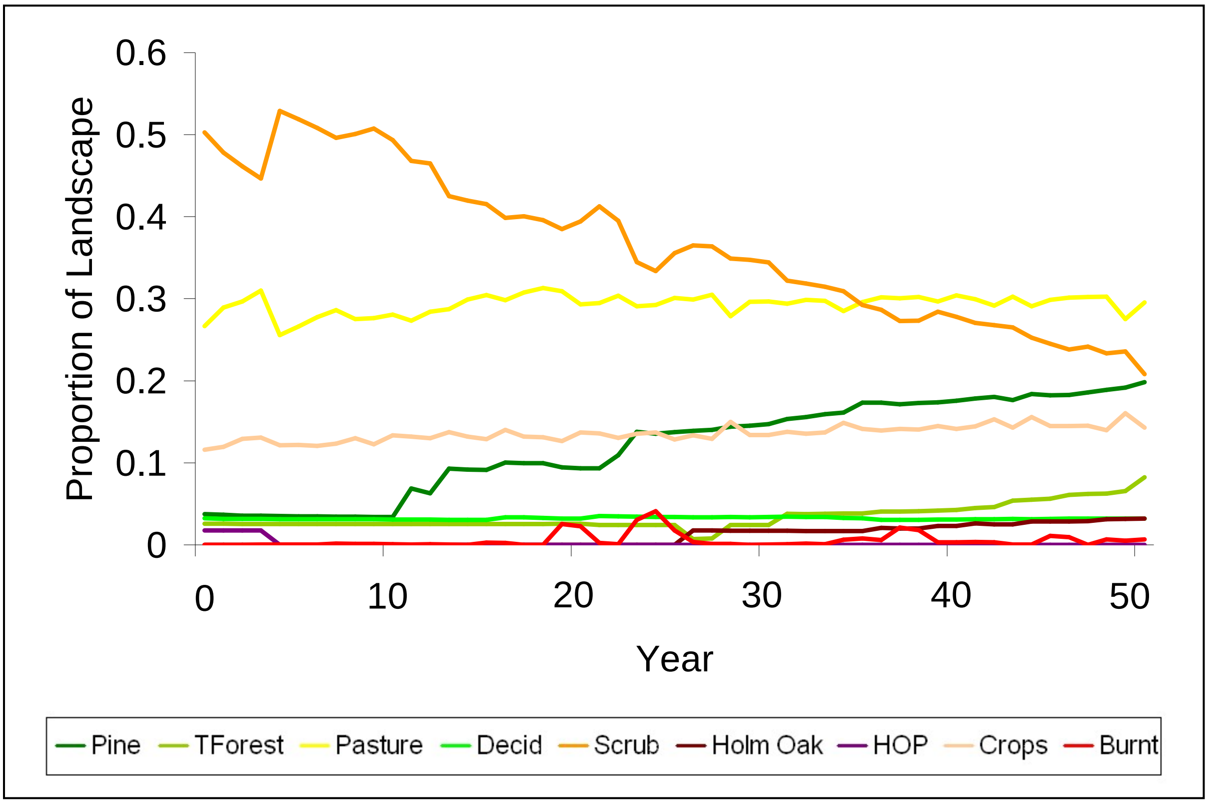

Figure 5.5 Land-cover and use time-series for the baseline 'business as usual' scenario. Pasture and crop abundances remain stable whilst biophysical processes result in shifts from scrub to pine and other forest land-covers.

5.4.5 Summary

This section has described the structure and rationale of the ABM/LUCC. Two agent types (perspectives) have been developed after interviews with local actors in the study area suggested that not all farmers are profit maximisers. Many local actors in the study area work only part-time on the land and do so largely for cultural rather than economic reasons. The model therefore considers 'commercial' agents that make decisions with the aim of maximising profit and 'traditional' agents that make decisions based upon traditional (existing) land-uses. Other factors considered in the model are the spatial locations of farmland, neighbours' influence on farmer perspective, and the attitudes of younger generations that will potentially inherit land. The integration of this model with the LFSM presented in the previous chapter was outlined and example output presented.

5.5 SENSITIVITY ANALYSIS AND TESTING

5.5.1 Introduction

The sensitivity analyses and testing undertaken here are performed with the same objectives as for the biophysical model in section 4.6 (i.e. to verify implemented model structure and to examine the relative importance of model parameters). The simple sensitivity analysis technique of varying one parameter whilst holding all others constant is only possible for parameters that are universal to all agents. These are loss_resilience, bad_years, and conversion_cost. Agent properties (*age*, *perspective*, *personal_choice*) and land-tenure vary between individual agents and combine to form the context in which agents' decisions are made. Varying individual agents' properties individually is both infeasible computationally and unlikely to make any discernible influence on system-wide properties given the large number of agents (i.e. 6,328). Market conditions (i.e. valueC, valueP, costC, and costP) are universally set and examined, and initial land-cover configuration is also tested. Test results are compared with parameters and market conditions set to represent baseline conditions relative to the model structure (termed 'baseline model configuration', Table 5.2).

The relative importance of agent properties is tested using model replicates (runs) tailored to maximise, minimise or cover the range of possible situations they provide (see Table 5.3). Testing is done on a subset of the original study area data, considering 519 agents on a grid measuring 101 pixels square. Tests for land-cover and land-tenure are made using random maps of the same dimensions. Using a data subset of this size ensures system dynamics are adequately represented whilst reducing computational demands and time. Two state variables are considered for the sensitivity analysis and testing of the ABM/LUCC – proportion of the landscape in crop land-use and proportion of the landscape in pasture land-use (Table 5.4). These are the two main state variables that agents make decisions about. Other variables and measures of system-state that are examined are number of parcels (land-tenure), number of patches (land-cover) and mean pixel profit.

Table 5.3. Parameter values for ABM/LUCC testing. Parameter values were chosen to span the parameter space of the model to examine the range of possible model states and investigate important parameters and input data.

Scenario	*Agent Property*	*Distribution of Values*
A1	Age	Random normal distribution ± 10 years about 35 year mean
A2	Age	Random normal distribution ± 10 years about 50 year mean
A3	Age	Random normal distribution ± 10 years about 65 year mean
A4	Age	Random normal distribution ± 20 years about 65 year mean
BY1	Bad Years	1 year (fixed value)
BY2	Bad years	10 years (fixed value)
CC1	Conversion Costs	-50% on baseline
CC2	Conversion Costs	-10% on baseline
CC3	Conversion Costs	+10% on baseline
CC4	Conversion Costs	+50% on baseline
LC1	Land-cover	Percolation probability p = 0.2 (SPA land-tenure, 509 LC patches)
LC2	Land-cover	Percolation probability p = 0.4 (SPA land-tenure, 291 LC patches)
LC3	Land-cover	Percolation probability p = 0.5 (SPA land-tenure, 198 LC patches)
LC4	Land-cover	Percolation probability p = 0.6 (SPA land-tenure, 94 LC patches)
LC5	Land-cover	Percolation probability p = 0.6b (SPA land-tenure, 83 LC patches)
LC6	Land-cover	Percolation probability p = 0.8 (SPA land-tenure, 1 LC patch)
LC7	Land-cover	Percolation probability p = 0.8b (SPA land-tenure, 1 LC patch)
LR1	Loss Resilience	-0.11(fixed value)
LR2	Loss Resilience	-0.25 (fixed value)
LR3	Loss Resilience	-0.50 (fixed value)
LT1	Land-tenure	Percolation probability p = 0.20 (511 Agents, 2653 LT patches)
LT2	Land-tenure	Percolation probability p = 0.40 (478 Agents, 1297 LT patches)
LT3	Land-tenure	Percolation probability p = 0.45 (442 Agents, 1005 LT patches)
LT4	Land-tenure	Percolation probability p = 0.50 (404 Agents, 791 LT patches)
LT5	Land-tenure	Percolation probability p = 0.55 (313 Agents, 480 LT patches)
LT6	Land-tenure	Percolation probability p = 0.60 (224 Agents, 296 LT patches)
LT7	Land-tenure	Percolation probability p = 0.80 (19 Agents, 19 LT patches)
M1	Markets	ValueC = 2.50 ValueP = 1.25, costC = 1.0, and costP=1.0
M2	Markets	ValueC = 2.50 ValueP = 1.25, costC = 0.5, and costP=0.5
M3	Markets	ValueC = 10.00 ValueP = 5.00, costC = 2.0, and costP=2.0
M4	Markets	ValueC = 10.00 ValueP = 5.00, costC = 1.0, and costP=1.0
PC1	Personal Choice	Normally distributed range about -1.0
PC2	Personal Choice	Normally distributed range about -0.5
PC3	Personal Choice	Normally distributed range about 0.5
PC4	Personal Choice	Normally distributed range about 1.0
PP1	Perspective	100% traditional
PP2	Perspective	25% commercial, 75% traditional
PP3	Perspective	25% traditional, 75% commercial
PP4	Perspective	100% commercial

5.5.2 Universal Parameters

Of the parameters governing system-wide properties for all agents, loss_resilience and bad_years were not found to have any discernible influence on land-cover proportions (i.e. less than ±15% change, Table 5.4). However, increased conversion costs were found to result in significant differences in agricultural land-covers. For doubled baseline scenario conversions costs, pasture land-cover decreased while crop-land increased. This difference in behaviour between pasture and crops highlights that increased conversion costs demand that increased profit afforded by conversion is greater. Thus, agents are more likely to convert to crops than pasture when conversions costs are high.

Table 5.4 ABM/LUCC sensitivity analysis and testing results for land-cover composition. Resulting land-cover proportions for Crops and Pasture are presented with percentage change due to each treatment. Treatments causing a change of greater than ±15% in the state variable are shown in bold. PD is difference between the base and specified replicate (rep) as a proportion of base. Scenarios are specified in Table 5.3.

	Crops			*Pasture*		
Scenario	*Mean (Base)*	*Mean (Rep.)*	*PD*	*Mean (Base)*	*Mean (Rep.)*	*PD*
A1	0.14	0.15	0.08	0.30	0.29	-0.03
A2	0.14	0.15	0.07	0.30	0.29	-0.01
A3	0.14	0.15	0.07	0.30	0.27	-0.10
A4	0.14	0.15	0.07	0.30	0.28	-0.05
BY1	0.14	0.15	0.04	0.30	0.29	-0.04
BY2	0.14	0.15	0.07	0.30	0.28	-0.06
CC1	0.14	0.14	-0.02	0.30	0.31	0.06
CC2	0.14	0.15	0.02	0.30	0.28	-0.06
CC3	0.14	0.14	-0.04	0.30	0.29	-0.01
CC4	0.14	0.16	0.13	0.30	0.25	**-0.16**
LR1	0.14	0.14	-0.03	0.30	0.29	-0.02
LR2	0.14	0.15	0.07	0.30	0.29	-0.01
LR3	0.14	0.13	-0.09	0.30	0.29	-0.01
LT1	0.14	0.08	**-0.43**	0.30	0.26	**0.26**
LT2	0.14	0.13	-0.12	0.30	0.19	**0.19**
LT3	0.14	0.14	-0.03	0.30	0.10	0.10
LT4	0.14	0.17	**0.17**	0.30	0.06	0.06
LT5	0.14	0.14	-0.02	0.30	-0.12	-0.12
LT6	0.14	0.07	**-0.51**	0.30	-0.85	**-0.85**
LT7	0.14	0.01	**-0.94**	0.30	-0.99	**-0.99**
M1	0.14	0.12	**-0.19**	0.30	0.07	**-0.77**
M2	0.14	0.13	-0.08	0.30	0.20	**-0.31**
M3	0.14	0.18	**0.23**	0.30	0.30	0.02
M4	0.14	0.17	**0.22**	0.30	0.56	**0.88**
PC1	0.14	0.08	**-0.42**	0.30	0.19	**-0.36**
PC2	0.14	0.11	**-0.25**	0.30	0.24	**-0.18**
PC3	0.14	0.15	0.04	0.30	0.28	-0.04
PC4	0.14	0.15	0.04	0.30	0.29	0.03
PP1	0.14	0.06	**-0.56**	0.30	0.17	**-0.42**
PP2	0.14	0.16	0.10	0.30	0.28	-0.06
PP3	0.14	0.15	0.05	0.30	0.29	-0.03
PP4	0.14	0.16	0.12	0.30	0.28	-0.04

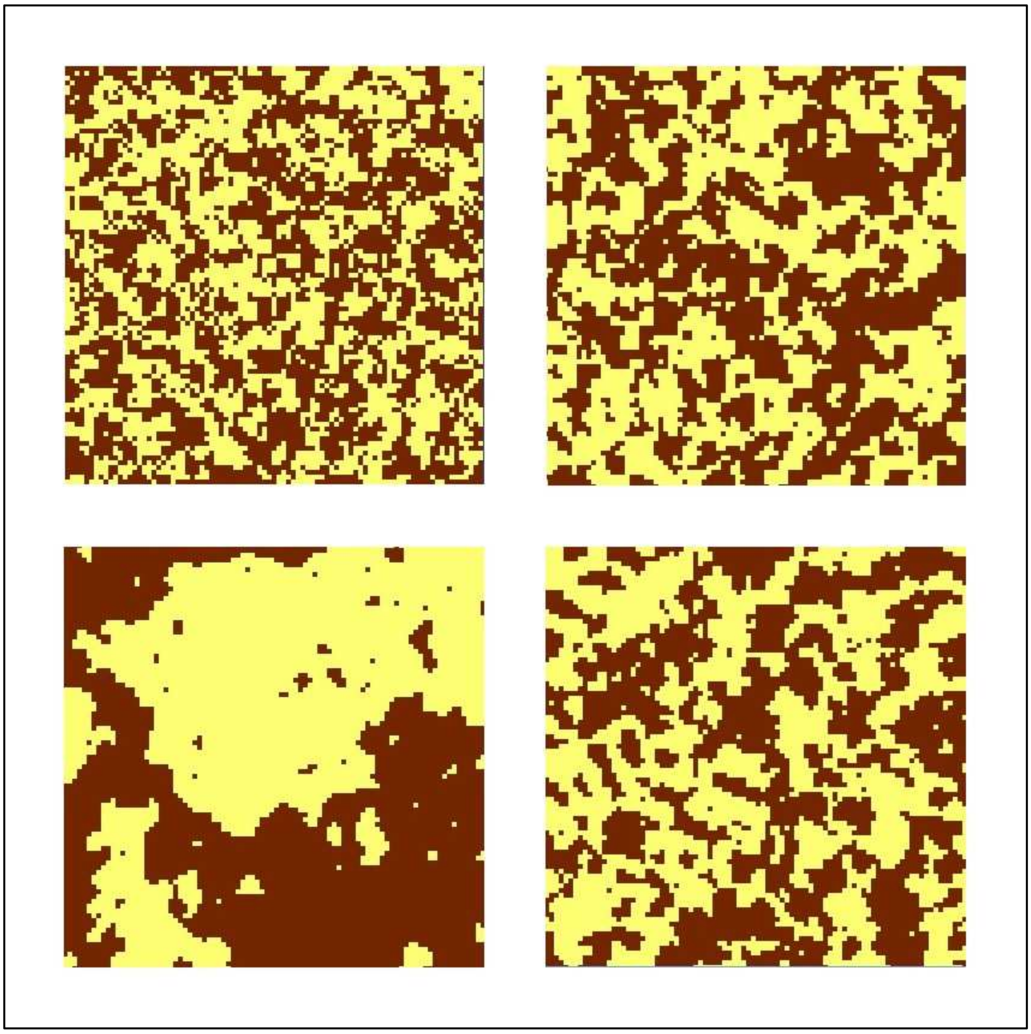

Figure 5.6 Examples of random maps generated with varying percolation probability parameter *p*. Clockwise from top left, maps are generated with $p = 0.2$, $p = 0.4$, $p = 0.6$ and $p = 0.8$. Colours designate two random land covers for the illustration of patch sizes only.

5.5.3 Land Tenure and Land-cover

Land-tenure configuration is an important determinant of land-use and land-use decision-making in the model, producing significant differences in land-use abundance across the range of possible land-tenure structures (Table 5.4). Maps with random land-tenure configuration were generated for testing using the modified random clusters method (Saura and Martinez-Millan 2000). This method specifies a percolation probability parameter p to generate landscape maps with clusters of pixels of varying (randomly distributed) size. With increasing p, number of patches in the landscape decreases and mean and maximum patch size increase. At the percolation threshold $p_c \approx 0.5928$ a cluster spanning the entire landscape is generated (Saura and Martinez-Millan 2000). Random land-tenure maps were initially generated for $p = 0.20$, $p = 0.40$, $p = 0.60$, and $p = 0.80$ (e.g. Figure 5.6), resulting in landscapes with numbers of agents and land-tenure parcels as specified in Table 5.3.

An inverse relationship between p and pasture abundance is observed (Figure 5.7). That is, as initial mean land-tenure parcel size increases, the simulated proportion of land devoted to pasture decreases. Relative to mean base values, pasture land-use is increased for $p \leq 0.4$ and decreased for $p \geq 0.6$. With a significant difference observed between these values for p, further model replicates for $p = 0.45$, $p = 0.50$ and $p = 0.55$ were examined. The dramatic decrease in pasture abundance observed between random initial land-tenure maps for $p = 0.55$ and $p = 0.60$ commensurate with p becoming greater than p_c, for reasons related to the maximum farm size rule. As the rules specified by the model favour crops on larger land-tenure parcels, the observed decrease in pasture abundance might be expected to be due to the replacement of pasture by crops. However, the proportion of the landscape used for crops peaks for an initial random land-tenure landscape with $p = 0.50$, with lesser abundance for smaller and greater p values (Figure 5.6). For $p \leq 0.55$ a dominance of pasture indicates this land-use to be the most profitable for farmers. For $p \geq 0.60$, land-tenure parcels are so large that the maximum farm size rule becomes an influence on land-use decision-making. Thus, fewer farmers only actively farming a subset of larger farms, results in decreased land-use (both pasture and cropland-uses – Table 5.5).

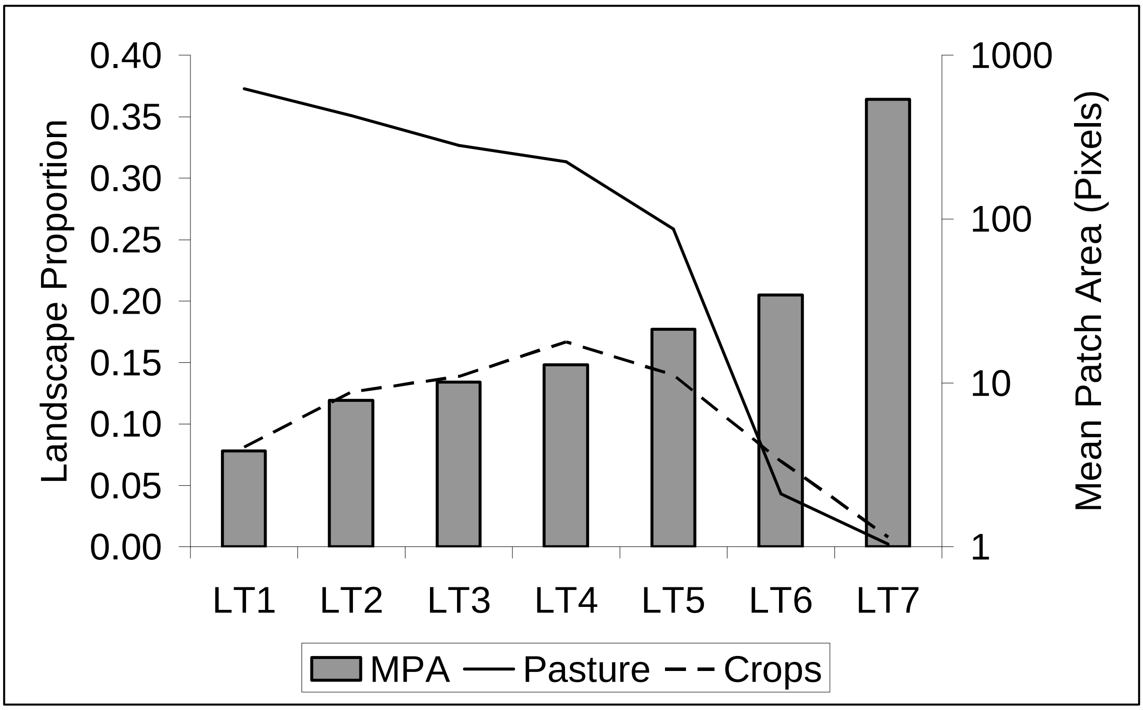

Figure 5.7. Landscape land-cover proportions and mean land-cover patch area for random land-tenure maps. An inverse relationship between abundance of pasture (solid line) and mean patch area (bars) is evident – abundance of crops (dashed line) peaks for median mean patch area. Scenarios are specified in Table 5.3.

Table 5.5. Agricultural landscape structure characteristics for land-tenure maps. Also presented are final agricultural land-use proportions for corresponding model test replicates. Land-tenure maps with $p \geq 0.60$ result in landscape with very lo agricultural land-use, due to large size of patches and farms.

	Initial			*Final*
Scenario	*Agent Parcels*[§]	*MPA*[‡]	*Small Farms*[†]	*Agricultural Land**
Baseline	2.34	8.41	1.00	0.44
LT1	5.18	3.85	1.00	0.45
LT2	2.78	7.87	0.96	0.48
LT3	2.27	10.15	0.87	0.47
LT4	1.96	12.90	0.76	0.48
LT5	1.53	21.25	0.43	0.40
LT6	1.32	34.46	0.21	0.11
LT7	1.00	536.89	0.01	0.01

[§]Mean number of parcels per agent

[‡]Mean landscape parcel area (pixels)

[†]Farms with size < *max_farm_size* (as proportion of landscape)

*Proportion of landscape in crops or pasture state

Comparison of these results with the observed land-tenure configuration of SPA 56 in 2005 indicates that the actual land-tenure configuration is near that for which most land can feasibly be used (given the model assumptions – i.e. traditional family farms, little mechanisation etc.). The current land-tenure configuration of SPA 56 is very similar to the initial land-tenure configuration of the model scenario resulting in the greatest area of agricultural land-use (LT4). From model results the maximum area in agricultural land-use after a 50-year model replicate is 48% of the landscape (scenario LT4), whilst agricultural land-use accounted for 41% of SPA 56 in 1999 (and 44% after a 50-year model replicate using original SPA 56 land-tenure). Initial mean number of patches per agent and mean land-tenure parcel area are comparable between actual SPA land-tenure and scenario LT4 (Table 5.5), as are spatial metrics of land-cover structure (Table 5.7). The similarity of these land-tenure configuration characteristics suggests that over time the land-tenure configuration produced in SPA 56, via division and agglomeration of land parcels due to inheritance and exchange, is near an optimal state for the type of agriculture present in these landscapes. This statement is made with the assumption that the processes of agricultural production and change are accurately represented by the simulation model, and remembering the initial land-tenure maps used in model testing were randomly generated. Maps generated by the random cluster method do represent the spatial complexity of SPA 56 land-tenure reasonably well, however (Table 5.6). Land-tenure becomes increasingly fragmented (increasing number of parcels) between initial and final conditions (Table 5.6) with corresponding increases in land-cover fragmentation (in conjunction with ecological processes of succession and invasion as discussed below).

Table 5.6 Spatial metrics of land-cover pattern for land-tenure scenarios. Differences in spatial metrics (NP, number of patches; FRACD, fractal dimension; PARA, perimeter-area ratio) between initial landscape land-cover pattern and final simulated LUCC show increasing spatial complexity through (model) time. PARA decreases for LT6 and LT7 as this metric decreases for larger patch sizes. Scenarios are specified in Table 5.3.

	Initial			*Final*		
Scenario	*NP*	*FRACD*	*PARA*	*NP*	*FRACD*	*PARA*
Baseline	1213	1.04	756	1138	1.06	877
LT1	2653	1.02	1018	2062	1.04	1034
LT2	1297	1.03	788	1169	1.06	924
LT3	1005	1.03	786	967	1.06	912
LT4	791	1.03	787	791	1.07	901
LT5	480	1.03	826	594	1.07	908
LT6	296	1.02	883	270	1.05	934
LT7	19	1.01	1151	16	1.06	925

Random land-cover maps were generated using the random clusters method to examine the influence of original land-cover configuration (Table 5.3). Maps LC1, LC2 and LC3 (with p = 0.2, p = 0.4 and p = 0.5 respectively) were generated with similar land-cover proportions to SPA 56 land-cover in 1999 (allowing results to be compared with baseline results). Whilst similar in land-cover composition, these random initial maps are spatially dissimilar to observed SPA 56 land-cover (Table 5.7). For maps generated with clusters spanning the landscape (i.e. $p > p_c$), land-cover proportions comparable with original SPA 56 land-cover are not possible. Maps LC4 and LC5 were generated with total landscape proportions of 0.75 for pasture and crops respectively (with p = 0.6) and maps LC6 and LC7 were uniformly entirely composed of pasture and crops respectively (landscape proportion = 1.0).

Each of LC1, LC2 and LC3 tend towards a similar end state, characterised by slightly increased crop land-cover proportions compared with the baseline model configuration.

Results for LC4 – LC7 show markedly different resulting landscape land-cover compositions from baseline results (as would expected given the difference in initial land-cover composition) with initial dominant land-covers remaining sustained at high levels. Such behaviour is linked to landscape profitability; with so much agricultural land at model initialisation, profitability is high resulting in a majority of commercial farmers. As profit maximisers in a stable economy, the predominance of commercial farmers prevents agricultural land-use from dropping to baseline levels. Resulting land-cover maps are more spatially fragmented than their initial land-cover configuration. This is a result of both land-use change and subsequent ecological invasion and succession processes – land abandonment and conversion occurring at different times across the landscape, allied with variation in ecological succession rates and directions due to spatial heterogeneity of environmental resources, produces a patchier land-cover mosaic. These more fragmented land-cover mosaics are characterised by more, smaller patches and smaller fractal dimensions (Table 5.7). Potential impacts of increasing spatial fragmentation on wildfire regimes are examined in the next chapter.

Table 5.7 Spatial metrics of land-cover pattern for land-cover scenarios. Spatial metrics (abbreviations as for Table 5.6) indicate land-cover to be more spatially fragmented for simulated LUCC than initial land-cover configuration. Scenarios are specified in Table 5.3.

	Initial			*Final*		
Scenario	*NP*	*FRACD*	*PARA*	*NP*	*FRACD*	*PARA*
Baseline	109	1.12	549	1339	1.04	1159
LC1	509	1.04	927	1513	1.04	1160
LC2	291	1.04	793	1386	1.04	1162
LC3	198	1.04	854	1262	1.04	1171
LC4	94	1.03	939	400	1.04	1080
LC5	83	1.03	796	412	1.04	1094
LC6	1	1.00	13	380	1.04	1105
LC7	1	1.00	13	368	1.04	1085

5.5.4 Agent Properties

Variation in mean and range of ages was not found to influence agricultural land-use abundance in any marked manner (Table 5.4). However, the *personal_choice* parameter (representing the propensity of a son to inherit his father's farm) is found to influence land-use directly. For mean *personal_choice* < 0.0 agents are less inclined to use or retain their land and abandonment increases. This occurs as farms are less likely to be inherited, causing reduced numbers of agents in total and increasing ratios of traditional to commercial agents (Table 5.8). For mean *personal_choice* > 0.0 a subtle shift (relative to base results) toward crop use away from pasture use is observed.

Table 5.8 Agent types for *personal_choice* and agent perspective scenarios. For mean *personal_choice* < 0.0 (PC1 and PC2) the ratio of traditional to commercial agents increases dramatically. Values are numbers of agents in the landscape. Scenarios are specified in Table 5.3.

Scenario	*Total*[§]	*Traditional*[§]	*Commercial*[§]	*Ratio (T:C)*
PC1	301	62	239	0.26
PC2	409	64	345	0.19
PC3	451	3	448	0.01
PC4	485	7	478	0.01
PP1	459	459	0	-
PP2	453	29	424	0.07
PP3	480	34	446	0.08
PP4	475	18	457	0.04

Examination of the influence of agent perspective reveals that only landscapes for which the original composition is 100% traditional result in a markedly different land-cover state than the baseline model configuration (lower pasture and crop abundance, Table 5.4). In this scenario, commercial farmers do not occur in the landscape as a major influence on agents' switch from a traditional perspective to a commercial perspective is the profitability of the landscape. This profitability is calculated from profit made by commercial agents. As commercial agents do not exist in the landscape, profitability equals zero and agents are not inclined to change perspective. For all other parameter sets, at some point during the simulation a large shift in agents' perspective from traditional to commercial is observed (e.g. Figure 5.8). On closer examination this shift coincides with the point in time at which some threshold value of mean pixel profit is reached. The threshold value is dependent predominantly on the ratio of traditional to commercial agents in the landscape, indicating that perspective of neighbouring agents influences economic decision-making. A positive relationship is found between the mean pixel profit threshold value and ratio of traditional to commercial agents (Figure 5.9). That is, as the mean number of neighbouring agents with a traditional perspective increases, the economic incentive to shift to a commercial perspective is driven up. The large shifts from traditional to commercial perspective causes a brief decrease in mean

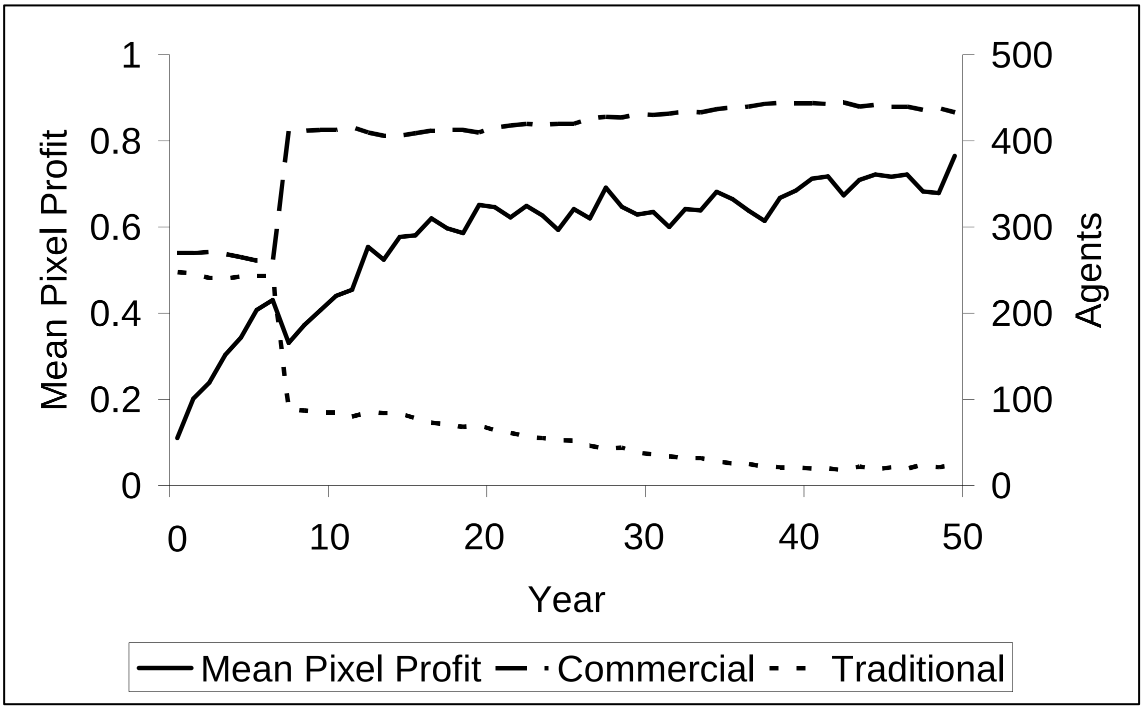

Figure 5.8 Time-series of agent perspective and mean pixel profit. All model scenarios (except PP1) exhibit a dramatic shift toward commercial agents at some point during model replicates. In the example shown (baseline) it occurs at year seven. All model scenarios including commercial farmers also exhibit increasing mean pixel profit through time as the landscape organises itself to maximise profit.

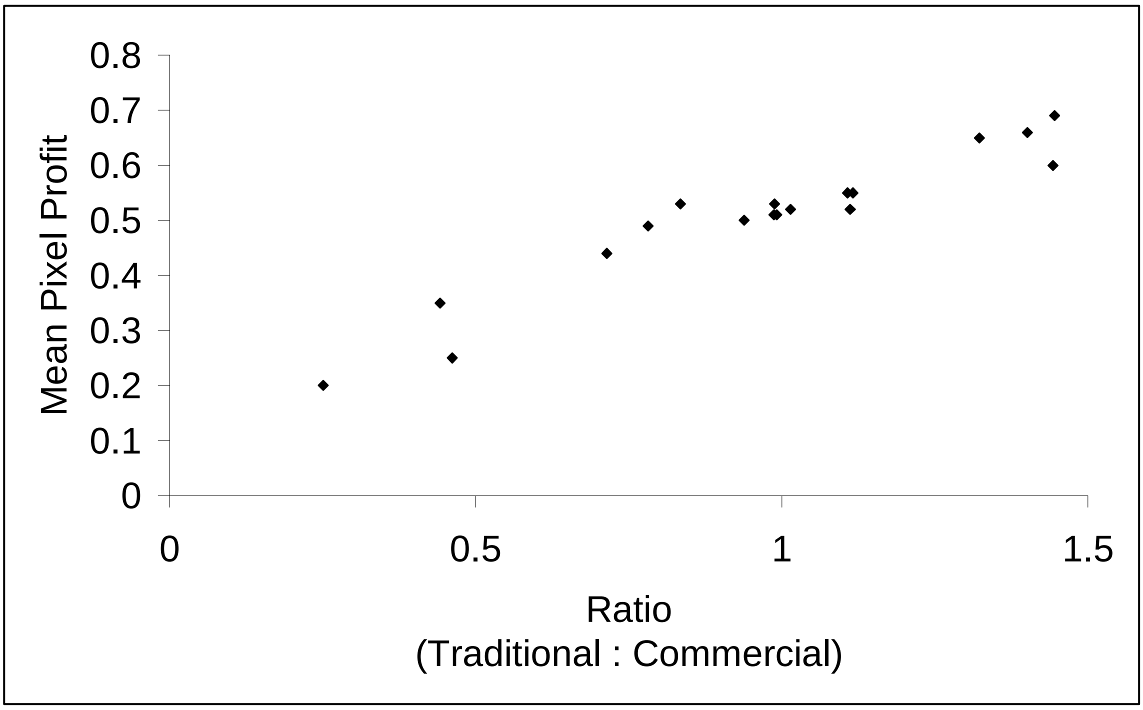

Figure 5.9 Relationship between threshold mean pixel profit values and ratio of farmer types. A positive linear relationship is evident. As the ratio of traditional to commercial farmers in the landscape increases, the mean pixel profit value at which the landscape shifts to a predominantly commercial perspective is driven up.

pixel profit, as traditional agents' land will be in use but it is very unlikely to be in the optimal profit-making configuration. Large shifts in perspective do not produce equivalently large shifts in land-cover change, again because traditional agents' land is already being maintained in an agricultural land-use.

5.5.5 Markets and Landscape Profitability

As expected simulated land-cover proportions are sensitive to market conditions (Table 5.4), with changes relative to market conditions observed as expected with respect to model structure. For example, depressed markets (e.g. M1) exhibit decreases in agricultural land-uses (particularly pasture) whilst buoyant markets (e.g. M4) show increased agricultural land-uses. Differences between 'depressed' (i.e. M2 and M2) and 'buoyant' (i.e. M3 and M4) scenarios due to variation in costs of production are also as expected with respect to model structure.

Through time mean pixel profit is observed to increase rapidly at first, then slowing possibly toward an asymptote (the value of which varies according to other model parameters – e.g. Figure 5.8). Such behaviour is evidence to suggest that as commercial agents convert their land between land-uses, and negotiate new ownership, total profitability increases. Thus, the landscape might be interpreted as 'self-organising' itself to the land-tenure and cover state that ensures optimal global profit. That is, individual agents' decisions on land-use and land exchange result in a landscape structure that maximises total system profit without any system-level organisational rules. Such self-organising behaviour has been found in other land-use decision-making models (e.g. Sasaki and Box 2003).

5.5.6 Summary

The parameters and variables composing each agent's context are closely intertwined in the agent-based model presented. The testing and sensitivity analysis presented in this section aimed to verify the intended functioning of the model and to establish variables with an important influence on land-use at the system level. Varying the individual attributes of over 500 agents, and the parameters that compose their individual decision-making context, is computationally infeasible and would result in an enormous volume of data for interpretation. Thus, the testing here examined parameter values across their possible range and compared resulting state variable values (agricultural land-cover and tenure composition and configuration) with a baseline scenario. Results indicate that market conditions are the predominant factor driving decisions and LUCC in general. However, land-tenure patterns were also found to be important determinants of land-use decision-making. Finally, the importance of interactions between parameters in agent decision-making was in evidence – examination of the causes of large shifts in agent perspective highlighted the interaction between pixel profit and neighbouring agents' perspectives. From this testing, the author suggests that the verification of the model's intended representation of land-use decision-making in SPA 56 has been achieved. Whether this representation is accurate will be addressed later in the thesis (chapter eight).

5.6 SUMMARY

This chapter has presented the structure of the agent-based model of land-use decision-making and the testing of that model once integrated with the biophysical model presented in chapter four. The development of the agent-based model drew on knowledge from several sources including previously established formal agricultural location theory, previous agent-based modelling projects, and the knowledge of local actors within the study area. The ABM itself takes an inherently 'bottom-up' representation of the study area, representing individual agents and their decisions made as a consequence of the context (spatial, economic, social) in which they are situated. However, several of the rules developed to represent this individual behaviour have their roots in von Thünen's formal agricultural theory (section 5.2), which assumes homogeneity of decision-making agents and environmental conditions. The roots of this theory and recent advances in simulating LUCC using agent-based approaches (e.g. Deadman *et al.* 2004, An *et al.* 2005, Manson 2005) were discussed to set the background rationale of much of the model structure (section 5.3). This literature review was supplemented by interviews with local actors, which highlighted that much of the decision-making made in the study area was not of an explicitly economic nature, leading to the establishment of two distinct agent types with differing behaviour.

Using this model structure revealed the importance of land tenure for the resulting agricultural land-cover compositions (section 5.5.3). Market conditions remained the predominant factor influencing land-use and land-cover compositions however. Other testing highlighted the importance of the interaction of economic and non-economic factors on the adoption of different agent perspectives. Model testing procedures and analysis have provided confidence that the model has been coded using the C++ programming language such that the modeller's intended representation of actors' behaviour in the study area has been manifested appropriately. That is, the model has been 'verified'.

In the next chapter the integration of this ABM/LUCC with the landscape fire succession model (presented in chapter four) is used to examine the potential impacts of agricultural LUCC on wildfire regimes for different scenarios of change for the study area. As such, development of the model ceases at this point and the model is used in the state in which it has been described. However, the iterative nature of the simulation modelling process means that there is always room for modification and improvement. Chapter eight presents the results of an attempt to assess the appropriateness of the model structure presented here (i.e. assess its validity or credibility), and to establish future improvements to the model, by engaging with local stakeholders in the study area.

CHAPTER SIX

SOCIO-ECOLOGICAL SIMULATION MODEL: INTERACTION BETWEEN LUCC AND WILDFIRE

6.1 INTRODUCTION

Human activity in the Mediterranean Basin is the primary cause of wildfire. Studies of fires in Spain have shown humans to be the cause of over 95% of all events (e.g. Moreno *et al.* 1998), in contrast to the conterminous United States where humans are responsible for approximately half of all wildfire ignitions (Brown *et al.* 2002). However, very few Landscape Fire Succession Models (LFSMs, see Keane *et al.* 2004 and chapter four) have attempted to *explicitly* represent the effect of contemporary human activity on wildfire regimes. The emphasis of previous LFSMs has been to gain a better understanding about vegetation dynamics under different disturbance regimes (e.g. Franklin *et al.* 2001), to understand important variables influencing fire frequency and behaviour (e.g. Miller and Urban 1999), and to explore large-scale ecological change (e.g. Thonicke *et al.* 2001). Those that have considered human activity have done so by considering broad scenarios of human impact upon vegetation formations, imposed in a 'top down' fashion (e.g. Baker 1995, Perry and Enright 2002a). These models are able to evaluate the effects of human activity on disturbance regimes across multiple spatio-temporal scales, but do not explicitly consider the impact(s) of human activity on wildfire ignition frequency and location or subsequent spread. Models are being developed to examine wildfire at the wildland – urban interface, however these models focus more on wildfire risk and less on modification of the wildfire regime itself (e.g. Platt 2006). In a novel approach the LFSM developed in this thesis (chapter four) is integrated with an Agent-Based Model of Land-Use/Cover Change (ABM/LUCC – chapter five) to form an integrated Socio-Ecological Simulation Model (SESM). This SESM is applied in this chapter in an effort to improve understanding of the impacts of human activity on the spatial-temporal relationships between vegetation and wildfire by considering frequency and location of human-caused fires explicitly. For the remainder of this thesis, this SESM will be termed SPASIMv1 (SPA SIMulator Version 1).

Five scenarios of economic and social change are used here (Table 6.1). These scenarios are run for 28 model years to the year 2026, using initial empirical data for 1999. This 28 year time interval was used because of issues regarding representation and the later stakeholder model evaluation exercise (chapter eight). These issues are discussed further below (section 6.6). Land-use maps and agent data were produced as output for each year-end during each model run for these five scenarios. For model replicates considering human activity, these data specify locations of agricultural land-use and human population demographic change at each year. Thus, five scenarios of land-use and human-population change are produced in a 'bottom-up' fashion, the results of which could then be used to simulate multiple realisations of wildfire regimes under these conditions. This approach was taken to reduce the extreme computational demands required to simulate 6,328 spatially-explicit land-use decision-making agents across 900,453 pixels. Other wildfire-related parameters with which to examine these scenarios include the number of years after fire which humans (mainly tourists) will not visit an area (see Eq. 4.6, section 4.5.2.1), the maximum burn size due to fire-fighting efforts (see section 4.5.3), and proportions and numbers of human- versus lightning-

Table 6.1 Model scenarios used to examine the influence of human activity on land-cover. These scenarios are based on those used in section 5.5 and reflect either changing markets or human-population demographics within the study area.

Model Scenarios
Market Scenarios
These scenarios examine how the state of the agricultural market influences LUCC alone. Three scenarios are considered;
M1 – Declining agricultural profitability (i.e. scenario M1 from section 5.5)
M2 – Maintenance of the *status quo* (i.e. baseline scenario from section 5.5)
M3 – Increasing agricultural profitability (i.e. scenario M4 from section 5.5)
Demographic Scenarios
These scenarios assume processes of free-market suburbanisation and increasing access to the study area from the nearby city of Madrid;
D1 – Declining agricultural markets (i.e. scenario M1 from section 5.5) plus increasing non-agricultural population
D2 – Maintenance of market *status quo* (i.e. baseline markets scenario from section 5.5) with decreasing propensity of sons to inherit farms from fathers (*personal choice* normally distributed range about -1.0; see section 5.4.3) resulting in ageing and declining agricultural population)

caused fires (see Eq. 4.6, section 4.5.2.1). These parameters allow the investigation of differences between characteristics of fires caused by humans and those with 'natural' causes, and an examination of the questions driving the first aim of the thesis (chapter one).

6.2 CHARACTERISING WILDFIRE REGIMES

SPASIMv1 is used here to examine the effects of human activity on the frequency and magnitude components of the wildfire regime. To establish what effects are present, appropriate measures to characterise wildfire regimes are required. From a wide number of possible heavy tailed frequency-area distributions (e.g. Schoenberg *et al.* 2003), the power-law distribution is the most parsimonious model (Millington *et al.* 2006) and has been suggested as an accurate descriptor of wildfire regimes for events over a large range of orders of magnitude and across many ecosystems (Malamud *et al.* 1998, Ricotta *et al.* 1999, Ricotta *et al.* 2001, Song *et al.* 2001). The power-law distribution is given as:

$$f(A) \sim A^{-\beta} \qquad \text{Eq. 6.1}$$

The power-law distribution produces a straight line in logarithmic space (e.g. Figure 6.1). The distribution, and the use of the exponent β, has been suggested as an efficient and effective measure for comparing wildfire regimes if frequency densities normalised by the temporal and spatial extents of the data set are used (Malamud *et al.* 2005, Millington *et al.* 2006). The exponent β is a measure of the ratio of small to medium to large fires; larger β values indicate 'large' fires are rarer relative to smaller fires, and *vice versa*. Millington *et al.* (2006) found that the range of β values for all empirical wildfire regime studies to that date using the power-law distribution fell in the range 1.1 – 2.2. Malamud *et al.* (2005) found β values for fires on United States Forestry Service (USFS) land across the conterminous USA fell between 1.30 – 1.81 during the interval 1970 – 2000. Values of β for Bailey's (1995) Mediterranean and Mediterranean Mountains ecoregions were 1.30 and 1.46 respectively. Ricotta *et al.* (2001) suggest fires for regions of Spain across the interval 1974 – 1999 fell in the range 1.1 – 1.5. When run for 500 years with an annual mean of five fires, model data show the vegetation flammability values presented in Table 4.5 give β values in the range 1.37 – 1.40. These values fit squarely within the range of empirical values found by (Malamud *et al.* 2005) and (Ricotta *et al.* 2001).

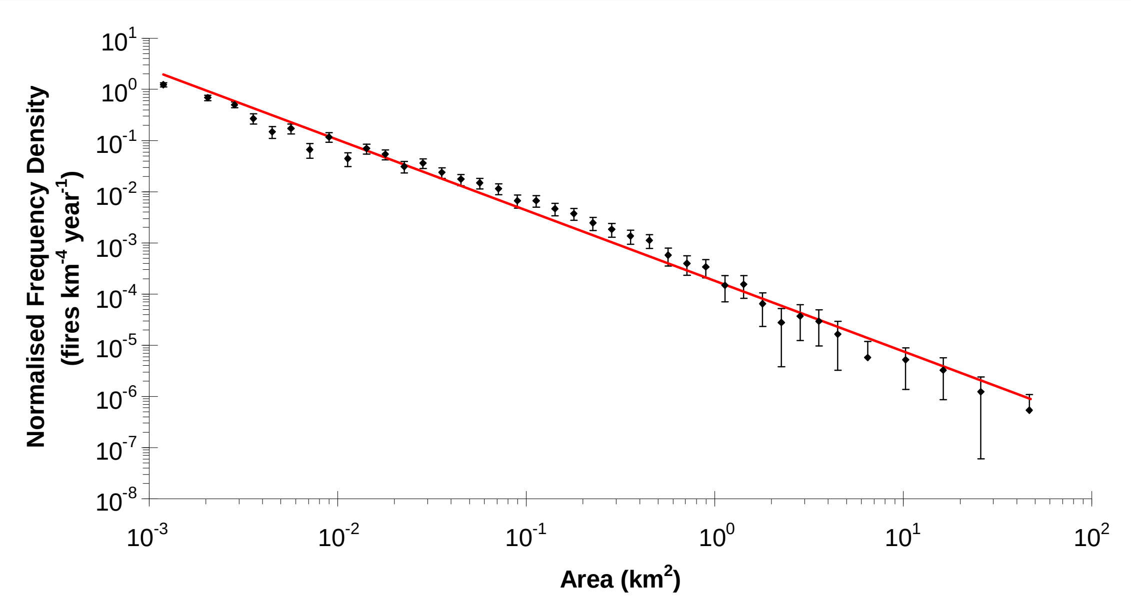

Figure 6.1 Example LFSM wildfire frequency-area distribution. The distribution exhibits power-law behaviour as demonstrated by the straight line relationship in logarithmic space between burnt area and frequency of occurrence (normalised by temporal and spatial extent of the data set). Error bars are ±1 SD.

However, questions have been raised as to what the power-law frequency-area relationship indicates and how it might be used to improve understanding about wildfire regimes (Millington *et al.* 2006). By comparing values of β for wildfire regimes with differing intensities and types of driving forces (e.g. vegetation, climate, human activity) Malamud *et al.* (2005) suggested that the influences of these driving forces might be revealed. While Malamud *et al.* (2005) used this approach to examine data across the whole of the conterminous United States, comparing β values might also prove useful for landscape-scale models (i.e. 1×10^3 km^2 extent) such as that being examined here. Indeed, the original motivation for investigating the potential of the power-law relationship to characterise empirical wildfire regime frequency-area statistics arose from studies of the forest fire cellular automata model (Malamud *et al.* 1998). The influences of various aspects of human activity on β are examined here. The influence of climate change on the wildfire regime will be explored using simple climate scenarios (for the LFSM alone). The sensitivity analysis performed on the LFSM in chapter four highlighted the importance of land-cover flammability on the behaviour of the model, as has been found for other cellular automata models of this type (e.g. Ratz 1995, McCarthy and Gill 1997, Perry and Enright 2002b). This chapter also examines the influence of land-cover flammability probabilities and spatial configurations on the wildfire regime independent of any human activity (i.e. using the LFSM alone). The interactions between wildfire and land-cover composition and configuration are also briefly examined. The discussion addresses the utility of the β value for characterising wildfire regimes at a regional level, and considers the success of the model at simulating the influence of human activity on the wildfire regime.

Alongside β, two other state-variables are used to characterise the simulated wildfire regimes. These state-variables are the largest burned area for a single event, and the total burned area for the duration of simulation replicates. Mean fire size is not considered as a state-variable as the power-law distribution does not have any defined moments (where the first moment is the 'average'). Land-cover composition is considered as the proportions of the total area for Forest, Agricultural and Scrub land-covers. Forest is composed of Pine, Transition, Deciduous, and Holm Oak forest land-covers. Agriculture is composed of Crops, Pasture and 'Holm Oak with Pasture' land-covers. Total number of land-cover patches, landscape fractal dimension and landscape contagion (as defined by McGarigal and Marks 1995) are used as measures of spatial landscape configuration. Unless otherwise specified all human activity scenarios were run with a maximum fire shell of 1000 (section 4.5.3), a human fires base count of five (*HF*, section 4.5.2.1) and recent burn value of five years (*RBY*, section 4.5.2.1). Unless specified, all model scenarios were replicated three times and the mean values, coefficient of determination (r^2, all values presented at 95% confidence interval) and/or standard error of measurement (SEM) of the state-variable reported. As noted above (section 4.6.1), sophisticated methods, such as Monte Carlo analysis (e.g. Xu *et al.* 2004) and latin hypercube sampling (e.g. Xu *et al.* 2005), have been found to be useful for analysis of spatial landscape models. However, these methods are not used here because probability density functions are not available for each parameter and because computational requirements limit the number of replications possible for each parameter set.

6.3 Human Activity, LUCC and Wildfire Regimes

6.3.1 Human-Activity Scenarios

Human-activity scenarios produced little variation in β values (relative to SEM – Table 6.2). Wildfire counts are also uninfluenced, with little variation around a mean of 153 fires for all scenarios. Mean largest fire and mean total burned areas do increase with decreasing agricultural activity, as would be expected. Mean total burned area is greatest for scenario M1 (agricultural market decline) and least for scenario M3 (agricultural market growth). When mean total burned area is plotted against the proportion of the landscape in agricultural land use, a linear inverse relationship is observed (Figure 6.2). This pattern is to be expected given that agricultural land-covers are least flammable.

Table 6.2 Wildfire summary statistics for scenarios of human activity. Relative to its SEM, β shows no significant difference between the scenarios. Mean largest fires and mean total burned areas are consistent with the levels of agricultural activity expected for each scenario. r^2 values are for power-law fits to wildfire burned area data. Error values are ± 1 SEM.

Scenario	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
M1	154	4.25 ± 0.67	14.40 ± 1.47	1.26 ± 0.13	0.97
M2	154	2.29 ± 0.76	12.57 ± 1.79	1.24 ± 0.10	0.98
M3	152	0.82 ± 0.12	7.33 ± 0.89	1.25 ± 0.12	0.98
D1	148	1.70 ± 0.26	8.63 ± 0.79	1.28 ± 0.09	0.99
D2	158	1.44 ± 0.28	9.45 ± 1.32	1.26 ± 0.12	0.98

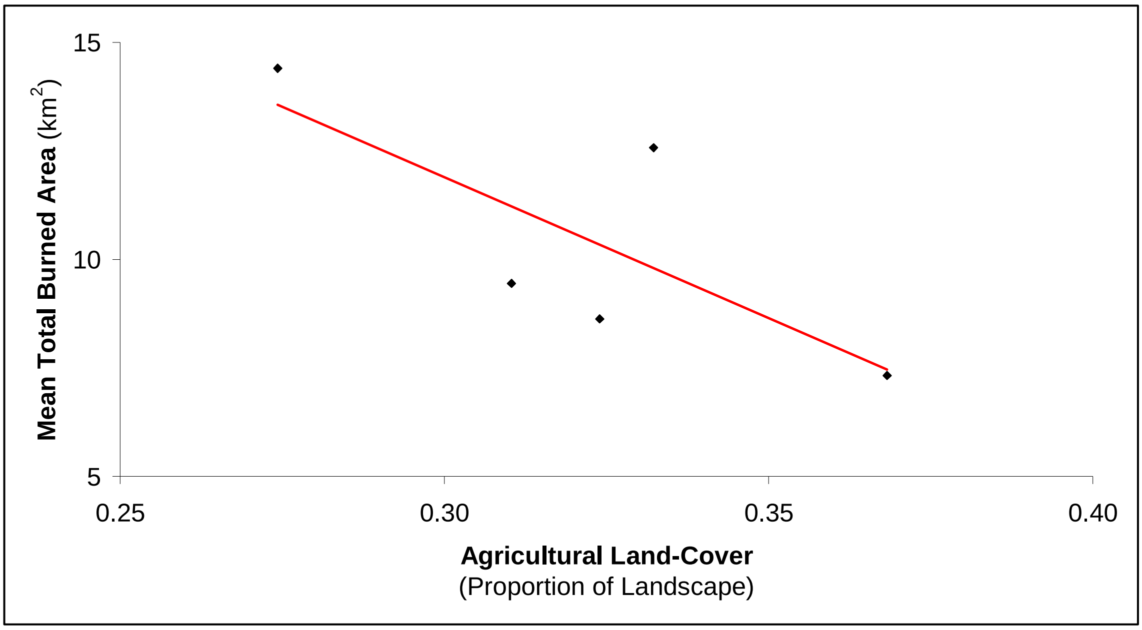

Figure 6.2 Relationship between mean total burned area and agricultural land-cover for scenarios of human activity. The linear regression line shows a negative correlation with $r^2 = 0.58$. Note the difference in scale between the axes – the slope of the regression line = -65.01. SEM of total area burned for each point ≤ 0.0001

Land-cover proportions for the year 2026 all suggest a lower abundance of Scrub than at present, which is due either to increased pasture land-cover or replacement by pine following agricultural abandonment. For scenarios M1, M2 and M3 (agricultural market decline, *status quo* and growth respectively – Table 6.1) inverse relationships are observed between Scrub and Agricultural land-cover abundance (Figure 6.3 and Figure 6.4). Land-cover composition varies more between scenarios (up to 10% of the landscape) than between individual scenario-replicates (up to 1% of the landscape) highlighting that human activity is a more important determinant of change than wildfire.

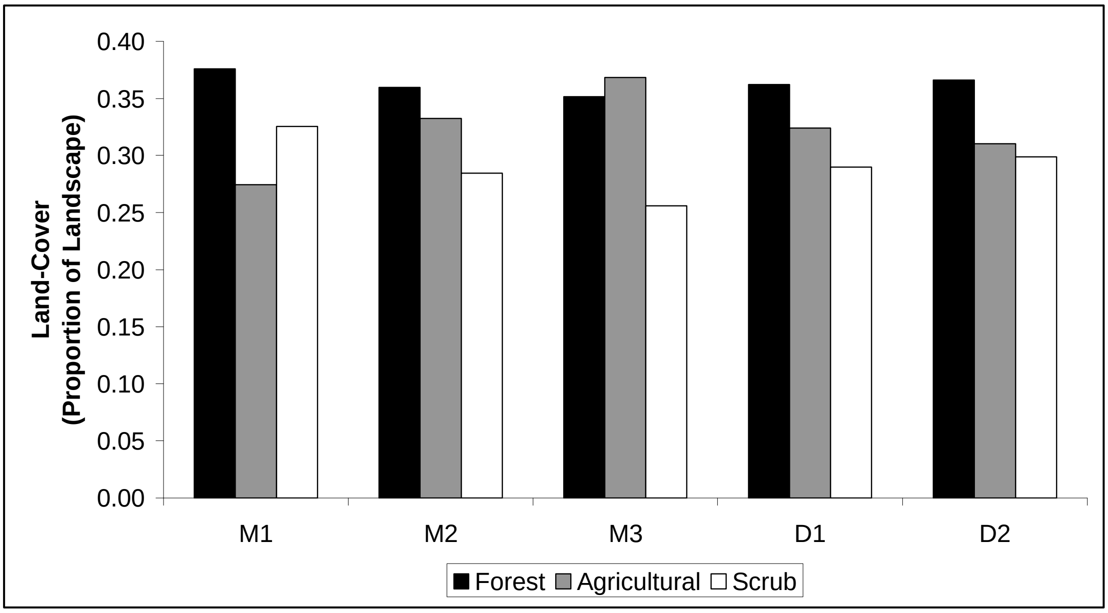

Figure 6.3 Mean land-cover landscape proportions for scenarios of human activity. For all scenarios Scrub is the land-cover occupying the greatest proportion of the landscape. Scrub proportions are inversely proportional to agricultural proportions. All values are for the final landscape state of the model scenario run (year 2026). SEM for all values < 0.002.

Landscape pattern metrics suggest that model repetitions for all scenarios result in a similar state (see Table 6.3). Visual inspection of resulting maps supports the values of these pattern metrics, highlighting very little structural (spatial land-cover configuration) variation between the scenarios (Figure 6.4). The variation in composition found above (Figure 6.3) is observable, for example replacement of scrub (light brown) by pasture (yellow) in scenario M3 compared with M1.

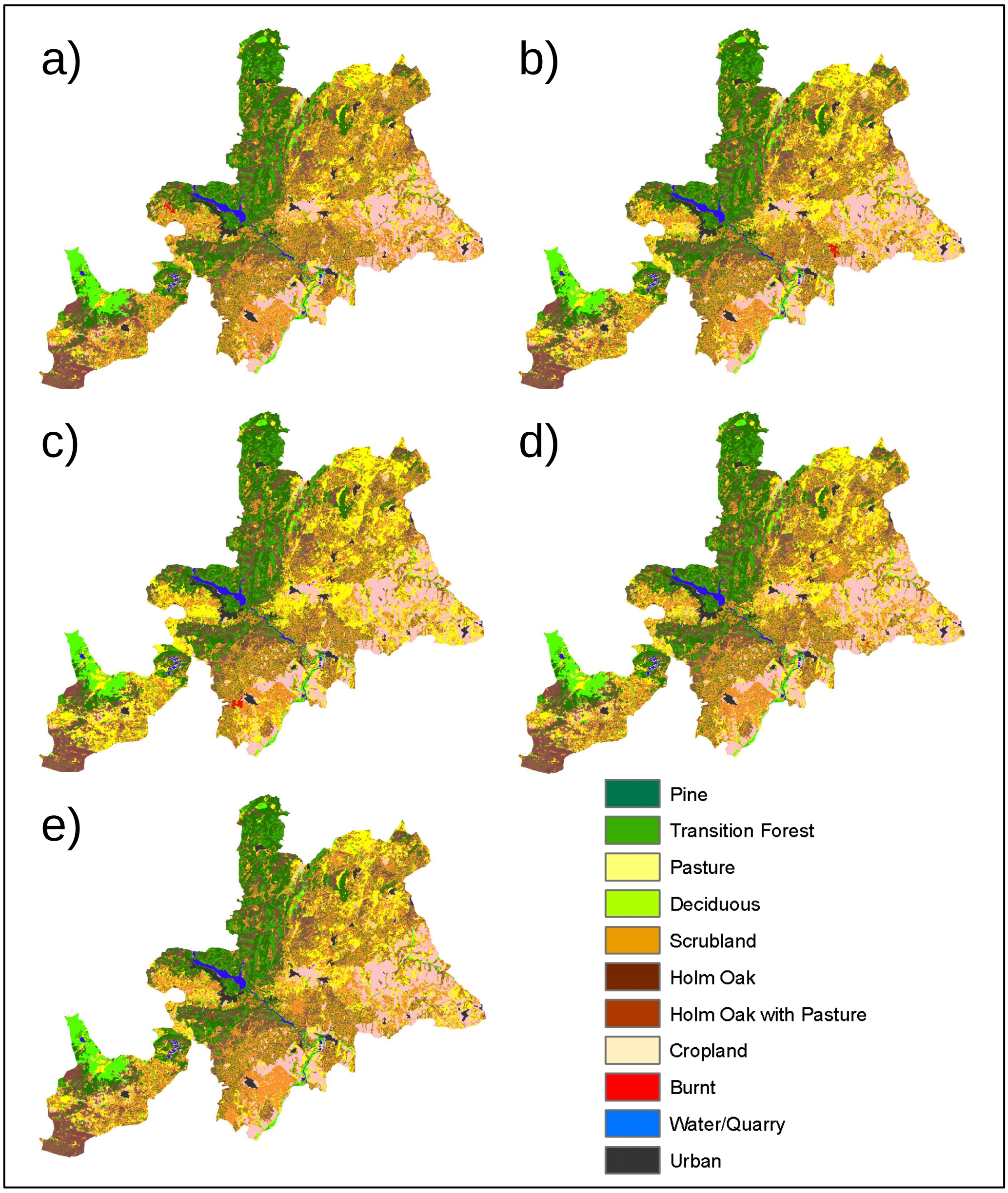

Figure 6.4 Maps of landscape land-cover in the year 2026 for human-activity scenarios. Scenarios are a) M1, b) M2, c) M3, d) D1 and e) D2. Visual inspection of the maps highlights very little structural (spatial land-cover configuration) variation between the scenarios, as suggested by the landscape pattern metrics (Table 6.3). Variation in composition is more obvious – for example replacement of scrub (light brown) by pasture (yellow) in scenario M3 compared with M1. Cropland composition and configuration vary little between the scenarios.

Alongside the examination of static end-states, the dynamic nature of the simulation model means time-series of measures of landscape state can be examined to provide a better understanding of processes of change. Time-series of landscape flammability, mean number of patches, mean fractal dimension and mean contagion are examined.

Table 6.3 Mean values of metrics of land-cover configuration for different scenarios of human activity. Results indicate all scenarios tend toward a similar landscape configuration. Error values for mean number of patches are ± 1 SEM (SEM for fractal dimension ≤ 0.002 and for contagion ≤ 0.2).

Scenario	Mean Number of Patches	Mean Fractal Dimension	Mean Contagion (%)
M1	65,164 ± 612	1.38	49
M2	72,411 ± 608	1.39	47
M3	71,295 ± 622	1.39	47
D1	72,440 ± 625	1.39	47
D2	72,710 ± 419	1.39	48

The time-series for mean total landscape flammability indicates that after a sharp rise (due to initial land abandonment as agents re-organise their land) flammability generally decreases through time (with the exception of scenario M1 – Figure 6.5). This pattern is due to replacement of Scrub with Forest and Agricultural land-covers through time, due to vegetation succession and agricultural decision-making (respectively). However, note that flammability for all scenarios varies across only ~ 2% of the entire range possible (i.e. 0 – 0.24).

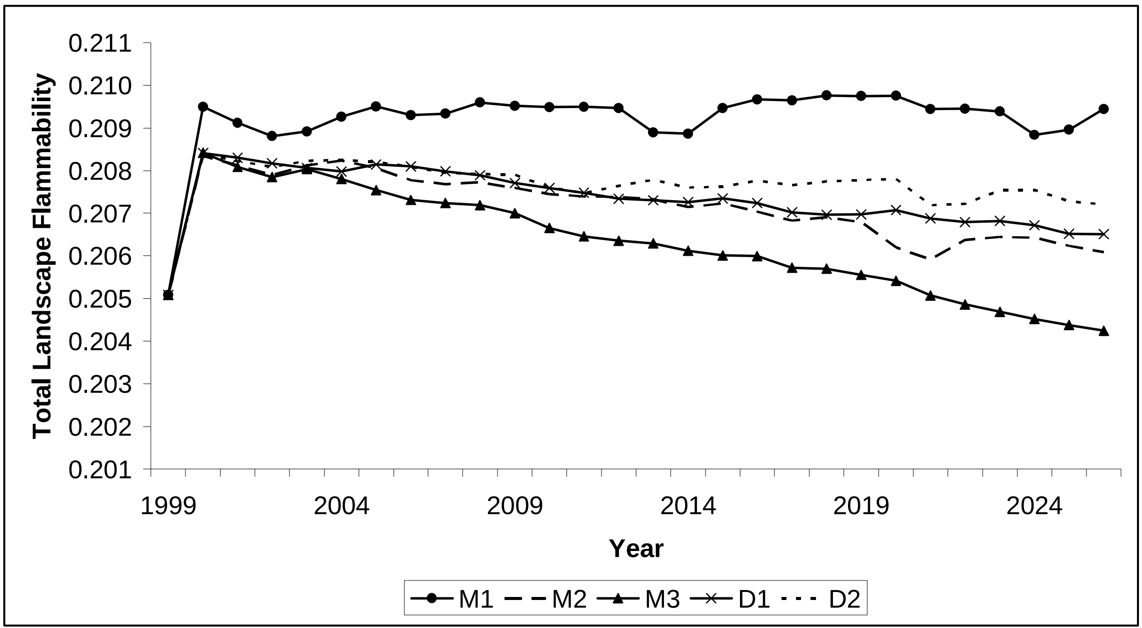

Figure 6.5 Time-series of mean total landscape flammability for human-activity scenarios. After an initial rise, values decrease steadily except for scenario M1. However, note the very small range of the y-axis – flammability probabilities range across only ~ 2% of the entire possible range (i.e. 0 – 0.24).

The mean number of patches rises sharply initially (due to initial land abandonment as agents re-organise their land) but in contrast to flammability continues to rise steadily through time (Figure 6.6). A closer examination of which land-covers are causing this rise indicates that Forest (predominantly early-succession Pine) is the main cause of the increasing number of patches, with a smaller contribution of scrub later in the time-series (e.g. Figure 6.7). This pattern is indicative of the spatially heterogeneous nature of vegetation succession-type processes occurring in the landscape as forest increases in abundance. Spatial variability in environmental conditions results in differential rates of establishment of forest vegetation across the landscape, producing patchy forest development (as observed across central and southern areas of the study area, Figure 6.4).

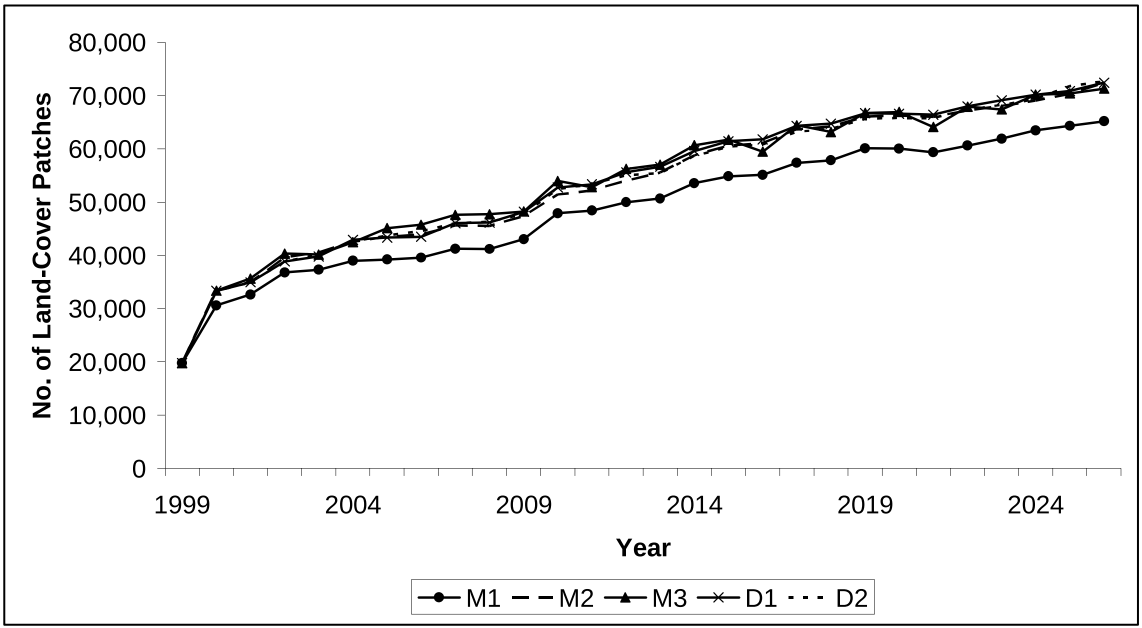

Figure 6.6 Time-series of mean number of patches for human-activity scenarios. Values increase steadily through time for all scenarios highlighting an increasingly 'patchy' landscape. This increase is due to increasing numbers of patches in Forest land-covers (e.g. see Figure 6.7) as vegetation succession-type processes occur.

Fractal Dimension varies little through time relative to the possible range of values (i.e. 1.0 – 2.0, Figure 6.8). When all land-cover patches in a landscape exhibit a similar degree of spatial irregularity, regardless of the number of patches or proportions of land-cover, fractal dimension has been found to be an insensitive measure of landscape pattern (Hargis *et al.* 1997). As such, the results here indicate that land-cover change

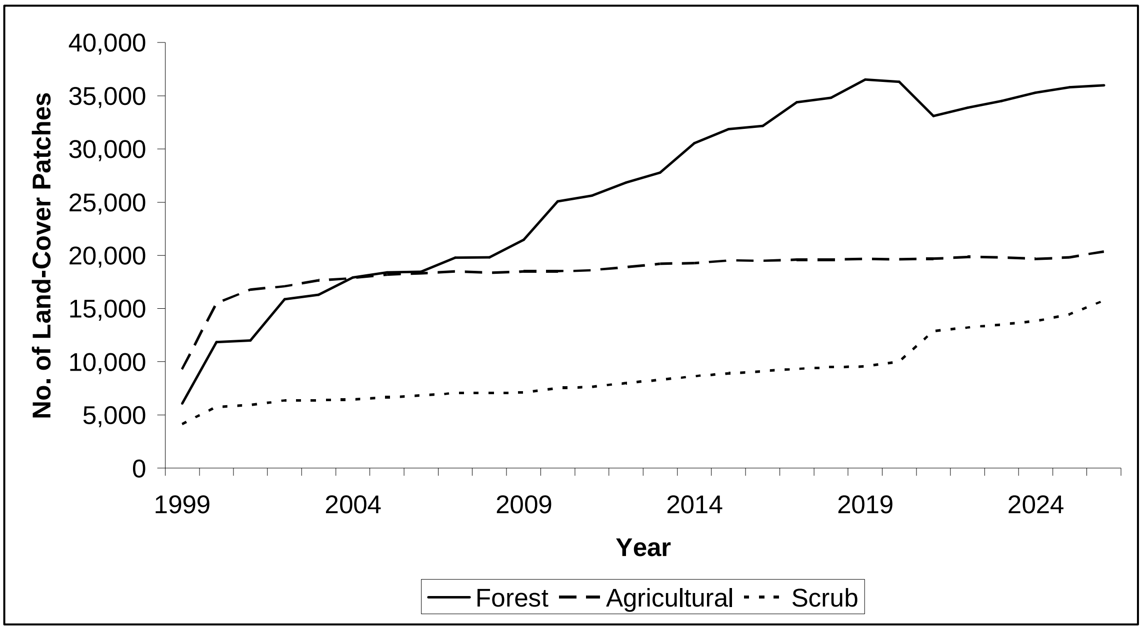

Figure 6.7 Time-series of mean number of patches by land-cover for scenario M2. Forest land-covers contribute most to total number of patches in the landscape (Figure 6.6). The number of Scrub land-cover patches begins to increase more rapidly late in the time-series, possibly linked to increases in contagion (see Figure 6.9).

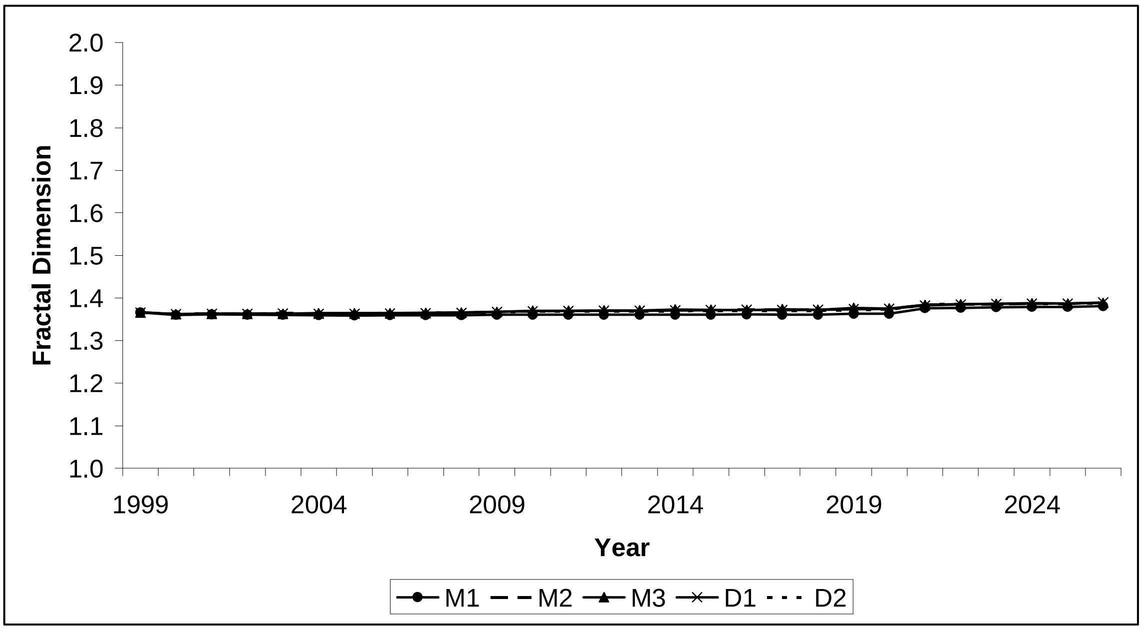

Figure 6.8 Time-series of landscape fractal dimension for human-activity scenarios. Relative to the possible range of values (1.0 – 2.0) fractal dimension changes little through time. This insensitivity and the nature of the landscape pattern metrics (see text) suggest land-cover change is proceeding in a manner spatially consistent with observed landscape structure.

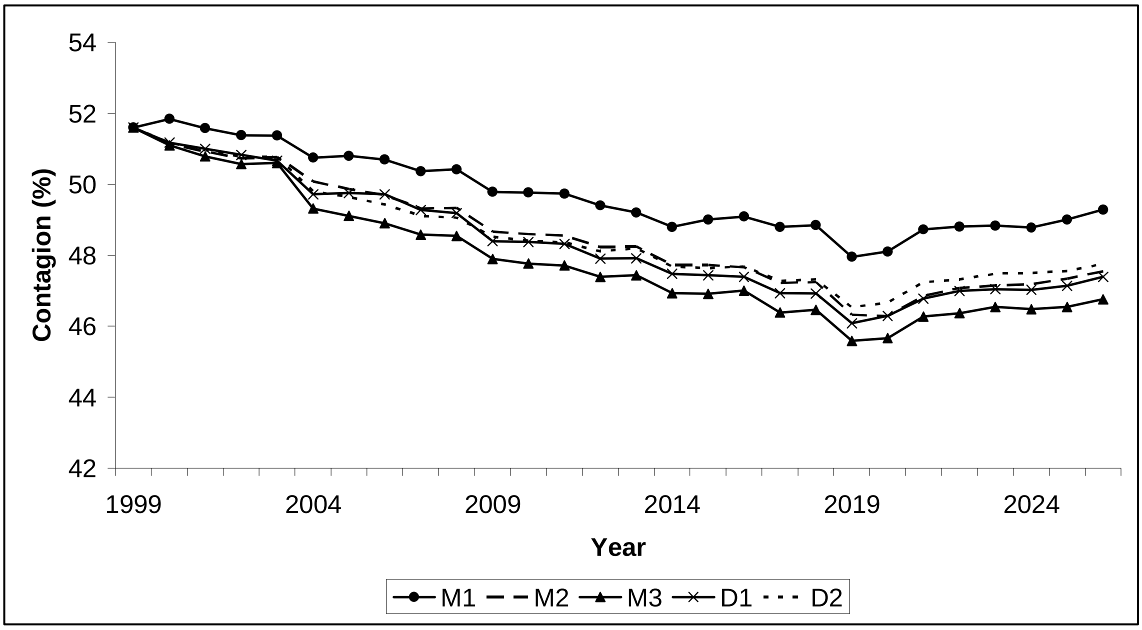

Figure 6.9 Time-series of landscape contagion for human-activity scenarios. Values decrease through time before increasing around the year 2019. This change in trend occurs at the point in time at which Scrub contributions to total number of landscape increase (Figure 6.6). This coincidence suggests this is the point at which, in some parts of the landscape, Forest is replacing Scrub as the land-cover matrix.

is proceeding in a manner consistent with the spatial landscape structure previously observed in the landscape (in 1999, Figure 2.10). This pattern is pleasing in some respects as it suggests that the model is reproducing landscape change in a spatially-similar manner to previous change (for example in the interval 1984 – 1999, Figure 2.10). However, given the non-stationarity of landscape processes (section 3.3.4) this should not be taken as indication of the validity of the model (discussed further in section 6.6 and chapter seven).

Contagion is a landscape pattern metric that quantifies the degree of aggregation in a landscape, and is essentially the probability that two randomly chosen pixels belong to the same land-cover class (Hargis *et al.* 1997). Increasing contagion suggests larger and simpler edge configurations. Model results show landscape contagion decreasing through time until year 2020, at which point it stabilises and begins to increase (Figure 6.9). Given that the model also shows number of patches increases in time (Figure 6.6), decreases in contagion would be expected and the observed increases are counter-intuitive. However, on closer examination of data on the number of patches, the point in time at which contagion begins to increase is also the point at which the number of Scrub patches begins to increase and the increase in number of Forest patches slows

(Figure 6.7). These patterns suggest that at this point in simulated time the landscape is beginning to shift from a Scrub matrix containing Forest patches to a Forest matrix containing Scrub patches. This occurs as the abundance of Forest land-cover has become so great that new Forest pixels are no longer necessarily occurring within a matrix of Scrub. Rather, in some areas new Forest pixels result in the merger of Forest patches, isolating areas of Scrub and increasing the number of Scrub patches. The number of Forest patches does continue to rise however, as this transition is not occurring at the same point in time simultaneously across the entire study area.

6.3.2 Human-Related Wildfire-Behaviour Parameters

For the values examined (0 – 20 years), the 'recent burned years' parameter (section 4.5.2.1) shows no recognisable trend or influence on wildfire regime behaviour (Table 6.4). There is no statistically significant difference between this parameter and mean largest fires or mean total burned areas at the 95% confidence interval (p = 0.64 and p = 0.15 respectively). That is, the effect of the (modelled) preference of humans to visit non-burned rather than burned areas has no influence on wildfire regime behaviour.

Table 6.4 Wildfire statistics for varying 'recent burned years' values. No significant change in β values are observed between parameter values. Mean largest fire and mean total burned areas show no clear trend. Error values are ± 1 SEM.

RBY (years)	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
20	152	0.86 ± 0.10	7.00 ± 0.31	1.27 ± 0.12	0.97
10	154	1.46 ± 0.17	7.71 ± 0.33	1.31 ± 0.09	0.99
5	154	2.29 ± 0.76	12.57 ± 1.79	1.24 ± 0.10	0.98
0	152	1.50 ± 0.41	7.90 ± 0.63	1.32 ± 0.09	0.99

Results indicate smaller maximum fire shells produce reduced mean largest fire and mean total burned areas (Table 6.5), as would be expected given the function of the parameter (i.e. controlling the probability of spread – section 4.5.3). Maximum fire shell values also show a positive relationship with β values – as largest fire sizes decrease, the ratio of large to medium to small fires decreases. However, again, the SEM of β suggests that this relationship should be treated with caution. Furthermore, power-law fits to the data are poorer for a maximum shell of 10 (though still very good). This is probably because the long tail of the power-law frequency-area distribution is being truncated (as largest fire size possible is 'artificially' restricted). As for section 6.3.1, no effect on land-cover composition or configuration is evident.

Table 6.5 Wildfire statistics for 'maximum fire shell' scenarios. As would be expected, largest fire and total area burned decrease with maximum fire shell. Variability in β means that decreases are not significant. Error values are ± 1 SEM.

Maximum Shell	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
1000	154	1.46 ± 0.17	7.71 ± 0.33	1.31 ± 0.09	0.99
100	154	1.21 ± 0.16	9.10 ± 0.37	1.26 ± 0.12	0.98
10	155	0.09 ± 0.00	2.62 ± 0.06	1.12 ± 0.19	0.93

6.3.3 Ignition Cause

One of the novel features of this simulation model is the spatially and temporally-explicit representation of wildfire ignition from both 'natural' (i.e. lightning) and human sources. This feature allows the analysis of wildfire regimes of fires ignited by these different sources. Wildfire events for all three model repetitions for each scenario examining ignition cause are aggregated before analyses. This was done in an attempt to ensure sufficient numbers of lightning fires to examine the regime using β, as care must be taken when interpreting β for less than 100 fires (Malamud *et al.* 2005). This aggregation prevents an estimation of error for mean largest fire area, mean total burned area and land-cover proportions. An examination of wildfire statistics shows that β values are consistently greater for human-caused fires than for lightning (Table 6.6). This suggests that large lightning-caused fires are rarer relative to smaller lightning fires. However, once again, the SEM of β values indicate that there is no significance difference between the causes. Standard errors are particularly large for lightning fires due to the low numbers of fires.

In an effort to overcome the low number of lightning fires, scenarios were re-run, forcing the occurrence of more lightning fires (by setting fire-frequency parameter m = 200, see section 4.5.2.1). More equal numbers of lightning and human-caused fires suggest no difference in β values between the causes (Table 6.7). The differences between β values for these different numbers of lightning fires highlights the dangers of using β with too few (i.e. < 100) fires. On examination of largest fire and total burned areas however, largest fire areas are found to be smaller for lightning fires, with

Table 6.6 Wildfire statistics as a function of cause for human activity scenarios. Values of β are consistently larger for human-caused fires, but variability highlights this difference may not be statistically significant. Largest fire and total burned area are also consistently greater for human-caused fires, which would be expected given the greater number of human-caused fires (see Table 6.7 for results with equal numbers of fires). Error values are ± 1 SEM.

Scenario	No. of Fires	Largest Fire (km^2)	Total Burned Area (km^2)	β	r^2
Human					
M1	405	6.50	40.05	1.33 ± 0.10	0.97
M2	411	4.93	34.26	1.30 ± 0.08	0.98
M3	405	1.22	19.82	1.28 ± 0.09	0.98
D1	405	2.25	23.22	1.31 ± 0.08	0.98
D2	431	2.41	25.46	1.33 ± 0.10	0.97
Lightning					
M1	57	0.75	3.16	1.07 ± 0.22	0.96
M2	51	0.57	3.46	1.10 ± 0.33	0.96
M3	52	0.52	2.16	1.25 ± 0.25	0.98
D1	39	0.75	2.68	1.12 ± 0.17	0.99
D2	42	0.93	2.89	1.17 ± 0.11	0.99

Table 6.7 Wildfire statistics as a function of cause for human activity scenarios with modified fire ignition. With greater numbers of 'lightning' fires than human, β values show no significant difference. Mean largest fire remain consistently greater for human-caused fires however. Error values are ± 1 SEM.

Scenario	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
Human					
M1	135	2.06 ± 0.47	7.87 ± 0.83	1.27 ± 0.14	0.98
M2	138	1.02 ± 0.21	6.10 ± 0.51	1.31 ± 0.11	0.98
M3	135	1.85 ± 0.08	7.35 ± 0.22	1.29 ± 0.12	0.98
D1	135	1.58 ± 0.04	7.04 ± 0.42	1.28 ± 0.12	0.98
D2	144	1.38 ± 0.13	8.28 ± 0.41	1.29 ± 0.11	0.98
Lightning					
M1	140	0.36 ± 0.05	3.94 ± 0.11	1.25 ± 0.13	0.98
M2	170	0.42 ± 0.03	4.91 ± 0.31	1.26 ± 0.13	0.98
M3	161	0.32 ± 0.02	4.59 ± 0.20	1.24 ± 0.14	0.97
D1	168	0.32 ± 0.00	4.55 ± 0.04	1.27 ± 0.11	0.98
D2	153	0.46 ± 0.03	5.38 ± 0.50	1.26 ± 0.11	0.98

correspondingly less total burned area. These smaller burned areas may be a result of the spatial location of lightning fire ignition at elevations above 1000 m ASL (section 4.5.2.2). As fires tend to burn upslope, the area available to burn (upslope) is reduced if ignition occurs at higher elevations. Alternatively, fire ignition in locations near areas of high human activity (e.g. near roads and trails, section 4.5.2.2) may result in human-caused fires preferentially burning highly flammable Scrub, due to agricultural abandonment of adjacent areas and repeated localised burning in areas of high human ignition risk.

To investigate the potential causes of the difference in mean largest fire and mean burned area between causes, the relative frequencies of ignition and the relative proportions of burned area in each land-cover can be examined. For ignition frequency for each land-cover *y* we calculate the ratio R_i:

$$R_i = \frac{prop_{iy}}{prop_y} \qquad \text{Eq. 6.2}$$

where $prop_{iy}$ is the proportion of total wildfire ignitions in land-cover *y* and $prop_y$ is the mean annual proportion of total area occupied by land-cover *y*.
For burned area for each land-cover *y* we calculate the ratio R_b:

$$R_b = \frac{prop_{by}}{prop_y} \qquad \text{Eq. 6.3}$$

where $prop_{by}$ is the proportion of total burned area burned in land-cover *y*. $R_x \approx 1.0$ indicates unbiased ignition or burning. $R_x > 1.0$ indicates a bias toward ignition or burning of a specific land-cover type (relative to the total area occupied by the land-cover in the landscape). In addition to model replicates considering both human- and lightning-caused fires, model replicates considering human-caused and randomly located fires were also examined. Considering randomly-located fires provides insight into the relative effects of any biases in land-cover ignition or burning. Results for R_i indicate a bias in lightning fires to ignite in Forest land-covers, while human-caused fires are biased toward ignition in Agricultural land-covers, and randomly located fires are unbiased (Figure 6.10). R_b values indicate a bias in lightning fires burning of Forest land-cover types, and that human-caused and randomly-located fires burning Scrub (Figure 6.11).

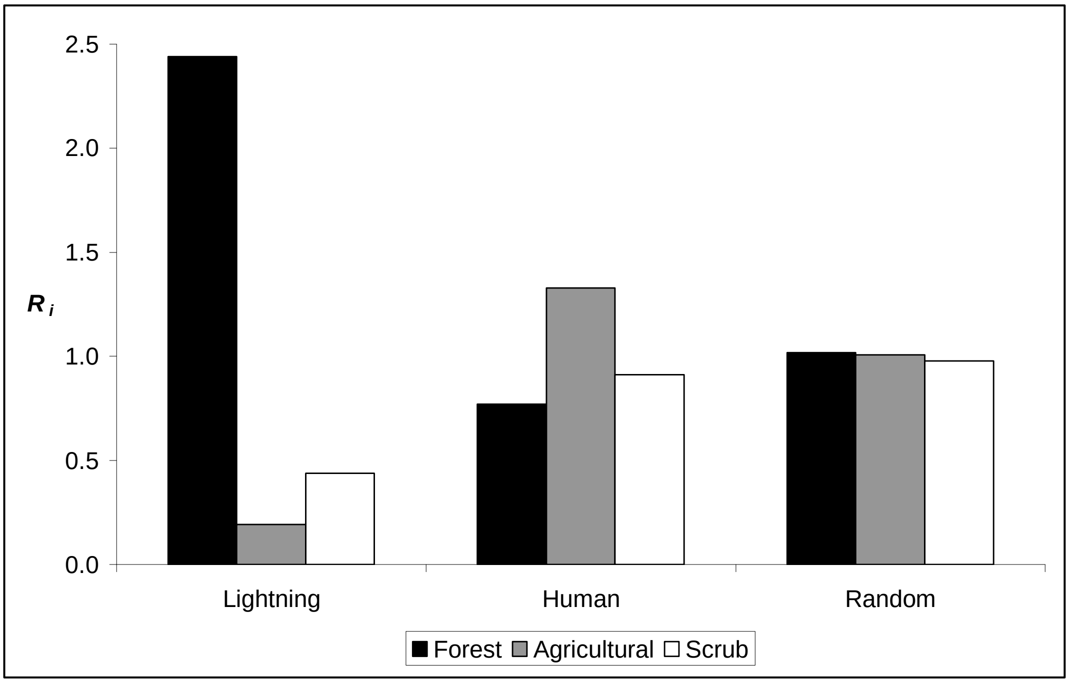

Figure 6.10 Mean R_i for lightning, human-caused and randomly located fires. R_i is defined in equation 6.2. Results indicate a bias in lightning fires to be ignited in forest land-covers, a bias in human-caused fires to be ignited in agricultural land-covers, and an absence of bias in the location of randomly located fires. SEM for all values < 0.05.

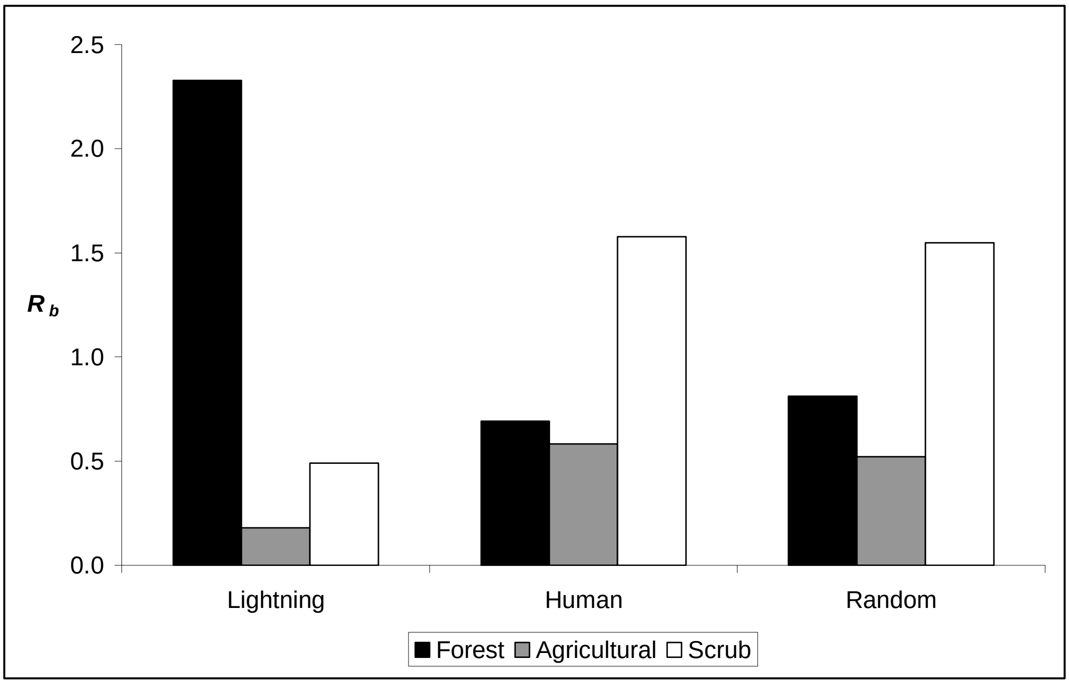

Figure 6.11 Mean R_b for lightning, human-caused and randomly located fires. R_b is defined in equation 6.3 Results indicate a bias in human-caused and randomly located fires toward the burning of scrub, but that lightning fires are biased toward burning forest land-covers. SEM for all values < 0.08.

These results support the reasons proposed above for the greater largest fire and total burned areas observed for human-caused fires (i.e. repeated localised burning and abandonment of neighbouring agricultural land in areas of high human ignition risk). Despite the bias of human-caused fires toward ignition in Agricultural land-covers, the highly flammable (and abundant) nature of adjacent Scrub results in larger fires. This nature also leads fires ignited at random locations (i.e. equal numbers of ignitions in each land-cover) to preferentially burn Scrub. Lightning fires are more likely to ignite in lower flammability Forest covers because of their predominance at higher elevations (section 4.5.2.2) where these land-covers are dominant.

6.4 CLIMATE CHANGE AND WILDFIRE REGIMES

To examine the potential impacts of a changing climate on the wildfire regime, four scenarios of temperature and precipitation change were examined using the LFSM alone (Table 6.8). Temperature scenarios reflect the minimum and maximum IPCC estimates for global temperature change over the interval 2000 – 2099 (IPCC SRES scenarios B1 and A2, IPCC 2007 – T1 and T2 respectively). Uncertainty in precipitation change is greater than for temperature change and therefore only one scenario of Mediterranean precipitation change is considered (that lying between the extremes of projected change – scenario SRES A1B, IPCC 2007).

Table 6.8 Climate change scenarios. These scenarios are used to examine the impact of potential climate change on wildfire regimes. Temperature and precipitation values are based on recent estimates for 2099 from global circulation models (IPCC 2007).

Scenario	Annual Temperature Change (°C/yr)	Annual Precipitation Change (mm)
None	0.000	0.0
T1	0.018	0.0
P	0.000	-1.3
T1P	0.018	-1.3
T2P	0.040	-1.3

LFSM repetitions are simulated for 100 years (i.e. to 2099) with an absence of human activity. Human activity was not considered as the socio-economic model does not currently consider the influence of climate change or extend over 200 years (see section 6.6). Results show no significant variation in β values given the SEM. Mean largest fire and mean total burned areas increase in hotter and drier climates, with greatest areas at the upper extreme of current estimates of climate change (Table 6.9 IPCC 2007). As a result, Forest land-cover area decreases at the same time as increases in post-fire Scrub (Figure 6.12).

Table 6.9 Wildfire statistics for scenarios of climate change. Relative to its variability, β shows no significant difference between the scenarios (scenarios defined in Table 6.8). Largest fires and total burned areas are larger for hotter, drier scenarios. r^2 values are for power-law fits to wildfire burned area data. Error values are ± 1 SEM.

Scenario	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
None	288	49.07 ± 3.07	162.5 ± 9.7	1.26 ± 0.07	0.99
T1	306	58.46 ± 11.59	215.6 ± 90.0	1.24 ± 0.07	0.98
P	286	139.80 ± 2.67	490.5 ± 24.5	1.25 ± 0.06	0.99
T1P	287	142.71 ± 5.53	969.6 ± 27.5	1.23 ± 0.06	0.99
T2P	297	292.19 ± 24.88	2232.9 ± 317.1	1.19 ± 0.05	0.99

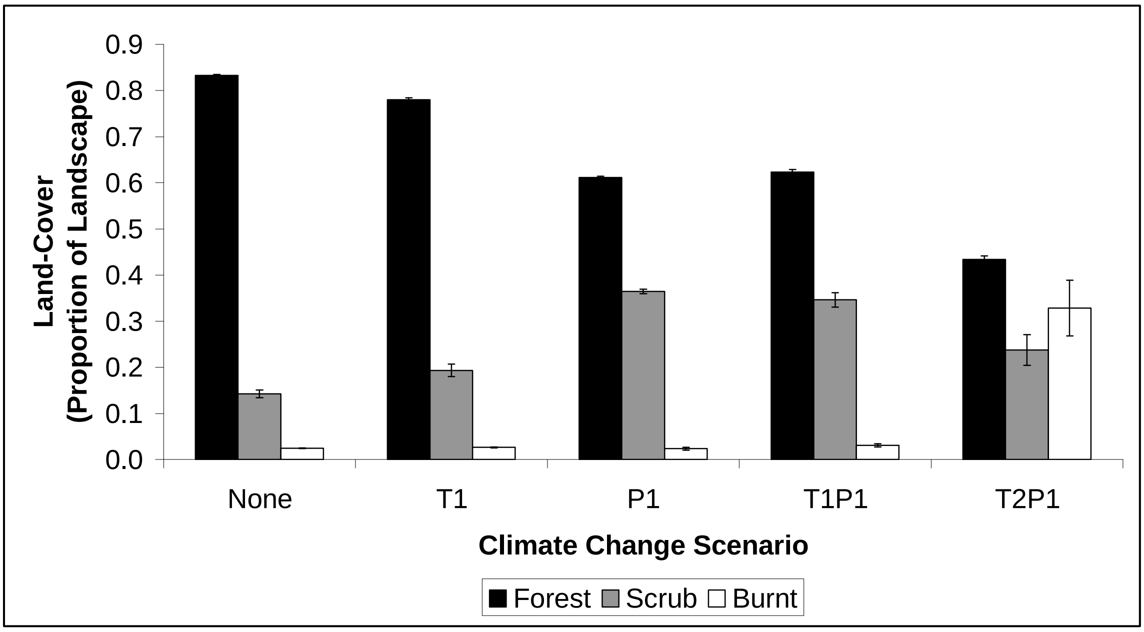

Figure 6.12 Land-cover landscape proportions for scenarios of climate change. For increasingly hot and dry scenarios (see Table 6.8) scrub area increases commensurate with decreases in forest land-cover area. All values are for the final landscape state of the model scenario run (year 2099). Error bars are ± 1 SEM.

6.5 Landscape Fire Succession Model Dynamics

The results of the sensitivity analyses of the LFSM highlighted the sensitivity of forest fire cellular automata models to the land-cover flammability parameterisation (section 4.6). Total land-cover flammability, the range of land-cover flammability probabilities and the spatial configuration of land-cover flammability probabilities are examined here in more detail using the LFSM (i.e. human activity and land-use are absent). Three model repetitions were made for each parameter set with each model repetition considering 250 years. Mean values for each state-variable from the three repetitions were calculated. This experimental design ensures the dynamics of the model wildfire regime for each parameter set can be examined appropriately.

6.5.1 Total Land-Cover Flammability

As the total flammability of all land-cover types represented in the model landscape increases, total burned area would be expected to increase. Nine sets of land-cover flammability probabilities, with varying total flammability, are examined (Table 6.10) to examine the sensitivity of the wildfire regime to total land-cover flammability.

Table 6.10 Land-cover flammability probabilities used to examine effect of total landscape flammability on model wildfire regime. The 'Base' values are those used when examining the effect of human activity (section 6.3)

	Pine	T. Forest	Deciduous	Scrub	Holm Oak	**Total**
TF1	0.19	0.19	0.18	0.20	0.18	**0.94**
TF2	0.20	0.20	0.19	0.21	0.19	**0.99**
TF3	0.21	0.21	0.20	0.22	0.20	**1.04**
TF4	0.22	0.22	0.21	0.23	0.21	**1.09**
Base	0.23	0.23	0.22	0.24	0.22	**1.14**
TF5	0.24	0.24	0.23	0.25	0.23	**1.19**
TF6	0.25	0.25	0.24	0.26	0.24	**1.24**
TF7	0.26	0.26	0.25	0.27	0.25	**1.29**
TF8	0.27	0.27	0.26	0.28	0.26	**1.34**

A strong negative relationship between total flammability and values of β is observed (Figure 6.13). This relationship indicates that as total land-cover flammability decreases, large fires become rarer relative to smaller fires. This behaviour is also indicated by trends in mean largest fire and mean total burned area which increase with total flammability (Table 6.11). Such behaviour is not unexpected and highlights that while the power-law frequency-area relationship holds β values change significantly (with respect to variability). That is, the nature of the relationship between the frequency of fires and their size does not change, but the frequency of fires of a given sizes does.

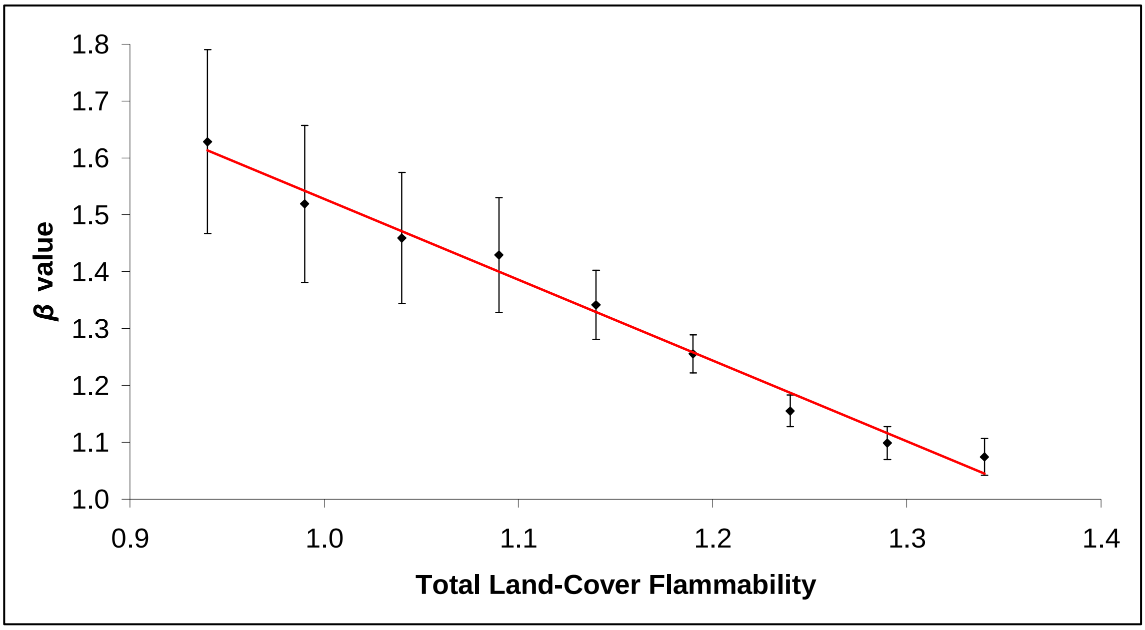

Figure 6.13 Relationship between β and total land-cover flammability. A strong (r^2 = 0.97) negative relationship is evident indicating that as total vegetation flammability increases large fires become more common with respect to smaller ones.

Table 6.11 Wildfire statistics for varying total land-cover flammability. Values of β decrease for increasing total flammability. Mean largest fires and mean total burned areas increase with increasing total flammability. r^2 values are for power-law fits to wildfire burned area data.

Total Flammability	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
0.94 (*TF1*)	1216	0.31 ± 0.02	14.6 ± 0.1	1.63 ± 0.16	0.96
0.99 (*TF2*)	1237	0.73 ± 0.10	23.2 ± 0.5	1.52 ± 0.14	0.96
1.04 (*TF3*)	1230	2.62 ± 0.58	40.8 ± 2.1	1.46 ± 0.12	0.97
1.09 (*TF4*)	1258	8.04 ± 0.48	81.7 ± 2.1	1.43 ± 0.10	0.97
1.14 (*Base*)	1256	50.41 ± 3.80	593 ± 66	1.34 ± 0.06	0.98
1.19 (*TF5*)	1241	162.26 ± 15.75	3048 ± 292	1.26 ± 0.03	0.99
1.24 (*TF6*)	1256	380.19 ± 4.46	14571 ± 563	1.16 ± 0.03	0.99
1.29 (*TF7*)	1260	540.65 ± 7.78	30886 ± 475	1.10 ± 0.03	0.99
1.34 (*TF8*)	1246	640.13 ± 2.02	45401 ± 592	1.07 ± 0.03	0.99

The increases in mean largest fire and mean total burned area are reflected in large shifts in landscape composition and configuration (Figure 6.14). For total flammability ≤ 1.19 Forest land-covers are dominant, with a patchy configuration (i.e. high number of land-cover patches). For flammability ranges > 1.24, Scrub becomes the dominant land-cover and the number of land-cover patches drops significantly (because of large burned areas and subsequent re-vegetation). In this manner, total flammability of all land-cover types in the landscape has important implications for land-cover.

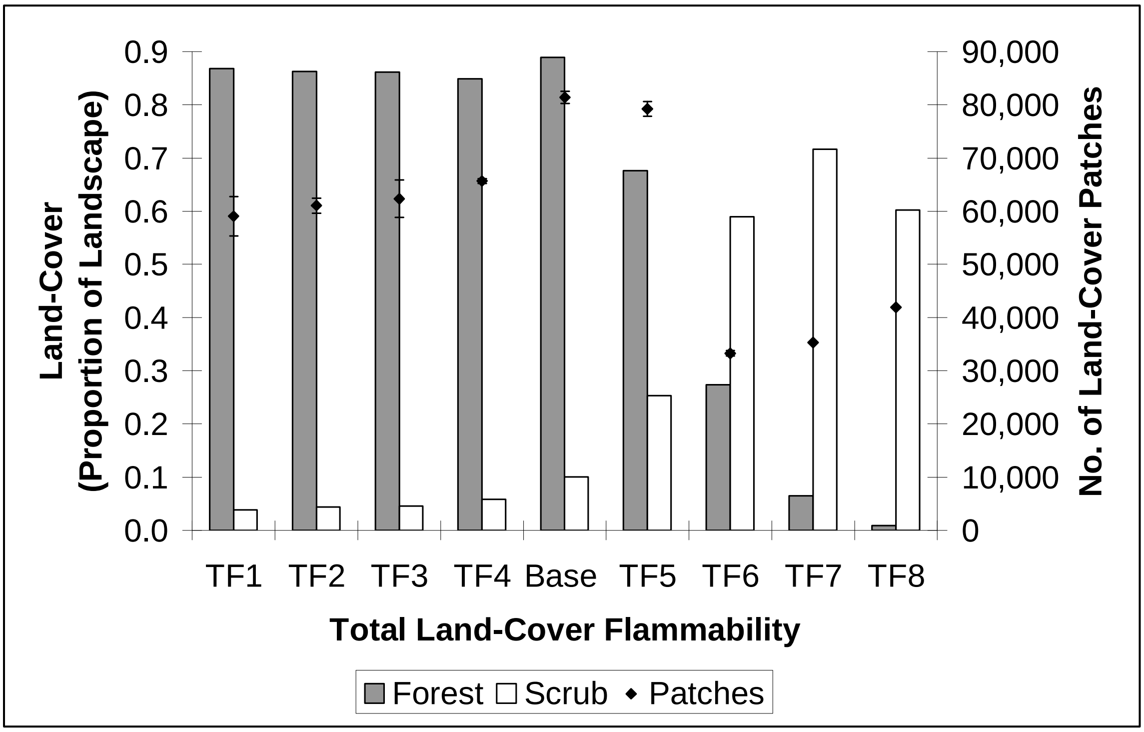

Figure 6.14 Land-cover landscape proportions for varying total land-cover flammability. Model repetitions with total flammability ≤ 1.19 (i.e. *TF1 – Base*; see Table 6.10) show no significant variation in land-cover proportions or patchiness. Model repetitions with range > 1.19 show reversed proportions of Scrub and Forest (later-succession vegetation relative to scrub) land-covers, the latter becoming dominant. Land-cover becomes increasingly patchy for greater ranges of flammability values. SEM for all land-cover values ≤ 0.01. Proportions for Burnt land-cover is not shown. Error bars for patches are ± 1 SEM.

6.5.2 Range of Land-Cover Flammability

The range of land-cover flammability probabilities across all land-cover types is a second attribute of flammability probabilities that might have an influence on modelled wildfire regimes. Maximum and minimum vegetation flammability probabilities are varied to increase and decrease the range of values relative to the 'base' values used in the scenarios of human activity (Table 6.12). Other land-cover probabilities are modified to ensure total land-cover flammability remains constant. Results indicate that β values do not vary significantly between land-cover flammability ranges ≤ 0.0350 (Table 6.13).

Table 6.12 Land-cover flammability probabilities used to examine effect of range of flammability probabilities on model wildfire regime. Total land-cover flammability is held constant while varying maximum and minimum values. The 'Base' values are those used when examining the effect of human activity (section 6.3).

	RF1	RF2	RF3	Base	RF4	RF5	RF6
Pine	0.2300	0.2300	0.2300	0.2300	0.2290	0.2280	0.2280
Transition Forest	0.2300	0.2300	0.2300	0.2300	0.2290	0.2280	0.2280
Deciduous	0.2050	0.2100	0.2150	0.2200	0.2250	0.2270	0.2275
Scrub	0.2700	0.2600	0.2500	0.2400	0.2320	0.2300	0.2290
Holm Oak	0.2050	0.2100	0.2150	0.2200	0.2250	0.2270	0.2275
Sum	1.14	1.14	1.14	1.14	1.14	1.14	1.14
Maximum	0.2700	0.2600	0.2500	0.2400	0.2320	0.2300	0.2290
Minimum	0.2050	0.2100	0.2150	0.2200	0.2250	0.2270	0.2275
Range	**0.0650**	**0.0500**	**0.0350**	**0.0200**	**0.0070**	**0.0030**	**0.0015**

Table 6.13 Wildfire statistics for varying range of land-cover flammability. Values of β decrease for increasing total flammability. Mean largest fires and mean total burned areas increase with increasing total flammability. r^2 values are for power-law fits to wildfire burned area data.

Flammability Range	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
0.0650 (*RF1*)	1242	413.75 ± 3.37	24688.3 ± 726.7	1.15 ± 0.04	0.99
0.0500 (*RF2*)	1215	303.76 ± 7.50	10918.1 ± 342.9	1.18 ± 0.03	0.99
0.0350 (*RF3*)	1231	92.18 ± 10.01	829.3 ± 113.7	1.35 ± 0.05	0.99
0.0200 (*Base*)	1256	50.41 ± 3.80	593.3 ± 66.7	1.34 ± 0.01	0.98
0.0070 (*RF4*)	1254	26.21 ± 4.29	211.5 ± 8.9	1.35 ± 0.08	0.97
0.0030 (*RF5*)	1243	27.16 ± 3.25	260.6 ± 26.1	1.36 ± 0.08	0.98
0.0015 (*RF6*)	1255	39.04 ± 6.27	357.7 ± 48.4	1.36 ± 0.07	0.98

The marked increases in mean largest fire and mean total burned area are reflected in large shifts in landscape composition and configuration (Figure 6.15) in similar manner to that for total land-cover flammability. For flammability ranges ≤ 0.0350 Forest land-covers are dominant, with a very patchy configuration (i.e. high number of land-cover patches). For flammability ranges > 0.0350 Scrub becomes the dominant land-cover and the number of land-cover patches drops significantly (because of large burned areas and subsequent re-vegetation).

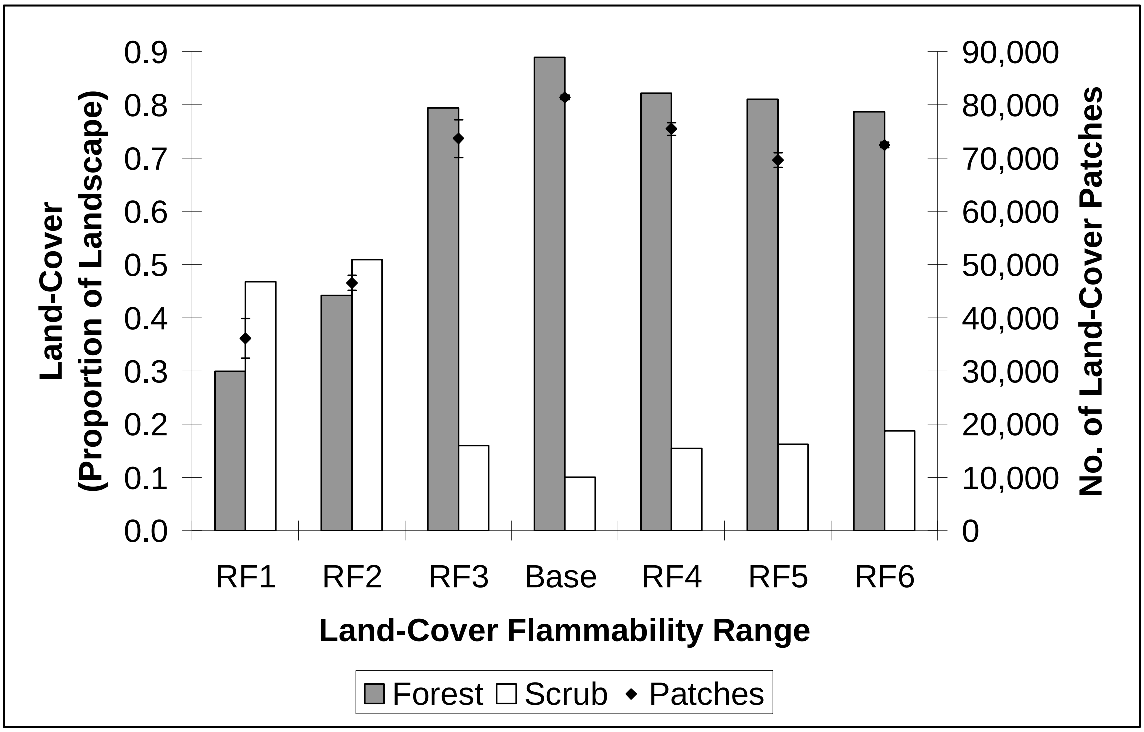

Figure 6.15 Land-cover landscape proportions for varying range of land-cover flammability. Model repetitions with range ≤ 0.0350 (i.e. Sets 3 – 6; see Table 6.12) show no significant variation in land-cover proportions or patchiness. Model repetitions with range > 0.0350 show reversed proportions of Scrub and Forest (later-succession vegetation relative to scrub) land-covers, the latter becoming dominant. Land-cover becomes increasingly patchy for greater ranges of flammability values. SEM for all land-cover values ≤ 0.05. Proportions for Burnt land-cover are not shown. Error bars for patches are ± 1 SEM.

It is unclear why greater ranges of flammability between land-cover classes would result in this threshold in themselves. The similar shifts in land-cover observed as a result of variation in total land-cover indicate that the maximum land-cover flammability may be the cause. As discussed above (section 4.6.2), percolation-type cellular automata models such as the wildfire spread model here exhibit critical threshold-type behaviour, often with respect to flammability probabilities (e.g. Ratz 1995, Perry and Enright 2002b). A critical value of land-cover flammability probability, such as the percolation threshold value p_c discussed in section 5.5.3, may be present here. For both total and range of land-cover flammability, threshold behaviour is observed for parameter sets with maximum flammability probabilities in the interval 0.25 – 0.26 (i.e. *TF5 – TF6*, Table 6.10, and *RF2 – RF3*, Table 6.12). To examine if a threshold is present between these values, the maximum flammability probabilities of the base parameter were substituted for those values (Table 6.14).

Table 6.14 Modified land-cover flammability probabilities used to examine the effect of maximum flammability probability on wildfire regime. Results from total (Table 6.11) and ranges of land-cover flammability probabilities suggest the potential presence of a critical threshold in maximum probability in the interval 0.25 – 0.26. Maximum probabilities for the base parameter set are therefore substituted by these values.

	Base	Base(0.25)	Base(0.26)
Pine	0.23	0.23	0.23
Transition Forest	0.23	0.23	0.23
Deciduous	0.22	0.22	0.22
Scrub	0.24	0.25	0.26
Holm Oak	0.22	0.22	0.22
Sum	1.14	1.15	1.16
Maximum	**0.24**	**0. 25**	**0. 26**
Minimum	0.22	0.22	0.22
Range	0.02	0.03	0.04

Results from the modified flammability probabilities (Table 6.15) indicate large differences in wildfire statistics (relative to standard error of measurement) between parameter sets *Base(0.25)* and *Base(0.26)*. For maximum flammability probability > 0.26, mean largest fire and mean burned areas are increased and β is decreased. Combined with the results above, this suggests that with the current model construction a critical threshold in land-cover flammability probability lies in the interval 0.25 – 0.26.

Table 6.15 Wildfire statistics for modified land-cover flammability probabilities. Values of β decrease for increasing total flammability. Mean largest fires and mean total burned areas increase with increasing total flammability. r^2 values are for power-law fits to wildfire burned area data.

Scenario	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
Base	1256	50.41 ± 3.80	593.27 ± 66.27	1.34 ± 0.06	0.98
Base(0.25)	1245	75.10 ± 6.10	871.07 ± 74.21	1.34 ± 0.05	0.99
Base(0.26)	1238	323.91 ± 4.86	12429.79 ± 244.53	1.18 ± 0.04	0.99

6.5.3 Land-Cover Configuration

The final aspect of land-cover flammability examined here is spatial distribution and configuration. Land-cover maps with randomly generated configurations were generated using the same method as described in section 5.5.3 for use as initial landscape state. Proportions of land-cover were approximately equal for all runs except $p = 0.6$, when spanning clusters occurred. Therefore two sets of model replicates were run for $p = 0.6$, one with Holm Oak as the dominant initial land-cover (*LF* – low flammability) and one with scrub as the dominant land-cover (*HF* – high flammability). 'Time in state' parameter values (section 4.4.1) were set to a value of one for all pixels. Results indicate β values are not sensitive to original landscape configuration (Table 6.16).

Table 6.16 Wildfire statistics for varying initial landscape configuration. Maps of random land-cover configurations were generated using percolation parameters specified and the method presented in section 5.5.3. No significant variation in β is observed. Greatest mean burned areas and mean largest fires are observed for the Scrub-dominated (i.e. most flammable) landscape.

Percolation Parameter p	Mean No. of Fires	Mean Largest Fire (km^2)	Mean Total Burned Area (km^2)	β	r^2
0.2	1247	49.09 ± 2.34	347.07 ± 41.81	1.38 ± 0.07	0.98
0.4	1265	48.48 ± 7.02	369.26 ± 47.47	1.35 ± 0.08	0.98
0.5	1265	33.76 ± 4.40	231.77 ± 5.60	1.36 ± 0.08	0.98
0.6 *(LF)*	1243	26.98 ± 2.56	179.69 ± 11.08	1.38 ± 0.09	0.97
0.6 *(HF)*	1236	75.13 ± 10.69	497.43 ± 39.36	1.36 ± 0.07	0.98

Examination of mean largest fire and mean total burned area indicates slight decreases with increased initial mean patch size (i.e. increased percolation parameter), with the exception of the initial scrub-dominated landscape. Greatest mean largest fire and mean total burned area are to be expected for the scrub-dominated landscape given the high flammability of this land-cover and the high connectivity of a landscape spanning cluster. Values of β do not vary (relative to SEM) for different starting percolation-parameter values, suggesting that wildfire frequency-area scaling is not sensitive to initial land-cover configuration.

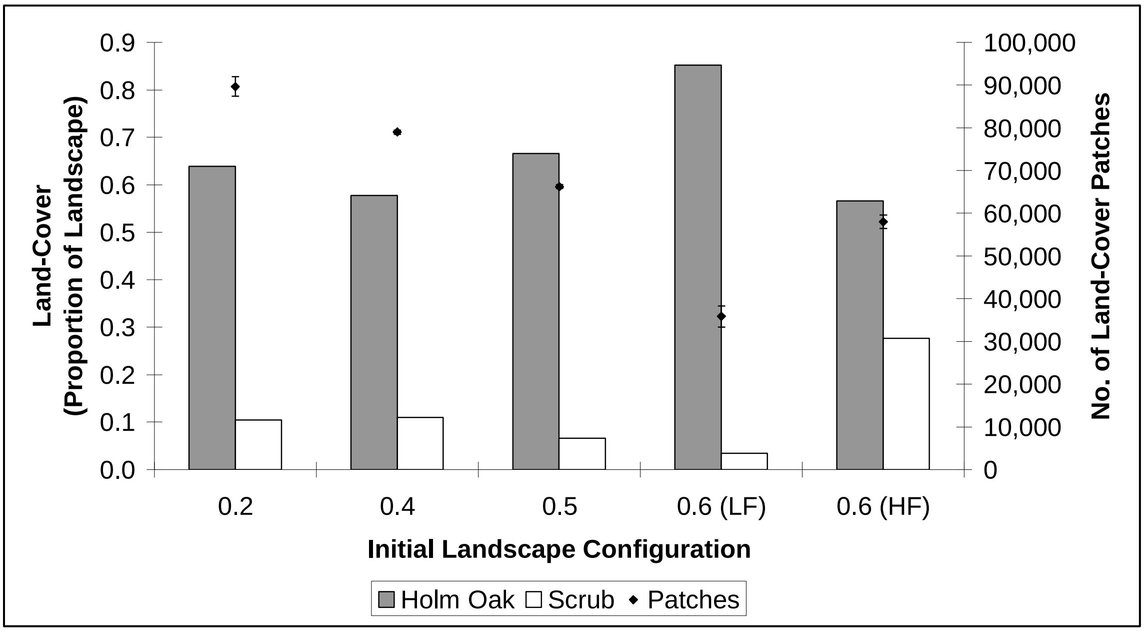

Figure 6.16 Land-cover landscape proportions for varying initial land-cover configurations. Holm Oak is the resulting dominant land-cover regardless of initial landscape pattern. However, for the initially Scrub-dominated landscape (0.6 *HF*) Scrub does compose a greater proportion than for other initial landscapes. Landscapes are less patchy (i.e. contain fewer land-cover patches) for less patchy initial landscapes. SEM for all land-cover values ≤ 0.02. Error bars for patches are ± 1 SEM.

From varying initial landscape configurations but similar landscape compositions, results show similar resulting landscape compositions dominated by Holm Oak (Figure 6.16). Resulting landscape patchiness reflects initial patchiness, with lower numbers of land-cover patches for greater *p* values.

6.6 DISCUSSION

Spatially-explicit simulation models are a useful tool for examining the broader environmental impacts of processes and activities occurring at finer scales and extents. Simulation models offer a framework to structure theories and information about the impacts of individuals' activities on activities at larger scales and extents. Such an approach is generative in nature – it considers the implications of microscale processes for macroscale phenomena (Brown *et al.* 2006). For example, in SPASIMv1 long-term processes such as the interaction between vegetation and wildfire dynamics are reconciled with human activity on an annual basis (even if not modelled explicitly). Broad epistemological issues regarding the scale and structure of environmental systems for the construction and evaluation of environmental simulation models are discussed in detail in the next chapter.

6.6.1 Human Activity and Wildfire

The model results presented above suggest mean largest wildfire and mean total burned area will increase if agricultural activity declines, that change in land-cover composition is driven more by human activity than wildfire, and that landscape configuration varies little across the scenarios of human activity simulated (section 6.3). However, the power-law frequency-area scaling exponent β was not found to vary significantly in response to human activity. In contrast, model replicates for longer time intervals in the absence of human activity did find variation in β values as a function of total and maximum land-cover flammability probabilities (section 6.5). In the light of these results, the remainder of this chapter discusses the appropriateness of the scale and structure of SPASIMv1 for examining the relationship between human activity and wildfire regimes. The utility of the β power-law frequency-area exponent for examining wildfire regimes at the landscape scales considered here are also discussed.

Model replicates for the scenarios of human activity considered only 28-years in the interval 1999 – 2026. Relative to the 150-year duration over which major shifts in land-cover occur due to vegetation dynamics (e.g. Figure 4.4) this 28-year interval is too short to reveal any changes in the rank order of land-covers. Results indicate that over the 28-year period land-cover change is driven largely by human activity – increases in scrub commensurate with decreases in pasture are observed for scenarios of reduced human (agricultural) activity. Wildfire scaling parameter β varies very little. Mean largest fire sizes increase with decreasing agricultural activity, highlighting a potential environmental issue resulting from abandonment. Such behaviour has been suggested in the literature (e.g. Moreno *et al.* 1998, Romero-Calcerrada and Perry 2002, 2004) giving confidence to model results (in these qualitative terms). The implication is that if agricultural abandonment and decline continues larger wildfires than are observed at present will become increasingly frequent, at least in the short term. With regards to the impacts of wildfire regimes on spatial land-cover configuration, there has been debate in the literature as to whether interactions between wildfire and agricultural abandonment will cause increased or decreased spatial land-cover heterogeneity (Lloret *et al.* 2002, Perez *et al.* 2003 – see section 2.4.4). Results for spatial land-cover configuration here suggest, over the decadal time interval at least, that whilst the number of land-cover patches will increase the spatial structure of the landscape will not change (section 6.3.1).

SPASIMv1 was not run for longer than the 28-year interval for several reasons. The first is an issue of representation. Uncertainties in socio-political, economic and demographic change over more than a couple of decades into the future make projection of agricultural land-use change over extended periods infeasible. This problem is highlighted by the failure of the stationarity assumption (the assumption that relationships between variables are constant in time) to hold in the study area over the interval 1984 – 1999 (Romero-Calcerrada and Perry 2004). As discussed above (section 3.3.4), this failure is more likely to be due to socio-economic changes such as the expansion of the European Union and consequent expected restructuring of the Common Agricultural Policy (CAP) than to changes in biophysical processes. Thus while the LFSM is based on a theory of relationships that are expected to hold over centuries (or at least, change in a gradual manner over this extent) this is unlikely to be the case for the ABM/LUCC model. Such a situation highlights the likely importance of future ABMs to represent agents that can 'learn' and adopt new land use strategies (e.g. Manson 2006). However, under changing conditions even the biophysical relationships may not remain stationary. For example, for the modelled effects of 100-year climate change on wildfire regimes, potential reciprocal influences of climate on vegetation dynamics, via changes in phenology for example (e.g. McCarty 2001), are not mechanistically represented.

A second issue, related to that of representational constraints, is one of computational constraints. Whilst LFSM analyses can extend over (simulated) centuries for the entire study area, the nature of the ABM/LUCC model means that replicates can only feasibly consider decades over the same spatial extent. The high computational demands of simulating the activities of over 6,000 agents across 900,000 grid cells, resulted in individual scenario replicates taking ~ 40 hours (on a desktop PC, 2.13 GHz, processor, 2 GB RAM). This led to the adoption of a methodology that used the output from ABM/LUCC model scenarios of human activities as input to the LFSM of vegetation and wildfire dynamics. This methodology allowed multiple replicates for varying wildfire parameters (e.g. land-cover flammability, ignition type etc.) to be examined, but prevents the representation of a direct interaction between wildfire occurrence and agricultural decision-making. As the ABM/LUCC model is currently constructed agents do not make decisions based upon wildfire occurrence. In reality, the costs of re-establishing arable farmland (e.g. an olive grove) are likely to influence land-cover decision making. Pasture burning is unlikely to present the same issues, and some farmers still actively burn land to improve pasture. Future iterations of model development will need to incorporate these issues and local stakeholders will need to be questioned further about their actions in such situations. However, this representation will need to be incorporated such that computational requirements are not increased significantly.

These issues highlight some of the problems of attempting to reconcile the influence(s) of short-term, local human activity on longer-term succession-disturbance dynamics over large(r) spatial extents. As the research emphasis here is the wildfire regime, SPASIMv1 must be able to represent the dynamics of wildfire and vegetation over greater than decadal time intervals, and across 1×10^3 km^2. Simulation of human activity over that spatial extent has proven possible, but similar temporal extents are currently unfeasible due to both representational and computational issues. To represent human activity over longer time intervals (i.e. up to 100 years) 'top-down' scenarios of change might be more useful. Broad-scale scenarios of socio-economic and political change for the Mediterranean Basin and Europe, such as the VISIONS scenarios (Rotmans *et al.* 2000), might be reconciled with the results of human-activity scenarios for agricultural land-cover composition and configuration found here. In turn, maps produced could be used in the same manner as was done here to parameterise the LFSM. For now, however, SPASIMv1 would be more applicable for shorter-term examination of LUCC trends in the study area in conjunction with local 'stakeholders'. The utility of SPASIMv1 for such a purpose is examined in a 'stakeholder evaluation' exercise presented in chapter eight. This model evaluation exercise is placed in the context of recent discussion about the practice of environmental simulation model 'validation' (chapter seven). Chapter seven also suggests that the 'non-certified' expertise of the locals may be valuable for improving model design and parameterisation. With this exercise in mind, model results for 20 years hence to 2026 (interviews were undertaken in 2006) were required, and thus model scenarios here run for 28 years from 1999 data. This two decade interval provided interviewees with an intuitive time interval over which to assess land-cover change and SPASIMv1's representation of it.

While SPASIMv1 suggests that human activity will have little effect on wildfire regimes (in terms of frequency-area scaling at least), the innovative approach to the representation of human and 'natural' (i.e. lightning) fires is useful and has produced novel results. The results suggest that 'lightning' fires (i.e. those igniting at elevations above 1000 m ASL) are biased toward igniting and burning Forest land-covers, while human-caused fires are biased toward igniting Agricultural land-covers and burning Scrub. A bias toward the burning of Scrub is also found in fires ignited at random locations in the landscape, highlighting the propensity of this most flammable land-cover to burn. The model rules determining the location of lightning and human-caused fires are based on previous research (Chuvieco and Congalton 1989, Chuvieco and Salas 1996, Moreno *et al.* 1998, Vazquez and Moreno 1998, Chuvieco *et al.* 1999, Vazquez and Moreno 2001). This literature is empirical in nature and in some cases is anecdotal. The bias observed in SPASIMv1, and the resulting variation in wildfire state-variables, indicates that patterns in the underlying spatial distributions of ignition may have important influences on wildfire regimes. Further research into the spatial distribution of wildfire ignition by cause will provide greater confidence in the assumptions of models that explicitly consider the spatial location of wildfire ignition. The representation of human ignition in the model here is one that assumes *unintentional* ignition and that there is a direct association between levels of human presence and ignition frequency, predominantly via proximity to roads, trails and recreation areas. However, given the predominance of wildfire caused by arson in regions such as Spain, establishing actual patterns of location of these *intentional* fires (if indeed there are any) will be difficult. Rumours abound about the reasons for arson (e.g. land tenure and boundary disputes between landowners are an often cited reason – Anonymous 2006) but unbiased evidence is difficult to come by and a model based on speculation are unlikely to be accepted. The explicit representation of wildfire ignition cause has produced some novel results regarding the differences in burned area by each cause, but poor empirical data and understanding still hinders the modelling of this aspect the wildfire regime.

6.6.2 Wildfire Frequency-Area Scaling

Just as the results here suggest no difference in β values between lightning- and human-caused fires, Malamud *et al.* (2005) found no significant difference between lightning and anthropogenic fires for the 30 year interval 1970 – 1999 across the conterminous U.S.. Malamud *et al.* (2005) did find significant variation in β values spatially for all fires regardless of ignition, highlighting an east-west gradient in values across the country (greater β values in the east U.S. compared with west). This variation in β values, in the light of an absence of trend due to ignition cause, suggests the classification criteria of Bailey's (1995) ecoregions (i.e. vegetation, climate and topography) capture the drivers of the frequency-area scaling of wildfire regimes at broad scales. 'Significant' variations in β values (i.e. variation between scenarios greater than variation between scenarios replicates) are only observed in the results here for extreme climate change over a 100 year interval and for variations in vegetation flammability probabilities over 250 year intervals. Two points are suggested by this variation in β (for results here and from Malamud *et al.* 2005), the first regarding the structure of the model and drivers of wildfire regime characteristics, the second regarding the appropriate measures of wildfire regime characteristics.

The importance of vegetation flammability probabilities highlighted by the LFSM sensitivity analyses and the results here, in association with the absence of an effect of ignition cause on β values, are consistent with the findings of Malamud *et al.* (2005) regarding the relative importance of anthropogenic versus environmental drivers. That is, environmental variables are a greater influence on wildfire frequency-area scaling than ignition source (i.e. lightning vs. anthropogenic). Further evidence of this is the biased burning of highly flammable scrub when ignition land-cover is randomly selected (i.e. equally distributed, section 6.3.3). Millington *et al.* (2006) advocate better use of the frequency-area scaling exponent β (for both empirical and modelled data) for investigation into the underlying drivers of wildfire-regime behaviour. Given the importance of land-cover flammability probabilities (i.e. spread probabilities) a focus on improving the representation of vegetation flammability and spread will be useful to better understand the importance of vegetation as a driver. For example, the direct consideration of wildfire-risk fuel models used by the U.S. National Fire Danger Rating System (Deeming *et al.* 1977, Burgan 1988) will improve LFSM representation of, and comparison with, empirical wildfire regimes (e.g. see Berjak and Hearne 2002). Furthermore, cellular automata models that represent the third (i.e. vertical) dimension of wildfire spread will be useful to model the differing effects of surface fires versus crown fires (i.e. those that burn in tree-crowns). Malamud *et al.* (1998) found distinct variation in β for abstract forest-fire cellular automata models with different parameters. However, the results here indicate that for cellular automata models that are more empirically-based and used to simulate temporal and spatial extents corresponding with real-world landscapes, β is not the most useful measure by which to characterise the wildfire regime. These more empirically-based models will be better analysed using other measures of the wildfire regime (e.g. total burned area).

Although previous research has suggested the β wildfire frequency-area scaling exponent is a useful measure by which to compare wildfire regimes (i.e. Malamud *et al.* 2005), minimal variation in β for the short-term model replicates considering human activity suggest that it will not be useful for comparing regimes that differ but subtly. Comparing regimes over larger spatial scales (i.e. comparing regimes at continental rather than regional scales) or regimes with more distinct differences in vegetation and climate will likely be a better use of this measure. Comparing regimes in time (for example before versus after some management intervention) may also be possible, but large time intervals would be required for the observation of adequate numbers of fires. Furthermore, this minimum number of events (e.g. Malamud *et al.* 2005 suggest a minimum of 100 events) will demand longer time intervals of observation for comparison of regimes in smaller regions or with infrequent burning. These observation issues highlight the problems of using the β scaling exponent for examining the effects of day-to-day, localised human activities on decadal wildfire activity. More useful for comparing results at these scales here was the examination of largest fire and total burned areas.

6.7 SUMMARY

Results from SPASIMv1 for human activity scenarios until 2026 suggest that as agricultural activity declines the frequency of large fire sizes will increase and more land-area will be burnt (section 6.3.1). However, human activity will remain the predominant influence on land-cover composition. Differences were found in wildfire behaviour for fires ignited by lightning compared to those ignited by human-causes (as represented by the model, section 6.3.3). LFSM replicates not considering human activity allowed longer time-series to be examined, and did show differences in β values as a function of land-cover flammability probabilities (section 6.5). The discussion (section 6.6) highlighted this point and suggested the wildfire frequency-area power-law distribution exponent β is more suited to examining and comparing wildfire regimes at larger time and space extents than the models replicates of human activity considered here. Given the representational difficulties of modelling socio-economic systems it was suggested that in its current state SPASIMv1 would be more applicable for shorter-term examination of LUCC trends. These issues of representation are considered in more detail in the next chapter.

CHAPTER SEVEN

VALIDATING SOCIO-ECOLOGICAL SIMULATION MODELS

7.1 INTRODUCTION

This chapter discusses the validation, evaluation and interpretation of environmental simulation modelling. The discussion and argument are focussed on simulation models that represent socio-ecological systems (section 1.3). Socio-Ecological Simulation Models (SESMs), as they are referred to here, are those that represent explicitly the feedbacks between the activities and decisions of individual actors and their social, economic and ecological environments. To represent such real-world behaviour, models of this type are usually spatially explicit and agent-based (e.g. Evans *et al.* 2001, Moss *et al.* 2001, Evans and Kelley 2004, An *et al.* 2005, Matthews and Selman 2006) – SPASIMv1 is an example of a SESM. One motivating question for the discussion that follows is, considering the nature of the systems and issues they are used to examine, how we should go about approaching model evaluation, 'validation', or other label by which we want to name the process of identifying the level of confidence that can be placed in the knowledge produced by the use of a SESM? A second question is, given the nature of SESMs, what approaches and tools are available and should be used to ensure models of this type provide the most useful knowledge to address contemporary environmental problems?

The discussion here adopts a (pragmatic) realist perspective (Richards 1990, Sayer 2000) and recognises and the importance of the open, historically and geographically contingent nature of socio-ecological systems. The difficulties of attempting to use general rules and theories (i.e. a model) to investigate and understand a unique place in time are addressed. As increasingly acknowledged in environmental simulation modelling (e.g. Sarewitz *et al.* 1999), the socio-ecological simulation modelling is a process in itself in which human decisions come to the fore – both because human decision-making is being modelled but also, importantly, because modellers' decisions during model construction are a vital component of the process. If these models are intended to inform policy-makers and stakeholders about potential impacts of human activity, the uncertainty inherent in them needs to be managed to ensure their effective use. Fostering trust and understanding via a model that is practically adequate for purpose may aid more orthodox scientific forms of model validation and evaluation. The conceptual discussion in the first section of this chapter paves the way for later examination of the use of stakeholder input to assess and evaluate the SESM previously presented (SPASIMv1 in chapters four – six).

7.2 VALIDATING MODELS OF 'OPEN' SYSTEMS

The hypothetico-deductive (critical rationalist) scientific method has been prevalent in the majority of natural science throughout the 20th and 21st centuries. The method states that claims to knowledge (i.e. theories or hypotheses) should be subjected to tests that are able to falsify those claims. Once a theory has been produced (based on empirical observations) a consequence of that theory is deduced (i.e. a prediction is made) and an experiment constructed to examine whether the predicted consequences are observed. Until evidence is found to disprove the theory, knowledge based upon it (i.e. laws and facts) is held as provisional. In this manner, null hypotheses are often constructed in order that they might be refuted to prove the validity of an alternative hypothesis (believed to be a more accurate theory about the state of the world).

A simulation model should be an internally logically-consistent theory of how a real-world system works. Although simulation models are recognised by environmental scientists as powerful tools (Mulligan and Wainwright 2004), the ways in which these tools should be used, the questions they should be used to examine, and the ways in which they can be 'validated' are still much debated. For example, the hypothetico-deductive method is based upon logical prediction of phenomena independent of time and place and is therefore useful for generating knowledge about logically 'closed' systems. However, the 'open' nature of many real-world, environmental systems is such that the hypothetico-deductive method is often problematic to implement in order to 'validate' the knowledge they produce. 'Validation' has frequently been used, incorrectly, to mean 'verification' and as synonymous with demonstrating the ultimate *truth* of the model (Oreskes *et al.* 1994 – emphasis added). By contrast, 'validation' in this discussion refers to the process by which a model has been shown to represent or simulate some 'real world' system or phenomena *well enough to serve that model's intended purpose*. That is, 'validation' means the establishment of legitimacy – usually of arguments and methods. As such, 'validation' should be thought of a synonymous with 'evaluation', meaning to ascertain the worth or value of a model (relevant to its intended use). It is in this 'evaluation' sense that 'validation' is used here. The nature of these 'open' systems and the problems they present for model validation are now discussed.

7.2.1 The Nature of Open Systems

Issues of validation and model assessment are largely absent in discussions of abstract models that purport to represent the fundamental underlying processes of 'real world' phenomena such as wildfire (e.g. Bak *et al.* 1987, Clar *et al.* 1996), social preferences (e.g. Solomon *et al.* 2000) and human intelligence (e.g. Wakeling and Bak 2001). These 'metaphor models' (Perry and Millington 2007) do not require empirical validation in the sense that environmental and earth systems modellers use it, as the very formulation of the system of study ensures it is closed. That is, the system the model examines is logically self-contained and uninfluenced by, nor interactive with, outside statements or phenomena. The modellers do not claim to know much about the real world system which their model is purported to represent, and do not claim their model is the best representation of it (e.g. Solomon *et al.* 2000). Rather, the modelled system is related to the empirical phenomena via 'rich analogy' (Edmonds and Hales 2003) and investigators aim to elucidate the essential system properties that emerge from the simplest model structure and starting conditions.

In contrast to these virtual, logically closed systems, empirically observed systems in the real world are 'open' (von Bertalanffy 1950). That is, they are in a state of disequilibrium (Kay *et al.* 1999) with flows of mass and energy both into and out of them (Lane 2001, Brown 2004). Examples in environmental systems are flows of water and sediment into and out of watersheds and flows of energy into (via photosynthesis) and out of (via respiration and movement) ecological systems. Furthermore, empirical systems containing humans and human activity are open not only in terms of conservation of energy and mass, but also in terms of information, meaning and value (Naveh 2001, Matthews and Selman 2006). Political, economic, social, cultural and scientific flows of information across the boundaries of the system cause changes in the meanings, values and states of the processes, patterns and entities of each of the above social structures and knowledge systems. Thus, system behaviour is open to modification by events and phenomena outside the system of study.

Alongside being ‘open’, these systems are also ‘middle-numbered’ (Kay and Schneider 1994). Middle-numbered systems differ from small-numbered systems (controlled situations with few interacting components, e.g. two billiard balls colliding) that can be described and studied well using Cartesian methods, and large-numbered systems (many, many interacting components, e.g. air molecules in a room) that can be described and studied using techniques from statistical physics. Rather, middle-numbered systems have many components, the nature of interactions between which is not homogenous and is often dictated or influenced by the condition of other variables, themselves changing (and potentially distant) in time and space. Such a situation might be termed complex (though many perspectives on complexity exist – see O'Sullivan 2004). At the landscape scale (section 1.3), observed and located in the real world, socio-ecological systems are complex and middle numbered, existing in a unique time and place. The non-ergodic nature of the (observed) universe makes individual events within them virtually irreproducible (Kauffman 2000). In these systems history and location are important and their study is necessarily a ‘historical science’ that recognises the difficulty of analysing unique events scientifically through formal, laboratory-type testing and the hypothetico-deductive method (Frodeman 1995). Most ‘real-world’ systems possess these properties, and socio-ecological systems are a prime example.

7.2.2 Epistemological Problems Presented by a Critical Realist Ontology

The recognition of the ‘open’ and middle-numbered nature of real-world systems has led to a growing acceptance of realist (Richards 1990, Richards *et al.* 1997) and relativist (Beven 2002) perspectives on the modelling of these systems in the environmental and geographical sciences. Richards (1990) initiated debate on the possibility of adopting a critical realist (CR) perspective toward research in the environmental (and geographical) sciences (specifically geomorphology) by criticising the then emphasis on rationalist (hypothetico-deductive) methods. For example, in the search for the ‘laws of nature’, a rationalist approach leaves open the possibility of the creation of laws as artefacts of the experimental (or model) ‘closure’ of inherently open systems (see below for discussion on model ‘closure’). The CR perspective states that reality exits independently of our knowledge, and that it is structured into three levels: ‘real’ natural generating mechanisms; ‘actual’ events generated by those mechanisms; and ‘empirical’ observations of actual events. Whilst mechanisms are time and space invariant, actual events are not because they are realisations of the generating mechanisms acting in particular conditions and contingent circumstances (Richards

1990). Thus, in open systems identical mechanisms will not necessarily produce identical events at different locations in space and time, and experimental (and modelling) methods may not identify the underlying mechanisms because of the closed conditions imposed (Richards *et al.* 1997).

In turn, the CR perspective does not claim to be able to find absolute truth (or attempt temporal prediction – see below). Rather, the focus is on the development of progressive theories that will lead science closer to true underlying causal mechanism. Critics of CR have suggested that such a progression is flawed simply because of the presence of periodic 'paradigm shifts' (i.e. shift in fundamental thinking and theory within a scientific discipline, Rhoads 1994) and because it leaves open the question of demarcation criteria (i.e. when do realists know they have reached the underlying causal mechanisms and hence to stop studying it? Bassett 1994). Whilst CR suggests why the ontological nature of reality might hinder the development of simulation models of open environmental systems, it does not provide a method by which to overcome the remaining epistemological problem of knowing whether a given model structure is appropriate.

One of the problems of determining the appropriateness of model structure is caused by the presence of equifinality. The necessity to delineate boundaries on open, middle-numbered systems for investigation by modelling invokes issues regarding model 'closure' – the positioning of model boundaries in space, time and process consideration. Model closure is not a problem for metaphor models of the type described above as the very formulation of the system of study ensures it is closed (i.e. it is logically self-contained). Furthermore, the inherently closed nature of these models ensures the scale of study is such that the dependence of results upon observer behaviour is minimal, if not non-existent (Baveye 2004). However, model closure and scale have been an important point of discussion for geography, ecology and the environmental sciences as the systems studied in these disciplines are inherently open and at scales on the order of the human observer (e.g. Richards 1990, Richards *et al.* 1997, Lane 2001, Beven 2002, 2004, Brown 2004, Lane *et al.* 2006). Beven (2002) suggests that the characteristics and boundary conditions of many places (systems) are likely to be 'unknowable', highlighting what all modellers intrinsically know – that their model is not a 'true' representation of reality – and emphasises the presence of model equifinality. Equifinality is the characteristic of all open systems that a final system state may be reached from multiple initial conditions and via different sequences of system state (i.e. pathways, von Bertalanffy 1950). In modelling terms, equifinality implies that there are multiple (closed) model structures that may reproduce empirically observed behaviour of an open system. Equifinality generates uncertainty in the appropriateness of model structure.

Models that consider human activity are particularly difficult to 'close' due to their consideration of 'interactive' kinds (classifications). Hacking (1999) distinguishes between 'interactive' and 'indifferent' kinds. Kinds of people are interactive kinds for example, and are defined as such because people are aware and can respond to how they are being classified. Hacking contrasts the interactive kinds of the social sciences with the indifferent kinds of the natural sciences that are not aware (and therefore cannot recognise that they have been classified). The representation of interactive kinds potentially results in a 'looping effect' (Hacking 1999) that has implications for model closure and validation – SESMs are logical and factual constructs that have the potential to feedback into, and therefore transform, the systems they represent via the conscious awareness of local stakeholders using the model or its results (or participating in the modelling process). If this transformation occurs, the factual construct (the model system) will no longer represent the empirical system accurately and will require modification. Such a situation implies that an SESM may never truly represent a socio-ecological system (if it used by those it represents) and that iterative model development and use will be required to ensure continued utility.

A third problem, and tied closely to the problems described above, regards the assessment of whether a model accurately represents the structure and function of a real-world system. Comparison of models' predictions with empirical events has frequently been used in an attempt to show that the model structure is an accurate representation of the system being modelled (i.e. demonstrate it is 'true'). These *temporal* predictions about events occurring at explicit points in time or geographical space have often been treated with the same respect given to the *logical* prediction of the hypothetico-deductive method (Oreskes 2000). However, it is unclear whether the comparison of a temporal prediction produced by a simulation model with empirical events is a test of the input data, the model structure, or the established facts upon which the structure is based (Oreskes 2000). Furthermore, if the model is refuted (i.e. temporal predictions are found to be incorrect) given the complexity of many simulation models it would be hard to pin-point which part of the model was at fault. In the case of spatial models, the achievement of partially spatially accurate prediction does little to establish where or why the model went wrong (e.g. as found in the empirical models presented in section 3.3). If the model is able to predict observed events, this is still no guarantee that the model will be able to predict into the future given the stationarity requirement (section 3.3.4). Regardless, Oreskes *et al.* (1994) have argued that temporal prediction is not possible by numerical simulation models of open, middle-numbered systems because of theoretical, empirical, and parametric uncertainties within the model structure (also see Oreskes 1998). As a consequence, Oreskes *et al.* (1994) warn that numerical simulation modellers must beware of '*affirming the consequent*' by deeming a model invalid (i.e. false) if it does not reproduce the observed data, or valid (i.e. true) if it does.

7.2.3 A Potential Relativist Response

As a result of these epistemological problems (specifically equifinality), Beven (2002) forwards a modelling philosophy that accepts a relativist perspective and uncertainty. This realist approach demands greater emphasis on pluralism, use of multiple hypotheses, and probabilistic approaches when formulating and parameterising models. When pressed to comment further on his meaning of relativism (Baveye 2004), Beven (2004 p.2150-51) highlights the problems of rigidly objective measures of model performance and of 'observer dependence' throughout the modelling process (Table 7.1);

"Claims of objectivity will often prove to be an illusion under detailed analysis and for general applications of environmental models to real problems and places. Environmental modelling is, therefore, necessarily relativist."

For example, the differences in the approach by ecologists and economists to close their models of similar environmental systems were highlighted in section 3.4.2. The operator dependencies present in the modelling presented in this thesis are discussed further in chapter nine. The importance of individual modellers' perspectives on the process of model closure and model development that Beven discusses have been highlighted in the past (Morgan 1982), but have also attracted more attention recently by geographers (Demeritt 2001, Brown 2004, O'Sullivan 2004, Lane *et al.* 2006),

Table 7.1 Sources of relativistic operator dependencies. The presence of operator dependencies in the model construction process, particularly model closure, implies both a relativist modelling perspective and the presence of equifinality. Source: Beven (2004)

Sources of Relativistic Operator Dependencies
1. Operator dependence in setting up one or more conceptual model(s) of the system, including subjective choices about system structure and how it is closed for modelling purposes; the processes and boundary conditions it is necessary to include and the ways in which they are represented.
2. Operator dependence in the choice of feasible values or prior distributions (where possible) for 'free' parameters in the process representations, noting that these should be 'effective' values that allow for any implicit scale, nonlinearity and heterogeneity effects.
3. Operator dependence in the characterization of the input data used to drive the model predictions and the uncertainties of the input data in relation to the available measurements and associated scale and heterogeneity effects.
4. Operator dependence in deciding how a model should be evaluated, including how predicted variables relate to measurements, characterization of measurement error, the choice of one or more performance measures, and the choice of an evaluation period.
5. Operator dependence in the choice of scenarios for predictions into the future.

environmental scientists (Oxley and Lemon 2003), social scientists (Agar 2003) and philosophers of science (Collins 1985, Winsberg 2001). Notably, although with reference to experimental physics rather than simulation modelling, Collins (1975, 1985) identified the problem of the 'experimenter's regress'. This problem states that a successful experiment occurs when experimental apparatus is functioning properly – but in novel experiments the proper function of the apparatus can only be checked by whether or not the experiment is successful. That is, in situations at the boundaries of established knowledge and theory, not only are hypotheses contested, but so too are the standards and methods by which those hypotheses are confirmed or refuted. As a result, experimentation becomes a 'skilful practice' and Collins (1985) suggests that experimenters accept results based not on epistemological or methodological grounds, but on a variety of social (e.g. group consensus) and expert (e.g. perceived utility) factors.

Such a stance is echoed in many respects by Winsberg's (2001) 'epistemology of simulation', which suggests simulation is a 'motley' practice and has numerous ingredients of which theoretical knowledge is only one. The approximations, idealisations and transformations used by simulation models to confront analytically intractable problems (often in the face of sparse data), need to be justified internally (within the model construction process) on the basis of existing theory, available data, empirical generalisations, and the modeller's experience of the system and other attempts made to model it (Winsberg 1999, 2001). Similarly, Brown (2004) suggests that in the natural sciences uncertainty is rarely viewed as due to the interaction of social and physical worlds (though Beven's environmental modelling philosophy outlined above does) and that modellers of physical environmental processes might learn from the social sciences where the process of gaining knowledge is understood to be important for assessing uncertainty. However, whilst an extreme rationalist perspective prevents validation and useful analysis of the utility of a model, its output, and the resulting knowledge (because of a mis-placed emphasis on objective prediction), so too does an extreme relativist stance (which states model and model builder are inseparable – Kleindorfer *et al.* 1998). Rather, as Kleindorfer *et al.* (1998 p.1098) suggest, modellers need to develop the means to increase the credibility of the model such that "meaningful dialogue on a model's warrantability" can be conducted. How and why this might be achieved is now discussed.

7.3 Post-Normal Science and Public Engagement

Concerns about the implications of the open nature of environmental systems for simulation modelling have been accompanied by, and possibly even arise from, a changing context in which these models are expected to operate. The changing context of the construction, use and assessment of SESMs is related to the view that 'traditional science' is at root of the risks and problems contemporary societies face (Beck 1992, Funtowicz and Ravetz 1993, Giddens 1999). Beck (1995) describes 'traditional science' as the *science of data* – it is specialised, laboratory-based and uses language of mathematics to explore the world. Rather than reducing risk and remedying the problems it is the root of, 'traditional science' will simply propagate them further (Beck 1992). The acknowledgement of this situation has led some to question how science should proceed in order to face up to and deal with its consequences (e.g. Funtowicz and Ravetz 1993, Allen *et al.* 2001). The consequences of science are no longer simply further academic and scientific problems, but are important socially, politically, culturally, and environmentally. Indeed, they may be imminent, potentially necessitating action before the often lengthy traditional scientific method (hypothesis testing, academic peer review etc.) has produced a consensus on the state of knowledge about it.

Thus, just as realist and relativist perspectives have been suggested recently as more useful for examining open, middle-numbered systems from a philosophical stance, Funtowicz and Ravetz (1993) have presented ‘post-normal’ science as a response to the problems of the ‘risk society’. Post-normal science is a new type of science to replace the reductionist, analytic worldview of ‘normal’ science with a systemic, synthetic and humanistic approach. Such a science, is similar to Beck’s (1995) *science of experience*, which identifies consequences and threats, and publicly tests its objectives and standards to examine the doubts the traditional *science of data* ignores. The term ‘post-normal’ deliberately echoes Kuhn’s (1962) formulation of ‘normal’ science functioning between paradigm shifts, to emphasise the need for a shift in scientific thinking and practices that takes it outside of the standard objective, value-free perspective. The implication is that the ‘puzzle-solving’ that ‘normal’ science does will remain important but that it will not be enough for the ‘problem-solving’ it has produced the need for.

Therefore post-normal science, according to Funtowicz and Ravetz (1993), embraces the uncertainties inherent in issues of risk and the environment, makes values explicit rather than presupposing them, and generates knowledge and understanding through an interactive dialogue rather than formalised deduction. This dialogue extends to encompass new forms of ‘citizen science’ (Irwin 1995) and also ‘extended peer communities’ (Healy 1999, Ravetz 2004) which engage with actors influenced by, or at risk as a consequence of, the environmental issues in hand. As discussed below (section 7.3.2), this engagement with the public will move ‘upstream’ to earlier stages of the research process. The methodology of post-normal science emphasises uncertainties in knowledge, quality of method, and complexities in ethics, all of which are relevant in varying degrees to the issues facing SESMs. The epistemological problems discussed in section 7.2 are clear indications of the importance of considering uncertainties in model construction. The discussion that now follows examines the potential of evaluative, qualitative approaches to model validation as an alternative to the more ‘traditionally scientific’ deductive approach. Ethical questions of expertise and public engagement (i.e. who should be asked?; who is qualified to ask?; is a model acceptable for use?) are also examined below (section 7.3.2).

7.3.1 Evaluation Criteria for Simulation Modelling in the 'Risk Society'

Simulation models are tangible manifestations of a modellers' 'mental model' of the structure of the system being examined (Forrester 1993). In the case of metaphor models, the correspondence between the model system and the empirical system is via a 'rich analogical' mental model (Edmonds and Hales 2003). In environmental simulation models, including SESMs, the mental model makes a more direct and, hopefully, more accurate correspondence with the empirical system. In turn, the model system (hopefully) provides a more direct and accurate representation of the empirical system than in a metaphor model. However, the problems of equifinality suggest that there are multiple logical model structures that could be implemented to represent any particular open system. That is, establishing what is the truly accurate, correct, representation of an empirical system is very difficult (if not impossible according to the critical realist ontology) for SESMs and other models of open, middle-numbered systems. Furthermore, whilst accurate mimetic reproduction of an empirical system state by a model may be the most persuasive argument that a model structure is an accurate representation of reality in many eyes, the dangers of affirming the consequent make it impossible to prove this or that any temporal predictions produced by models of open systems are truly accurate (Oreskes 2000). Affirming the consequent means mimetic accuracy cannot be used to confirm structural accuracy (i.e. that the model structure matches that of the empirical system). Thus, simulation models may be based on facts about empirical systems, but accurate mimicry of empirical system states does not prove their structural accuracy and their results cannot be taken as facts about the empirical system they represent.

The inability to prove what the correct model structure is does not allow modellers to dispense the requirement that their model be logically coherent and based on the best available knowledge (i.e. facts). Equally, the logical fallacy of affirming the consequent does not mean that models should not aim to be mimetically accurate or realistic. However, some other criteria – alongside not being factually or logically invalid and producing mimetically 'realistic' results – will be useful to evaluate or validate SESMs. A third and fourth criteria, for SESMs at least, are available by specifically considering the user(s) of a model and its output. These criteria are closely linked. The third criterion is the establishment of user trust in the model. Trust is used here in the sense of 'confidence in the model'. If a person using a model or model results does not trust the model it will likely not be deemed fit for its intended purpose. The definition of validation being used here – that a model must represent an empirical system *well enough for that model's intended purpose* (section 7.2) – makes this point important. If confidence is lacking in the model or its results, confidence will consequently be lacking in any knowledge derived, decision made, or policy recommended based upon the model. Thus, the use of trust as a criterion for validation is a form of 'social validation', ensuring that user(s) agree the model is a legitimate representation of the system (Castella *et al.* 2005b). The fourth criteria by which a model might achieve legitimacy and receive a favourable evaluation (i.e. be validated), is the provision of some form of utility to the user. This utility will be termed 'practical adequacy'. If a model is not trusted then it will not be practically adequate for its purpose. However, regardless of trust, if the model is not able to address the problems or questions set by the user then the model is equally practically inadequate.

The addition of these two criteria, centred on the model user rather than the model, suggests a shift away from falsification and deduction as model validation techniques, toward more reflexive approaches. The shift in emphasis is away from establishing the truth of the model's structure via mimetic accuracy and toward ensuring trust in the model's results via practical adequacy. By considering trust and practical adequacy, validation becomes an exercise in model evaluation and reclaims its correct meaning 'to establish a model's legitimacy'. From his observation of experimental physicists and work on the experimenter's regress (section 7.2), Collins (1985) has arrived at the view that there is no distinction between epistemological criteria and social forces to resolve a scientific dispute. The position outlined above seems to imply a similar situation for models of open, middle-numbered systems, requiring modellers to resort to social criteria to justify their models due the inability to do so convincingly epistemologically. This is not necessarily an idea that many natural scientists will sit comfortably with. However, the shift away from finding truth via mimetic accuracy should not necessarily be something scientific modellers would object to. First, all modellers know that their models are not true, exact replications of reality. A model is an approximation of reality – there is no need to create a model system if experimentation on the existing empirical system is possible. Furthermore, accepting the results of a model are not facts, in the sense that they are accurate predictions, in no way requires the model be built on incorrect logic or facts. As Hesse (1986 p.721) notes in criticism of Collins (1985), whilst resolution of scientific disputes might result from a social decision that is not *forced* by the facts "it does not follow that social decision has nothing to do with objective fact". Second, regardless of finding truly accurate model structures or mimicry of empirical system states, ensuring models are logically coherent and not factually invalid will already have come some way to make a scientific case. Thus, regardless of the failures of epistemological methods for justifying the structure of SESMs and other environmental simulation models, they must be built upon solid logical and factual foundations;

"The postmodern world may be a nightmare for … normal science (Kuhn 1962), but science still deserves to be privileged, because it is still the best game in town. … [Scientists] need to continue to be meticulous and quantitative. But more than this, we need scientific models that can inform policy and action at the larger scales that matter. Simple questions with one right answer cannot deliver on that front. The myth of science approaching singular truth is no longer tenable, if science is to be useful in the coming age." (Allen *et al.* 2001 p.484).

Post-normal science highlights the importance of finding alternative ways for science to engage with both the problems faced in the contemporary world and the people living in that world. As they have been defined here, SESMs will inherently address questions that will be of concern to more than just scientists, including problems of the 'risk society'. From a modelling perspective, a post-normal science approach highlights the need to build trust in the eyes of non-scientists such that understanding is fostered. Further, it emphasises the need for SESMs to be practically adequate such that good decisions can be made promptly. It also implies that the manner in which a 'normal' scientist will go about assessing the trustworthiness or practical adequacy of a model (such as the methods described above) will differ markedly from that of a non-scientist. For example, scientific model users will often, but not always, have also been the people who developed and constructed the model. In such a case the model will be constructed to ensure the model is practically adequate to address their particular scientific problems and questions. When the model is to be used by other parties the issue of ensuring practical adequacy will not be so straight-forward, and particularly so when the user is a non-scientist. In such situations, the modeller needs to ask the question 'practically adequate *for what*'? The inhabitants of the study areas investigated will have a vested interest in the processes being examined and will themselves have questions that could be addressed by the model. In all probability many of these questions will be ones that the modeller themselves has not considered or, if they have, may not have considered relevant. Further, the questions asked by local stakeholders may be non-scientific – or at least may be questions that environmental scientists are not used to attempting to answer.

It is of no doubt that continual investigation into pixel-by-pixel model assessment approaches (e.g. section 3.3.3), in an attempt to derive the most information out of spatial error matrices, is necessary to improve further technical methods to monitor and model LUCC. Here however, the emphasis is on what alternative methods for model validation (assessment) might be useful to utilise the additional information and knowledge which is available from those actors involved and driving the observed LUCC. In other words, there is information within the system of study that is not utilised for model assessment by simply comparing observed and predicted model maps. This information is present in the form of local stakeholders' knowledge and experience, of a nature that has been termed 'Traditional Ecological Knowledge' (TEK, Berkes *et al.* 2000). The broad theory of TEK has arisen recently in the Human Ecology and Conservation Ecology literature. TEK is local observational knowledge of environmental systems, resource activities and beliefs about human-environment interactions (Berkes *et al.* 2000). Many studies have examined the use of TEK in environmental management and conservation (e.g. Moller *et al.* 2004, Parlee and Berkes 2006) but the direct and explicit consideration of such information for use in the evaluation of environmental modelling has been limited (the exceptions were outlined in the introduction of this chapter, e.g. Castella *et al.* 2005b).

7.3.2 Stakeholder Participation and Expertise

The inability to establish the single 'correct' model structure and the dangers of elevating mimetic accuracy to the status of logical proof suggest alternative criteria by which to validate or evaluate SESMs will be useful. Above it has been suggested that trust and practical adequacy might be useful additional criteria. In light of the 'risk society'-type problems facing the systems that SESMs represent, and the proposed post-normal science approaches to examine and resolve them, the participation of local 'stakeholders' within the model validation process seems an important and useful approach to ensure and improve model quality. Local stakeholders are taken here to mean individuals or representatives with a vested interest in the study area. If local stakeholders are to accept decisions and policies based upon results from, or knowledge generated by, simulation models they will need to trust a model and, by consequence, the modeller(s).

Due to a perceived ‘crisis of trust’ in science over the last 20 years, Wilsdon and Willis (2004 p.16) suggest “scientists have been slowly inching their way towards involving the public in their work” and that we are now on the cusp of a new phase of public engagement that takes it ‘upstream’. This widely used, but somewhat vague term, is used to refer to the early involvement of the lay public in the processes of scientific investigation. As such, engagement is ‘upstream’ nearer the point at which the research and development agenda is set, as opposed to the ‘downstream’ end at which research results are applied and the consequences examined (e.g. Jackson *et al.* 2005, Figure 7.1). Whereas previously the theory of the ‘public understanding of science’ was a deficit model suggesting that the public would trust science ‘if only they understood it’, the contemporary shift is towards an engagement and dialogue between science and society. The implication of this new turn is that the public will trust science ‘if only they are involved in the process itself’. Recently, this move upstream has been advocated for forms of environmental modelling that address issues and concerns of rural populations (Lane *et al.* 2006). However, such a position has been criticised as devaluing the worth of science, for patronising the public, and being a mask for political face-saving or insurance (Durodie 2003). Regardless of other areas of science, in the case of developing simulation models for socio-ecological systems the participation of the public certainly does not result in the first two of these criticisms. Engaging with local stakeholders to ensure a model is both built on a logically and factually coherent foundation and to ensure it examines the appropriate questions and scenarios (a component of practically adequacy) is of great value to the modelling process and should improve representation of the empirical system. Contributing to successful iterations of this process, local stakeholders will gain both trust and understanding.

Stirling (2006) suggests there are three classes of motivation for a scientist to enter into a process of stakeholder engagement. These are *normative democratic* (‘I think talking to you is the right thing to do’), *instrumental* (particular ends; ‘talking to you gives me credibility and fosters trust – regardless of whether I take account of what you say’) and *substantive* (better ends; ‘I am talking to you because there might be things I haven't considered that you know or are concerned about, therefore improving my work’). The

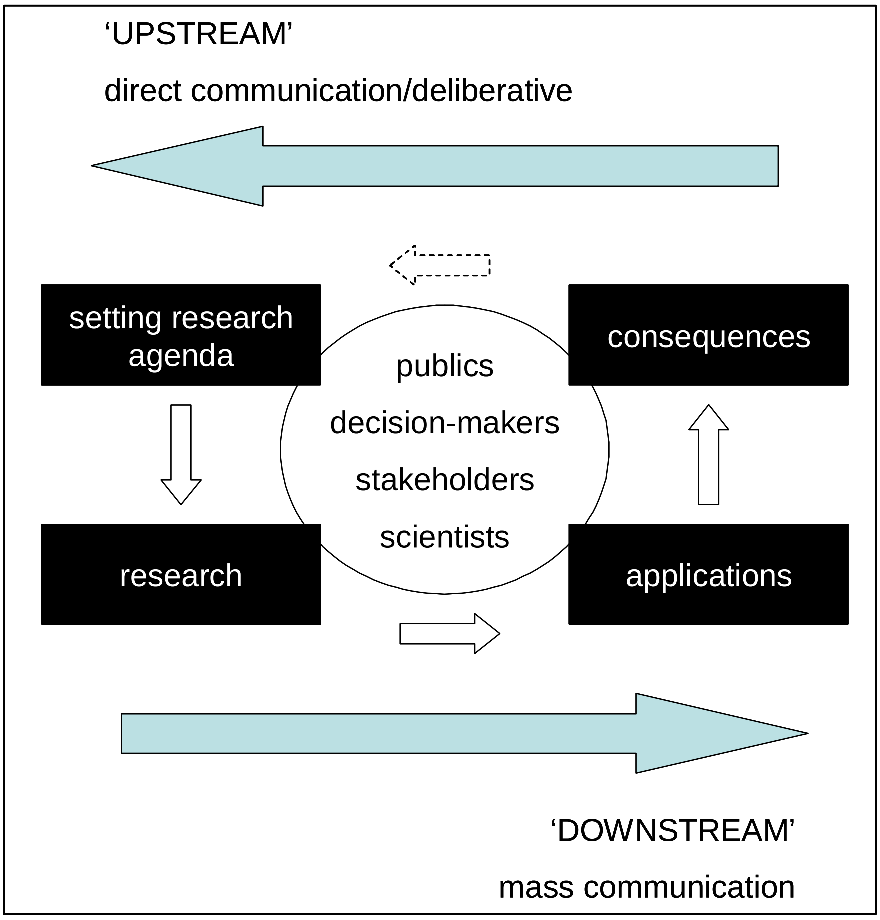

Figure 7.1 Public participation in the scientific research process. Recently it has been suggested that public engagement with the scientific process needs to move ‘upstream’ nearer the point at which the research agenda is set. After Jackson *et al.* (2005).

rationale laid out in previous sections should make it clear that the motivation for engagement proposed here is ‘substantive’. Some modellers may also fall into the normative democratic class, and whilst this should be no problem if their primary motivation is substantive, it is debatable (but won’t be here) as to whether a normative democratic motivation is justified alone. However, stakeholder participation for model validation and evaluation, as proposed here, should not be undertaken if solely instrumentally motivated. First, if stakeholder evaluation is successful under a substantive motivation, credibility and trust will be fostered regardless. But second, the instrumental motivation leaves the door open for criticisms (such as Durodie 2003) that the exercise is merely political face-saving or insurance with no sound methodological rationale.

Indeed, the rationale for the substantive motivation has roots in both ensuring practical adequacy and recent theory regarding experts and expertise. With parallels in the three phases Wilsdon and Willis (2004) have suggested, Collins and Evans' (2002) suggested we are have entered a third wave in the sociology of science. This third wave demands a shift from an emphasis on technical decision-making and truth to expertise and experience. Collins and Evans (2002) suggest three types of expert in technical decision-making (i.e. decision-making at the intersection of science and politics); 'No Expertise', 'Interactional Expertise', and 'Contributory Expertise'. Individuals possessing interactional expertise are able to interact 'interestingly' with individuals undertaking the science, but not to contribute to the activities of science itself (contributory expertise). 'Non-certified' experts are recognised as possessing a form of contributory expertise, for example a farmers' expertise regarding the functioning of agricultural landscapes. Such non-certified expertise might also be termed 'experience-based' expertise, arising as it does from the day-to-day experiences of particular individuals. The well-known of example of the (inadequate) interaction between Cumbrian sheep farmers and UK government scientists investigating the ecological impacts of the Chernobyl disaster (Wynne 1989) is a prime case-study of a situation in which two parties possessed contributory expertise, but neither interactional expertise. As a result, the certified expertise of the government scientists was given vastly more weight than the non-certified expertise of the farmers (to the detriment of the accuracy of knowledge produced).

The importance of considering non-certified, contributory experience is particularly acute for SESMs. Specifically, local stakeholders are likely to be an important, if not the primary, source of knowledge and understanding regarding socio-economic processes and decision-making within the study area. Furthermore, the particular nature of the interactions between human activity and ecological (and other biophysical) processes within the study area will be best understood and incorporated into the simulation model via engagement with stakeholders. This local knowledge will be vital to ensure the logical and factual foundations of the model are as sound as possible. But further, engagement with local stakeholders will highlight model omissions, areas for improved representation, and guide application of the model. It provides an opportunity to enlighten experts as to the 'blind spots' in their knowledge and questions (Stilgoe *et al.* 2006). As such, the local stakeholders become an 'extended peer community', lending alternative forms of knowledge and expertise to the model (and research) validation process than that of the scientific peer community (Ravetz 2004). This knowledge and expertise may be less technical and objective than that of the scientific community, but this nature does not necessarily reduce its relevance or utility to the modelling of a system that contains human values and subjects.

There is a case for stakeholder participation in the modelling project presented in this thesis. The individual farmers, represented in the ABM/LUCC as land-use decision-making agents, are the best source of knowledge *about their own activities.* For this reason, early in the development of the ABM/LUCC local stakeholders were interviewed with regards to how they made decisions and their understanding about landscape dynamics (see section 5.4.2). Furthermore, upon completion of model construction these stakeholders offer the prime source of criticism about the model representation of these decision-making activities. By engaging with these stakeholders a form of qualitative, reflexive model validation can be undertaken that overcomes some of the problems of a more deductive approach.

7.4 SUMMARY

Studying open systems, as socio-ecological simulation models (SESMs) are designed to do, requires boundaries to be specified and placed on the system such that it may analysed effectively. Recent debate in the geographical and environmental modelling communities has highlighted the importance of observer dependencies when identifying the appropriate model 'closure'. Furthermore, because an 'open' system can be 'closed' for study in multiple ways whilst still adequately representing system behaviour, the issue of model equifinality is present when attempting to model these systems. In the context of recent calls for science to become more relevant and useful for aiding contemporary pressing environmental problems, a discussion regarding the implications of these issues has been presented. It has been suggested that a more reflexive approach, emphasising trust via practical adequacy over the establishment of true model structure via mimetic accuracy, will put SESMs in a better position to provide understanding for non-modellers and contribute more readily to the decisions and debates regarding contemporary problems facing many real world environmental systems. This is not to say issues regarding mimetic accuracy and model structure should be totally ignored – these model validation criteria will still have a role to play. However, emphasising trust via practical adequacy over truth via mimetic accuracy, ensures the model validation question is ‘how good it this model for my purposes?’ and not ‘is this model true?’. Engagement with local stakeholders throughout the modelling processes, contributing to model development and application should ensure practical adequacy, but also, in parallel, trust. As a result of this participatory model evaluation exercise, confidence in the model should be built, hopefully to the level where it can be deemed to be ‘validated’ (i.e. fit for purpose). The next chapter presents the results from one such attempt to go through this model evaluation process for SPASIMv1. The focus is on issues directly relating to model quality – how the model can be assessed by stakeholders, how models of socio-ecological systems can be improved via input from stakeholders, and how trust in SESMs and credibility in their results is established. The mimetic accuracy and appropriateness of the model structure developed are assessed as an integral part of this evaluation. The results of interviews with local stakeholders in SPA 56, performed with these objectives in mind, are presented and discussed with reference to the issues of ‘upstream engagement’ and ‘expertise’ presented in this section.

CHAPTER EIGHT

STAKEHOLDER MODEL EVALUATION

8.1 INTRODUCTION

The previous chapter suggested that an emphasis on trust via practical adequacy over truth via mimetic accuracy will put socio-ecological simulation models (SESMs) in a better position to provide understanding for non-modellers and contribute more readily to decisions and debates regarding many real world environmental systems. The four criteria for model evaluation discussed (model structure, mimetic accuracy, trust and practical adequacy) can be evaluated by engaging with local stakeholders with knowledge of the study area the model represents. After presenting and discussing the results of such a stakeholder engagement exercise in SPA 56, this chapter evaluates the performance and worth of SPASIMv1 against these criteria. Recent work in the sociology of science has suggested a greater appreciation of the nature of expertise in technical decision-making is needed, and there have been suggestions that the public engagement of science needs to move further 'upstream' toward the research agenda-setting stage (i.e. the initiation of the research process). This chapter also examines the practical considerations of engaging with local decision- and policy-makers to test the utility and credibility of SPASIMv1.

8.2 STAKEHOLDER ENGAGEMENT IN THE MODELLING PROCESS

Ensuring the modeller's mental model (Forrester 1993, section 7.3.1) appropriately matches that of system being modelled requires an integrative and interactive research method (Lemon 1999). In other words (as highlighted in section 5.3.3.3), contact with local stakeholders, especially if iterative in nature, is likely to result in a model structure that more closely resembles the decision-making strategies of the actors within the system. Such a participatory approach has been suggested as particularly useful for agent-based modelling exercises (Parker *et al.* 2003). Participation may vary from iterative involvement at all stages of model development, through involvement in model testing (and refinement) once constructed, to the use of a model as a software package once completed (Parker *et al.* 2003). Iterative contact with local stakeholders throughout the modelling process will develop user trust and ensure practical adequacy by improving understanding (of the model structure and the process of construction) and by developing a sense of ownership. This iterative process, from establishing model agenda (purpose and scenario generation), through model construction (conceptualisation) to use (testing and analysis) both ensures the model is validated (in the sense presented in the previous chapter) and moves research upstream (Figure 8.1).

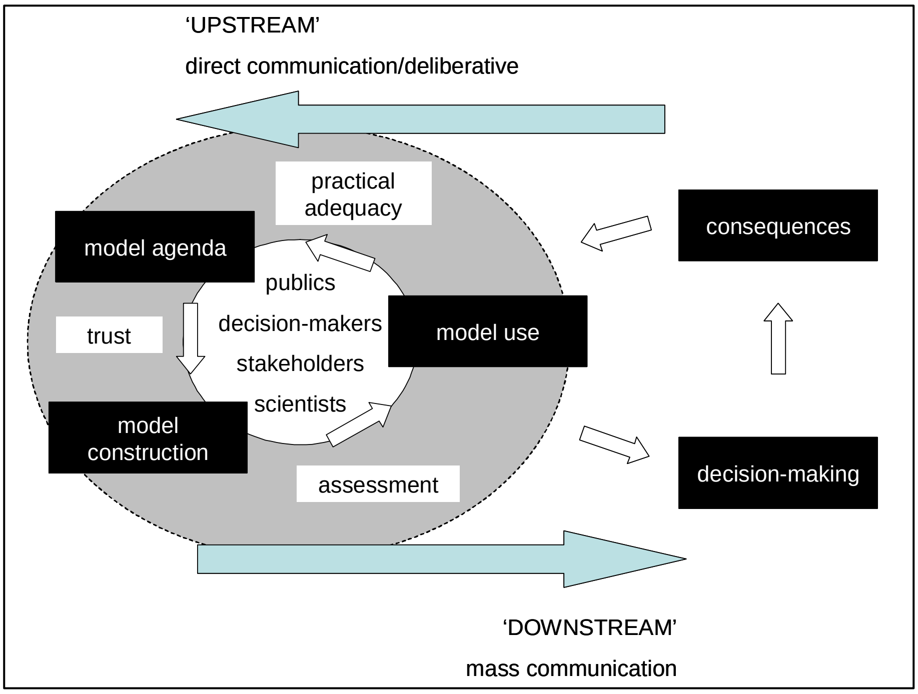

Figure 8.1 Public participation in the socio-ecological modelling process. The grey circle represents the modelling process as a whole. Engaging with stakeholders throughout this process moves it 'upstream'. This should improve model trustworthiness in the eyes of the stakeholder and ensure practical adequacy.

This iterative modelling process, encouraging stakeholder engagement throughout, differs from the view of the scientific research process envisaged by Jackson *et al.* (2005, Figure 7.1). The modelling process is presented here (Figure 8.1) as a single coherent component of the scientific research process that is difficult to split into explicit agenda-setting/research/application components. Previous investigations into the relationships between modellers (expert) and non-modellers (the lay public, non-experts) have considered this modelling process to be a 'black box' (*sensu* section 3.2) into which non-modellers cannot see. That is, non-modellers have not been involved in the construction process of the model and have been asked to assess (i.e. accept or reject) the model based on a presentation of the completed model. For example, an air-pollution model designed to provide real-time air-quality information for the city of

Sheffield, UK, has been used to examine issues regarding the public understanding of scientific models (Yearley 1999) and issues regarding expert knowledge in the context of post-normal science (Yearley 2000). Yearley (1999) found that the members of the lay public whom he interviewed assessed the model in light of local political agendas, social assumptions underpinning model structure, and technical knowledge about air-pollution in their city that they were confident they possessed. The air pollution model was presented, and issues of trust examined, at the point between 'applications' and 'consequences' in the formulation of the scientific research process {Jackson, 2005 #1567, Firgure 7.1}. Participants were sceptical of the validity of the model (i.e. its logical and factual basis, its trustworthiness and its practical utility) because the pertinent knowledge that they believed they possessed had not been incorporated into any of the 'research agenda-setting', 'research' or 'applications' components. With specific regard to environmental or LUCC modelling, there has been little, if any, work examining issues of expertise and trust situated within or between these components.

Similar issues to those encountered by Yearley (1999, 2000) have been encountered in other studies. Cockerill *et al.* (2004) surveyed a large group of local stakeholders about their perceptions of a hydrological model used for water planning in New Mexico. Again, local participants were confident they had relevant technical knowledge that allowed them to understand the model, and findings highlighted that trust in the model developers was important for acceptance of results. Similarly, Olsson and Berg (2005) found that the confidence and trust in model data and application were important factors in the acceptance of data generated by a water-quality model of a Swedish river catchment. At a general level the data served as a focal point to unite stakeholders, giving them an insight into one another's perspectives. However, concerns over model structure became apparent when examined in more detail. These case-studies suggest that greater trust will be fostered by including local stakeholders' 'non-certified expertise' and knowledge – which they believe to be pertinent – throughout the iterative modelling process. Furthermore, these case-studies also indicate that regardless of a modeller's perspective about the objectivity of their model, as soon as any modeller or model engages with non-modellers, issues of trust and practical utility arise. Rather than reject such issues as non-scientific or irrelevant, modellers must address them explicitly if SESMs are to be constructed so that they are relevant to contemporary problems and accepted as such. However, as the role-play games approach to agent-based modelling highlights (e.g. Barreteau *et al.* 2001, Castella *et al.* 2005b – section

5.3) participatory approaches can be highly resource-intensive. Thus, Matthews and Selway (2006 p.208) warn that, “participatory modelling should not be seen as a panacea” and they suggest that drawbacks include increased resources demands, potential inclusion of bias in model structure, and reduced academic credibility (also see Panebianco and Pahl-Wostl 2004).

The application of these participatory approaches to modelling LUCC has been very recent, as the examples in section 5.3 demonstrate, and is still much in the exploratory and developmental stage. The remainder of this chapter presents an exercise in ‘stakeholder model assessment’ from interviews with actors from within the study area which is a less resource-intensive approach than the RPG. Specifically, two questions guiding the rationale for these interviews were:

a) from a technical/modelling standpoint, how can we utilise local actors’ knowledge and understandings of LUCC to improve understanding of the performance of our simulation models of it and improve them?
b) if ‘normal’ science does not generate pertinent knowledge or create consensus to deal with pressing environmental problems efficiently (section 7.3), what is the role of the public and public engagement in model building and evaluation?

8.3 Stakeholder Assessment of SPASIMv1

8.3.1 Introduction

Seven interviews were undertaken in November 2006 with local stakeholders from within SPA 56, each of whom had knowledge of specific regions of the study area due to their occupation and, in many cases, place of residence. Interviewees were selected from a range of institutional contexts from private, individual land owners with no governmental connections, through to the head of one of the subsections of the Autonomous Community of Madrid’s department of environment (Consejería de Medio Ambiente y Ordenación del Territorio). Other interviewees included local agricultural co-operative officials, municipality-level planning officials, and local council (ayuntamiento) officials. Three of the seven interviewees had previously been interviewed during development of the agent-based model of agricultural land-use decision-making (the ABM/LUCC presented in chapter five). This ensured some continuity throughout the process, but also allowed the responses of individuals that had not previously encountered the model to be examined. As the majority of interviewees had knowledge and understanding about only their local area (i.e. their municipality and those neighbouring it) sections of interviews that addressed model results focused on output for their particular area. Specifically, the municipalities of Villa del Prado and San Martin de Valdeigleisias and their environs were examined (Figure 2.5). The semi-structured interviews contained five distinct sections, each motivated by an explicit research question. These interviews are now outlined along with materials presented to interviewees as discussion aids. Whilst these questions and materials were used to guide the research, interviewees were generally free to respond to and discuss whatever points they deemed relevant to the questions asked. The main focus of this model evaluation process was on the agricultural decision-making process (i.e. the ABM/LUCC component of SPASIMv1) as this is the area in which interviewees had more relevant understanding and knowledge (compared with the LFSM). Interviews were conducted in Spanish, with a Spanish collaborator (Dr. Raúl Romero Calcerrada, Universidad Rey Juan Carlos) directing discussion and interpreting language when necessary (translations of quotes used are presented in Appendix III). Interviews were recorded and notes taken throughout. Later, recordings were reviewed and the pertinent passages transcribed and translated into English. Following completion of the interviews, notes and recordings were used to interpret results presented below.

8.3.2 Drivers of Change

Research Question: What does the local stakeholder believe to be the important drivers of change in the study area, and what spatial LUCC will they produce?

Interviewees were asked how they think LUCC in their local area of SPA 56 will proceed over the next 20 years from their own perspective and understanding of the current trajectories of change. Outline maps of their municipality were presented to allow the interviewee to sketch their projected LUCC for the year 2026. Maps of initial land-cover used in the model (i.e. land-cover in 1999) were also presented as a starting point (e.g. Figure 2.10). Interviewees were then introduced to the five model scenarios for which they were to be shown model results (Table 6.1) and asked if they could envisage what LUCC would occur given the scenarios of change.

These questions were asked to establish interviewees' understanding of what LUCC will most likely be over the next 20 years to 2026, from their own 'mental model' *before* contact or mediation with the simulation model. Establishing a definite answer to this question prior to presentation of the simulation model allowed contrasts to be made with interviewees' opinions after presentation. This approach allowed an examination of how interviewees' understanding or mental model of LUCC had changed due to contact with the model (i.e. how has their mental model changed?) and to contrast what the simulation model suggested were important drivers of change with what they believe to be important (i.e. how does their mental model differ from the simulation model?). This approach also allows a comparison of the model's representation of LUCC with that expected by the stakeholder (i.e. mimetic accuracy of model to represent expected LUCC).

Interviewees stressed the importance of economic conditions on land-use decision-making. They emphasised that if agriculture is the sole income for farmers they will take whatever land-use measures required to ensure profitability. For instance:

"Where production has the organic stamp; yes, agricultural activity is profitable. There are a few organic farmers here who are competitive businessmen and live well. But traditional [forms of] agriculture are disappearing because of a lack of profitability and low product prices. … If you have a livestock exploitation producing cheese, you need to have commercial links with supermarkets to be profitable. Or, you need to associate your agricultural activity with other commercial activities on your land. Then you can be profitable and all the family is involved. If not, livestock exploitations are abandoned."

Local Development Agency Manager

This quote is indicative of the implications of the changes in agricultural markets (e.g. purchasing power of large supermarkets) and the wider Spanish economy (requiring diversification to enter into emerging markets) interviewees discussed elsewhere (see below). Linked to this were comments emphasising the importance of agricultural subsidies via the EU Common Agricultural Policy:

"I don't believe agricultural activities in San Martín are [economically] sustainable. Vineyards are in decline – either abandoned or those that are not often remain unharvested – and, in the future, the removal of subsidies is going to make the situation worse. I'd say more than half [of vineyards] will disappear."

Local Manager for Regional Government Development Council

Generally however, interviewees discussed the wider impacts of changes in lifestyle as the national economy grows and diversifies away from the primary agricultural sector. If one spends time driving around the current landscape of the city of Madrid, the number of cranes and rate of apartment and road building is noticeable (also see Economist 2006a). The Spanish economy is growing rapidly (GDP per capita rose from US$ 20,200 in 2001 to US$ 22,690 in 2003 and continues to rise, Economist 2006b) and interviewees implied that this growth was a major underlying driver of change in the study area. Such national economic changes directly influence the profitability of agriculture and thus land-use decision-making, but perhaps even more important is their effect on social attitudes. Several interviewees spoke about the trend of younger people within the study area seeking employment in construction and the service sector to secure what they perceive to be a more 'modern' and preferable lifestyle:

"…most farmers are part-time, maintaining the traditional agriculture. The children or grandchildren of those [farmers] do not have interest [in agriculture] because is it not profitable and requires a lot of dedication. The youths go or they seek other work."

Local Development Agency Manager

This more 'modern' lifestyle is one that affords leisure time at specified times of the week and at regular intervals (i.e. the weekends and paid holidays):

"Building is the type of work that interests people here. Why don't people want to work in the greenhouses? Because the bricklayers and builders have the weekends free. In the greenhouses you work many hours, and weekends, and in many cases you must take products to market at night."

Local Council Development Officer

Agriculture labour requirements vary by season and 'commercial' farming (i.e. agriculture represented by commercial agents in the agent-based model) in the study area can demand labour six or seven days a week. Part-time farming combined with other employment results in a similar situation. Securing full-time employment in manufacturing, construction or services results in labour conditions that do not vary as much by season and guarantee a five (or possibly five and half) day week. Economic development in recent decades has increased living standards and, despite increasing living costs, these standards are now such that members of the younger generation are prepared to take jobs providing lower income than ultimately possible in the agricultural sector to ensure a 'modern' lifestyle:

"Now, two of 24 farmers want to follow the footsteps [occupation] of their parents. As soon as the father retires the children go into construction or others leave agricultural production and move into [agricultural] product distribution. It [farming] is very hard work."

Local Agricultural Co-operative Officer

Comments like these provide anecdotal evidence to support previous suggestions that easier ways of making a living are on of the underlying causes of abandonment in the Mediterranean Basin (e.g. Grove and Rackham 2001, and see section 2.3.4). Many interviewees adopted a tone that suggested they believed this trajectory of change was largely inevitable and is how the majority of people want to live in the future. This feeling of inevitability also had consequences for the interviewees' projections of change. Whilst all shared the same general understanding of the underlying trajectory of social and economic change, this common understanding did not manifest itself in identical projections of spatial LUCC (e.g. Figures 8.2a and 8.2b). However it did preclude the willingness or ability of interviewees to suggest how the different scenarios (Table 6.1) would play out in terms of spatial LUCC. Rather, interviewees were more at ease projecting the spatial LUCC they expected from their own understanding of the drivers (i.e. their own scenario of drivers). Sketch maps were broadly drawn (e.g. Figure 8.2). Some individuals had problems envisaging how change would occur *spatially* and needed encouragement. Those that had planning experience were best suited to this exercise, likely because their work demands they deal with maps and think in spatial terms. The difficulties of some interviewees to project change spatially suggest implications for later model assessment. If an interviewee does not personally understand how change occurs in a spatial manner, it is unlikely they will be the best evaluator of spatial model output (see section 8.4.1 below).

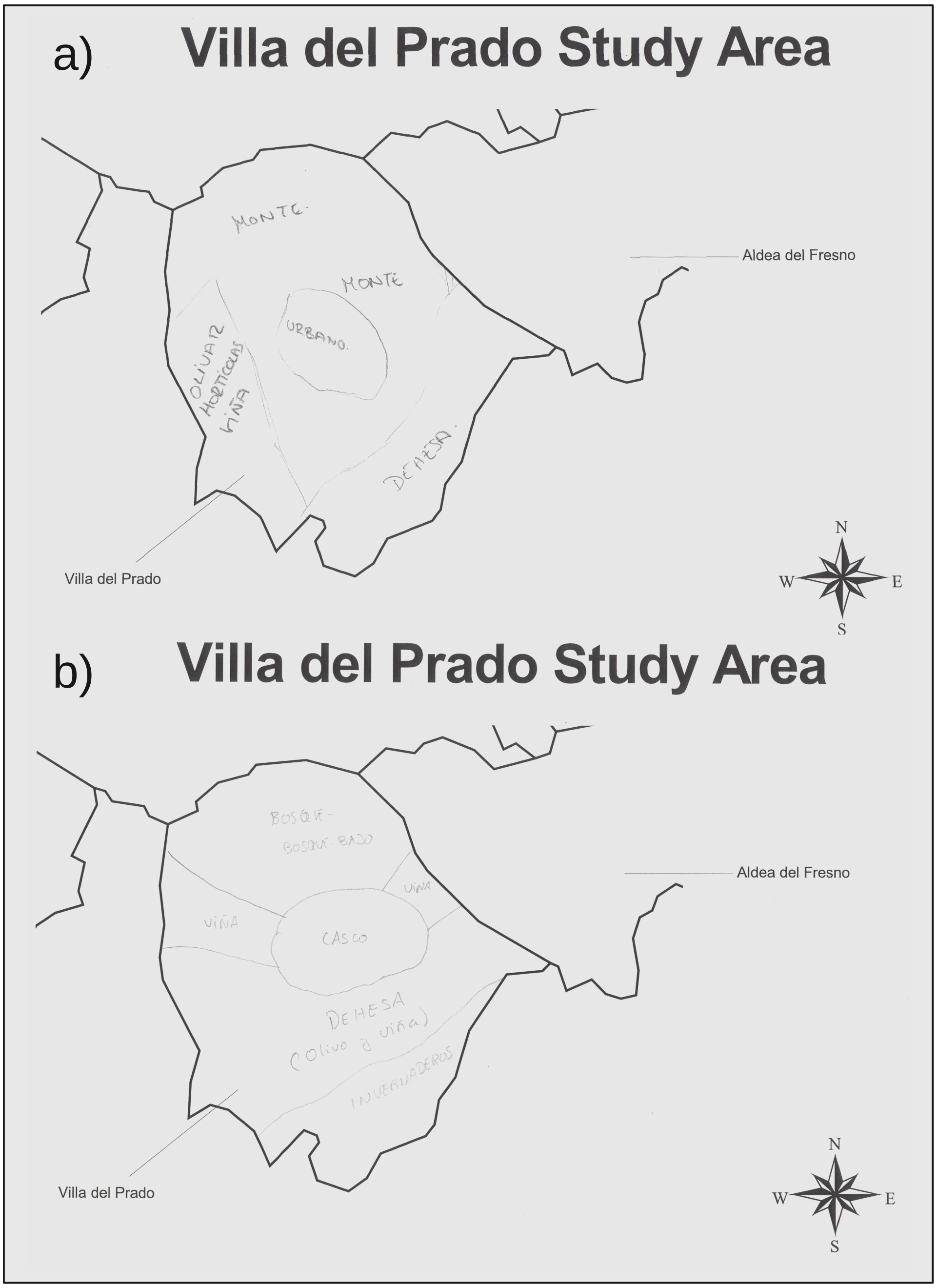

Figure 8.2 Sketch maps of two interviewees' expectations of LUCC in Villa del Prado by 2026. This map provides examples of those made by other interviewees. Sketch maps were drawn roughly at a broad spatial scale, and corresponded well with the results of at least one model scenario (at this level of analysis).

8.3.3 Model Results

Research Question: How plausible/feasible are the projected simulation model maps and movies given the scenarios they were generated for?

Interviewees were then presented with the results of the simulation model for each of the scenarios specified in Table 6.1. These results took the form of maps (presented in Microsoft PowerPoint, e.g. Figure 8.2) and movies of LUCC (presented in Microsoft

Media Player – e.g. see Appendix IV) over the period 1999 – 2026. Interviewees were also asked to comment on how plausible or feasible they thought the results were, given the scenarios they were run for. Yet another question interviewees were asked to consider (if they did not make the comparison themselves) was how the simulation model projections matched those they had suggested at the outset of the interview (section 8.2.2).

This section of the interview was designed to gauge interviewees' opinions of the simulation model and its results before they knew anything about how its structure or how/why it produced those results. Thus, again, this is a test more of mimetic accuracy than structural accuracy. Throughout this part of the interview the research team (i.e. the modeller and collaborator) were careful to observe the interviewees, and from their tone and body language assessed whether stakeholders were impressed by the maps and movies, or whether they were sceptical. This observation was with the aim of assessing whether the interviewee 'accepted' or 'trusted' model results, or whether they were sceptical. How the interviewees' attitude toward the model changed as they gained a better understanding of the model structure and assumptions (section 8.2.4 below) was also assessed.

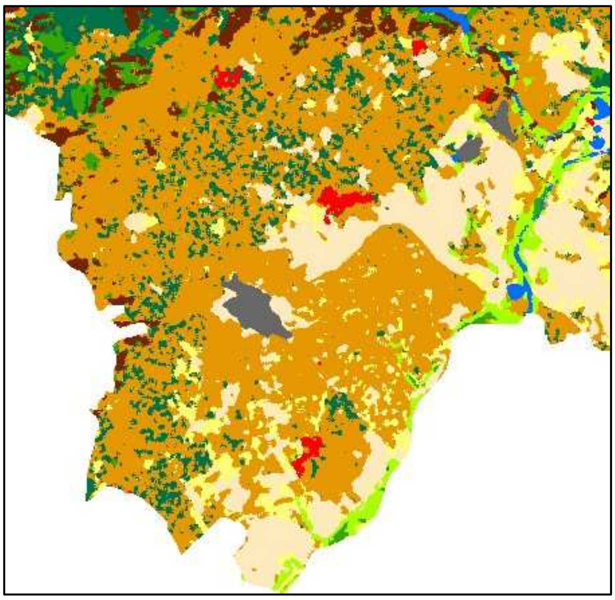

Figure 8.3 Villa del Prado Land-cover for 2026 under scenario M1. This map is an example of those shown to interviewees during presentation of model results. Maps were accompanied by videos to demonstrate the land-cover dynamics that led to this state. Parameters for scenario M1 are presented in section 5.5 and Table 8.1. Legend as for Figure 2.10.

When asked to consider output (i.e. maps and movies) from the simulation models for the scenarios presented, interviewees were very accepting of the results. Comparisons of simulated maps for 2026 with their own sketches showed that model output from at least one scenario closely matched the patterns of change suggested. The chosen scenario was one of M1, D1 or D2 but never M2 or M3 (Table 6.1). On the broad scale at which interviewees' made their sketches model results were comparable, but at finer resolutions model results provided a 'patchier' picture than sketches. Direct quantitative comparison with model maps on a pixel-by-pixel basis was not feasible because of the broad spatial resolution of interviewees' sketches. Whilst interviewees were encouraging about the feasibility and accuracy of the model output, rigorous spatial criticism of the maps at a fine resolution (i.e. 30 m pixel) was not forthcoming. One interviewee highlighted an area of the Villa del Prado municipality that he 'knew' would make a transition from agriculture to a private hunting concession, which the model did not predict, but otherwise accepted the model results:

"In the north east of Villa of the Prado agricultural activity is going to disappear. There are large properties that were cultivated of cereals or Dehesa. Now one will become hunting for sure. This has already happened in one other property."

Local Manager for Regional Government Development Council

Some other interviewees made minor comments along these lines but generally interviewees were happy to examine the resulting model maps in a holistic manner and were satisfied that the maps looked realistic and feasible. Interviewees (possibly inevitably) made the strongest comparisons with model output for the scenario which best reflected their expectations of LUCC (established via questions and discussion from section 8.3.2). In some cases interviewees identified one of the five maps presented as being the 'right' one, despite not being asked to select among the results for the 'best' representation of change. At this point in the interview process, the model structure had not been introduced. It was clear however, that it was the processes of change rather than the resulting landscape patterns (in 2026) that interviewees were interested in talking about, and about which they had a more detailed understanding. The maps produced by the model were seemingly accepted as a credible representation of reality with little criticism (the example above the only exception) and conversation would immediately shift back to the previous discussion on processes and causes of change (section 8.3.1). The positive feedback from the comparison of model maps for these preferred scenarios suggests that the model had achieved a level of mimetic realism and reflected the expected change well enough to encourage discussion about processes of change in the study area (before model structure had been introduced).

8.3.4 Model Assumptions

Research Question: Do the simulation model assumptions/rules/parameter values 'make sense' or 'sound right'?

Interviewees were presented with the key assumptions of the simulation model so that they might assess and evaluate their pertinence and accuracy. The key assumptions presented were:

a) Ecology diagrams with durations and times to change (Figure 4.3)
b) Agent behaviour
    i) Agent Perspective (Table 5.1)
    ii) Land-uses (Table 8.1)
    iii) Agent Age (Table 8.2)

Table 8.1 Comparison of profitability and costs of land-use types. This table was presented to interviewees when explaining the economic rationale of the model. This aspect of the model is described in section 5.4

	Crops	*Pasture*
Relative Profitability (per unit area)	Higher return and costs	Lower returns and costs
Spatial Configuration	Economies of scale reduce costs of larger fields located more centrally within farm	Costs not influenced by size of field but increase with distance from centre farm

Table 8.2 Model assumptions regarding agents' age. This table was presented to interviewees when explaining the behaviour of model agents. This behaviour is described in detail in section 5.4

Agent Age
There are three impacts of increasing age on agent behaviour; i) older agents are more likely to shift to traditional activities if market weakens ii) agents' ability to maintain land decreases with age iii) if an agent has no son at the age of retirement (65 years) commercial agents become traditional

Interviewees were also presented with the initial land-tenure map used in the model to assess their understanding (by generating a discussion) of the land-tenure structure of the study area (and possibly how it has changed over time). This discussion was followed up with a question regarding one of the key findings from exploration of the ABM/LUCC; after market conditions, the model suggests (spatial) land-tenure structure is an important driver of LUCC – does the interviewee agree?

This section of the interview aimed to elucidate how well the model structure fits with stakeholders' mental models of LUCC. Both model structure and initial conditions (i.e. input data) are uncertain and should be assessed relative to the interviewees' responses. In this part of the interview the research team considered how detailed responses were, and noted how aspects of the model focussed on were related to each interviewee's experience and relationship with the landscape.

Whilst all interviewees indicated that they viewed model assumptions and process representation as being adequate, most highlighted caveats. First, in keeping with their views on important drivers of change (section 8.3.2), interviewees suggested that representation of market mechanisms and the influence of subsidies required more explicit representation:

"If, when making your model, you leave out variables like land prices, you will be missing data that explain why crops are lost [i.e. abandoned]. These [variables] are tied – abandonment is mainly in the zone next to the urban areas. The immediate profit [of selling the land for urban development] means that people sell those agricultural properties."

Local Development Agency Manager

Feedbacks between supplies and demand, and explicit contribution of subsidies to values of crops and pasture land-uses, are not present in the model structure. Market prices and costs of production are provided by external scenarios of change. This method was adopted as the focus of investigation is on the result of market conditions for LUCC, rather than an examination of the functioning of agricultural markets. Furthermore, whilst subsidies are not explicitly represented, they are implicitly assumed to contribute to the values of crops and pasture land-uses. However, the criticism of interviewees is valid and future versions of the model will need to attempt to make more explicit representation of economic processes, as this is clearly one of the most important factors driving agricultural land-use decision-making.

Second, the agent types (i.e. commercial vs. traditional – section 5.4) were recognised as sensible representations of the attitudes and perspectives of actors in the study area. One interviewee even made such a distinction between two similar types of actor in the study area in the first section of the interview (section 8.3.2), before the model structure and assumptions had been introduced:

"Those that work and can live on the land really are only three or four persons. The others work part time, for additional income, or do it as cultural occupation, inherited from parents or grandparents."

Local Council Development Officer

This interviewee had been interviewed previously during development of the ABM/LUCC (chapter five). On that previous occasion he described what he believed were local attitudes toward agricultural decision-making. However, at that time the interviewee did not do so in terms of a dichotomy between full- and part-time (hobby) farmers. Other interviewees highlighted that such a straightforward dichotomy of agents was rather simple, pointing out that there would be differences between farmers cultivating olive groves, almond orchards and vineyards for example. However, it was acknowledged and understood that a limit on the number of agent types must be placed in any model and that the two used in the model here were the most appropriate dichotomy. Future versions of the model that represent market dynamics more explicitly may be able to capitalise on this improved representation by also considering types of agent based not only on their perspective but also on their preferred crop type.

Third, interviewees did not believe the influence of farm fragmentation and distance to the nearest road (Eq. 5.1 – Eq. 5.3) was of much importance in agricultural land-use decision-making:

"Vineyard abandonment is not a distance question, it is a question of whether it is productive or not. In addition the town is not a market centre … in fact, there could be a reverse effect. The closer areas to town will be abandoned first [due to higher land prices for urban development], whereas land further away will be maintained production."

Local Council Development Officer

Rather, absolute prices of products were deemed to be more important. If a product could be sold at a high price, interviewees suggested the locations and configuration of fields would not hinder or prevent production of those products:

"I do not think the distance rules are very useful. The subject of distance does not influence decisions here. Everything is so close; it's a maximum of 7 km [from field to town]. That's no distance."

Local Agricultural Co-operative Officer

Such comments suggest these spatial rules could be removed from the model, or at least modified such that the influence of spatial configuration is reduced. These comments also highlight that the interviewees think about the landscape and drivers of change in predominantly non-spatial terms. It is likely that these rules are one of the factors leading to the finding that the spatial configuration of land-tenure had an important influence on land-use decision-making in the model (section 5.5).

When asked about the importance of land-tenure within the study area (section 8.3.4), some interviewees responded that it was not important for land-use decision-making, whilst others suggested that the size of ownership parcels would have an influence. For example, one interviewee suggested (section 8.3.2) a transition from agricultural land-use to hunting in Villa del Prado would occur in this particular area because land-tenure patches were large enough there to make such a leisure-based enterprise financially feasible:

"Those that have big properties can maintain the activity through scales of economy."

Local Council Development Officer

Generally however, responses suggested that the emphasis on spatial influences of change is unwarranted when modelling land-use decision-making. Rather, the focus should be on establishing and representing explicitly the economic and social drivers of change. This mis-match highlights inadequacies in the current model structure (and is discussed further below – section 8.4.1).

8.3.5 Model Modification

Research Question: How could the model be improved to represent more accurately the processes of change in SPA 56 in a manner such that it is relevant to the interviewee's SPA 56-related activities?

Interviewees were asked to identify shortcomings of the simulation model, and asked how they would modify its structure to improve its representation of LUCC. Implicit in this question is the assumption that the stakeholder would modify the simulation model such that it would match more closely with their mental model of LUCC processes. Other relevant questions here are; is this model and/or its results useful to the stakeholder? If it is not very useful, how could the model be modified to become so? Specifically, which assumptions/rules need to be changed? How should these assumptions be changed and why? What new assumptions need to be introduced? These questions refer directly to the model structure and practical adequacy evaluation criteria.

The integrated socio-ecological simulation model (SPASIMv1) was developed with the intent of examining potential impact of LUCC on future wildfire regimes. When asking how interviewees thought the model could be improved it was clear that their responses were couched in their own interests, experience and expertise rather than the stated aims of the model. A universal suggestion by all interviewees was the consideration of urban growth and development, for example:

[What have you not considered in the model that is important?] "The urban aspect. The urban speculation. The price of land has risen. I would like to buy land but it is very expensive and I can't afford to. If somebody wants land to cultivate often they cannot buy, because it is being sold with urban speculation in mind."

Vintner

Urban growth and influence on land prices was seen as a primary concern, particularly for those considering San Martin de Valdeiglesias. Urban development is again linked to the wider economic and social changes occurring in the Madrid region and already in evidence across the study area (section 2.5). Incorporating some aspects of urban growth into the LUCC model was considered during model construction but not implemented as it was not primarily related to the main ecological focus of the research. The growth of existing urban centres will have little impact on the model's representation of fire ignition and growth, as they currently compose such a small proportion of the landscape (less than 2%). Possibly more directly related to the model's representation of fire spread was the suggestion by one interviewee that the model should consider road-widening projects currently being implemented in the study area. Providing greater and easier access to and from the city of Madrid these measures are likely to increase the number of visitors to the area. While not explicitly considered, the impact of increases in visitor traffic to the area on important processes can be represented (e.g. wildfire, section 4.5.2). Whilst deemed non-essential for the initial model construction phase, future iterations of the modelling process will need to consider the incorporation of urban and road-network growth.

When asked about the potential of the model to aid their SPA 56-activities all interviewees were positive about the broader potential planning and decision-making uses. The degree of enthusiasm was linked to the institutional situation of the interviewee and their general attitude toward future landscape change in the study area. Interviewees situated at higher institutional levels with a better view of system-wide change and a remit for planning (regardless of whether it was explicitly agricultural, environmental or otherwise) were most enthusiastic about the potential use of the model within their work:

"These [types of] models complement the perceptions of change we have. But it must be available for end-users for decision-making. … With forecasts of the future, we can make plans and assign funds for certain aspects. For example, we want to develop our tourism sector in the future, but we haven't considered the effects of the abandonment of the agricultural sector. This type of model helps us to make decisions knowing how this is going to happen."

Local Development Agency Manager

The presentation and discussion of the model with interviewees here focused on the agricultural decision-making and vegetation dynamics components of the model. However, the previous quote suggests the use of the model to consider the combined influences of tourism and agricultural abandonment on wildfire is something certain stakeholders will wish to discuss. Furthermore, it may provide an opportunity to fill some of the knowledge gaps and uncertainties highlighted in chapter six regarding ignition location and frequency of human-cased fires. Future iterations of the modelling process should be able to exploit the experience in this stakeholder evaluation exercise to improve the wildfire dynamics component of the model. Another regional planner particularly enthusiastic about the model as it (or more specifically, the results from one model scenario) confirmed his expectation of future change, and thus anticipated the use of the model as a rhetorical tool in future discussion and debate. Another interviewee responsible for local and regional development was particularly enthusiastic about the potential of the model for his use:

Observing these models helps you to see the [overall] situation. When you see it in a map you see it [LUCC] clearer. The model is excellent for planning or managing a zone [of the municipality].

Local Manager for Regional Government Development Council

The root of this interviewee's enthusiasm may lie in the close alignment of the model results from one scenario with his expectation of future change (section 8.3.3). Thus, he may have been anticipating the use of the model as a rhetorical tool in future discussion and debate. Individual land owners and those set within small agricultural institutions indicated that they thought the model would be useful, but probably not to the same degree as their own personal knowledge of the study area:

"[These models are useful,] yes. But when you are the subject of the model, already you have an idea of what is happening. You are already anticipating the change yourself."

Local Agricultural Co-operative Officer

This point exemplifies Hacking's (1999) discussion about interactive kinds (section 7.2.2). That is, the agricultural decision-making agents may be aware of their classification and representation within the model and therefore modify their behaviour based upon the results of the model. However, the implication of this quote by a representative of individual farmers is that many of these individuals (likely those with little influence over much else in the system but their own actions) will value their own knowledge more highly than the results of the model. Interviews and discussions with those explicitly represented in the model would be interesting to examine the degree to which actors' behaviour is modified by contact with the model.

8.3.6 Changes in Understanding

Research Question: Does the stakeholder think this discussion (i.e. steps one to four) has modified their mental model (i.e. understanding) of the processes of change occurring in SPA 56?

Towards the end of the meeting, interviewees were asked to reflect on whether/how discussing and thinking about the simulation model and its output had influenced their understanding/perception of potential change in their area of SPA 56. Follow-up questions explored how the interviewees thought that their involvement at other stages of the modelling process would have benefited both the simulation model (i.e. the modeller's mental model) and their understanding of SPA 56 (i.e. their mental model).

This conclusion to the interviews aimed to assess how (whether) the interviewee's understanding of change in the study area had been changed/modified by the interview. This was also an opportunity also to assess the interviewees' general attitudes toward this type of modelling. However, as not asked prior to the meeting about these attitudes, it is difficult to discern where/how these attitudes have been developed and thus how they have changed as a result of the interview.

Upon concluding interviews it was quite clear that no significant change in interviewees' understanding of the processes of ongoing landscape change in the study area had occurred. This is probably because the presentation of the model structure, and the model structure itself, is not as detailed or nuanced as their understanding of the processes of change. Much of interviewees' understanding of change within the study area is based in the social and economic changes due to wider regional and national shifts in industry (away from the primary agricultural sector) and lifestyle (away from traditional 'rural' lifestyles to increasingly modern lifestyles that afford greater leisure time). Furthermore, the interviewees' understandings of the study area are far more idiosyncratic than could possibly be represented in any simulation model. Greater affinity with the model may have been developed if it had represented in some way the activities of those actors in the study area once they had left the agricultural sector. Also, had the interviewee been able to interact with the model and specify parameters and scenarios of change themselves, a more detailed understanding of the model and its results and implications may have been possible. The possibilities of model interaction were not examined here as the focus was not on participatory approaches that have been explored elsewhere in the literature (e.g. Castella *et al.* 2005b).

8.3.7 Summary

This section has presented the results of interviews with stakeholders regarding the socio-ecological simulation model developed in this thesis. In turn, the interviews considered drivers of LUCC, model results, model assumptions, model modifications, and potential changes in understanding as a result of this process. The next section evaluates the model against four assessment criteria and discusses the value of this approach, as well as the problems encountered.

8.4 DISCUSSION

8.4.1 Model Validation Criteria

Alongside the model evaluation criteria of mimetic and structural accuracy, the previous chapter made a case for the additional criteria of trust and practical adequacy (at least in models of socio-ecological systems). Via the evaluation exercise above, the simulation model here can be measured against these criteria to establish its performance and worth. In general the model seems to be somewhat successful in representing LUCC in SPA 56, but with clear room for improvement in future model iterations across all four criteria (model structure, mimetic accuracy, trust and practical adequacy).

First, interviewees were of the opinion that the current model structure was acceptable as a representation of the processes of change occurring in SPA 56. For example, the distinction between 'commercial' and 'traditional' farmer types (section 5.4) was deemed useful, and the assumptions regarding vegetation dynamics (i.e. broad land-cover types and succession pathways – Figure 4.3) were accepted. However, several caveats were made and there seemed to be a mis-match between the modeller's mental model and those of the interviewees. First, interviewees were either indifferent to, or critical of, the spatial rules on decision-making (e.g. Eq. 5.1 – Eq. 5.3) and suggested that they need not, or should not, be included in the model. Suggested to be of greater importance was a more direct representation of crops prices, subsidies and the market. This implies that the spatial aspects of change of interest to the modeller (because of his academic background in geography and landscape ecology) are not the most important aspects of change to consider from the perspective of local stakeholders. Second, many interviewees highlighted that the model lacked the consideration of urban growth. A representation of urban growth and change was considered for inclusion in the model but not pursued as it was not believed (by the modeller) to be a primary driver of LUCC in the study area and therefore not worth the computational or model development costs inclusion would demand. Again this implies that the (funded and academically-relevant) research interests of the modeller are not as important to local stakeholders as other aspects of change in SPA 56. Third (but not a direct mis-match between mental models), whilst the dichotomy of 'commercial' and 'traditional' farmer types was accepted, it was suggested by some interviewees that a more detailed representation of specific agricultural types (particularly between arable farming types – olives versus grapes versus cereals for example) would be useful to represent decision-making more accurately. Thus, while the model structure was deemed adequate and not incorrect, the issues of equifinality described above (section 7.2.2) are evident in the fact that interviewees have highlighted several model structure modifications.

These mis-matches between the modeller's mental model and those of the interviewees mean that the model also suffers when evaluated against the practical adequacy criterion. For example, because the model does not address the concerns of interviewees regarding urban growth and associated infrastructural changes (e.g. school and health service requirements) it might be deeded practically inadequate. Indeed, the Local Development Agency Manager highlighted the importance of ensuring models were practically adequate for use (see quote in section 8.3.5). However, some interviewees were enthusiastic about the potential utility of the model (e.g. see Local Manager for Regional Government Development Council quote in section 8.3.5). Planning officials in particular were more enthusiastic than those directly concerned with agriculture and whose decisions are based on an individual (single farm) basis (e.g. the Vintner and Local Agricultural Co-operative Officer). This enthusiasm suggests a certain degree of trust in the model (the third assessment criterion) by some potential model users. Whilst trust is difficult to measure, and the research team did their best to observe interviewees' body language alongside their verbal discussion, it could be the case that interviewees were enthusiastic simply because they were happy someone (particularly a certified, academic) was taking a serious interest in the issues facing their local region. Furthermore, interviewees may themselves benefit from meeting with a certified expert, and have *instrumental* motivations (*sensu* Stirling 2006, section 7.3.2 – see next section for further discussion). Trust in a model will only be truly tested when those using the model have to live by the decisions or implications arising from their use of the model. In the case here however, the model was trusted to the degree that interviewees said they would be happy to work with it (and the modeller) in the future.

Finally, the mimetic accuracy of the model was implicitly evaluated by the interviewees when asked to assess the feasibility of the maps produced by the model (section 8.3.3). At a broad scale (e.g. Figures 8.2), interviewees accepted the model was a mimetically accurate representation of the change they expected to occur over the next two decades. When the model did fail to match their expectations, interviewees were able to point out regions where the inconsistencies were and, most importantly, *why* it would fail at these locations. For example (and as presented above section 8.3.3), one stakeholder highlighted an area of his municipality that he 'knew' would change from agricultural land-use to hunting. The interviewee's projection was based on his specific knowledge about the land-tenure structure and the personal relations between the land owners in that region. The model does not, and probably could not, take account of these aspects of LUCC dynamics. The need for (spatial and logical) model closure (section 7.2.2) highlights that the drivers of change for areas within SPA 56 are far more nuanced than the model can represent explicitly for the study area as a whole. The type of analysis made by this particular interviewee provides a test of mimetic accuracy based on process knowledge that a simple pixel-by-pixel comparison of maps (e.g. section 3.3) cannot provide. However, it should be noted that this criticism is a little unfair, as a pixel-by-pixel comparison would not be possible in this manner because no empirical map of 2026 is currently available. It is hard to assess how the model would perform on a pixel-by-pixel comparison, but it is hoped the model would perform better than an empirical model of the type presented in section 3.3. Recently, Brown *et al.* (2005) proposed a method to examine how well a model performs in predictive versus process terms, by evaluating which areas of (spatial) model output would be more or less likely to vary due to path dependence or stochastic uncertainty. Such a method offers promise for greater understanding about the roles of historical and geographical contingencies in landscape change, but has limited use for models with high resource requirements (and so cannot be replicated many, many times – such as the simulation mode presented here) and still does not address the specific source of either path dependencies or stochastic uncertainties. Advances in technical pixel-by-pixel approaches and the benefits of process-based evaluation of stakeholders ensure mimetic accuracy will continue to be an important model assessment criterion in the future.

8.4.2 Engaging Non-Modellers with Models

As outlined above (section 8.2), several studies have examined issues regarding the public understanding of scientific models (Yearley 1999, Yearley 2000, see section 8.2, Cockerill *et al.* 2004, Olsson and Berg 2005). These studies examined the relationship between, and understanding of, stakeholders and lay public and models of air and water quality. The interviews presented in this chapter differ from these studies in several ways. First, the subject of the model presented here considers LUCC, an environmental condition over which land owners and managers have a direct influence via their decision-making. Second, whilst the interviewees here can be considered stakeholders in the study area, as they have interests in the future state of the study area, they cannot be considered lay public as they all have (with the exception of the vintner) expertise in some particular area of land-use/cover change and decision-making. Thus, the discussion here cannot be interpreted as comments on the 'public' understanding of science. Rather, they should be understood as the examination of the engagement of individuals with no expertise in simulation modelling (i.e. non-modellers) with a simulation model representing environmental change (LUCC) that they have a degree of control over in their decision-making capacities. Third, engagement with local stakeholders in those previous studies occurred on completion of the modelling process, between the 'applications' and 'consequences' phases of the scientific research process as conceived by Jackson *et al.* (2005, Figure 7.1). In contrast, the engagement here is considered as an integral component of the modelling processes, particularly with regard to model assessment. This engagement has involved non-modellers in the actual process of model evaluation that modellers regularly undertake themselves in private. This approach opens the modelling process, rather than discussing the outputs of a closed and opaque modelling process as previous studies have (i.e. Yearley 1999, 2000, Cockerill *et al.* 2004, Olsson and Berg 2005). The foci of these previous studies were issues of model/modeller trust in the eyes of the lay public rather than the model itself. Bringing this engagement within the walls of the modeller's workshop has allowed a model evaluation based on criteria that are useful (given recent realist and relativist attitudes toward environmental modelling – section 7.2), but that would otherwise be impossible. Here, stakeholders were explicitly asked for comments on the current model structure, and ways in which the model could or should be improved to provide a more accurate representation of the processes driving LUCC and which would ensure practical adequacy of the model for stakeholders' purposes.

The value of the simulation approach over that of the empirical (statistical) approach is the dynamic consideration of process and interactions that allows the model to be experimented with via scenarios of different socio-economic change. However, several interviewees indicated that what they believed to the 'correct model' was the map resulting from the scenario that fitted best with their understanding of future change *before* they had been introduced to the model structure. Thus, these interviewees seem to be conflating several aspects of the modelling process and the possible evaluation criteria. First, the interviewees are using a test of mimetic accuracy to assess model structure. That is, given no other knowledge about a model, the non-modellers automatically used a mimetic accuracy test to evaluate the model. Such an approach is a likely response of anyone first approaching a model, the structure of which they are ignorant of. Such an approach might influence subsequent evaluation of model structure, but it is difficult to say if this happened here. This could be examined in future and other studies by introducing model assumptions prior to model results (i.e. reversing the order of section 8.3.3 and 8.3.4). Second, and important given the recent advocacy of public engagement and participatory approaches, stakeholders' conflation of the model with the scenarios it is run for presents a clear problem if they are to be asked to evaluate the model. If model and scenario are taken by stakeholders to be synonymous, assessment will be about how well each scenario matches their expectation of future landscape state, rather than how well the model matches their understanding of the processes occurring. If this conflation of model and scenario is apparent, clearer explanation about the difference between model and scenario will be required. An understanding of the distinction would need to be proven before model assessments could be accepted as useful. Furthermore, given that some interviewees here found it difficult to think spatially when asked to project future change, the assessment of spatial model output by those interviewees is likely to be weaker than those with a better spatial conceptualisation of change. In this case discussion with interviewees about the maps and how they represent the places they know may improve spatial understandings of change.

An inadequate understanding of the distinction between the model and the scenarios it is run for may result in unintentional bias toward or against the model by an interviewee. As highlighted above, intentional bias may arise if a stakeholder anticipates potential benefits of accepting and working with the model and modeller. One stakeholder interviewed here, with a role in local development, viewed the model in a particularly favourable light as the model matched well with his personal projections change. Thus he may have anticipated rhetorical uses of the model to the benefit of his organisation's objectives. Such (potential) bias will cause problems for stakeholder evaluation if not acknowledged and handled appropriately. For example, when attempting to use local stakeholder knowledge to improve model representation of an empirical system it should be remembered that each stakeholder will have their own agenda that may bias their interpretation of how the systems works. This may result in them (unintentionally or otherwise) mis-representing the functioning of the system to the modeller, in turn producing a model based on an inaccurate description of the system. To avoid this undesirable situation a broad and representative sample of local stakeholders should be canvassed for their understanding of the system function. This sample should attempt to represent all 'types' of stakeholder in the study area (e.g. local producers as well as multi-national corporations) and all sides of any known groups with conflicting interests regarding the issue at hand. Problems of intentional mis-representation are not believed to have been present in this study as the focus of the study is not particularly controversial and provides little opportunity for commercial, political or other gain by manipulation of the model.

In the interviews conducted here, step one (section 8.2.2) was very useful to initiate discussion and put the interviewee at ease by inviting them to talk about a subject of which they have intimate knowledge. This allowed interviewees to relax in a topic that they felt comfortable with before embarking upon discussion about a potentially alien way of understanding LUCC (i.e. simulation modelling). The level of criticism levelled at the model by interviewees was linked to the level of detail at which the model was presented. That is, interviewees were only able to criticise the model in general terms as the relatively short duration of the interviews (1 – 1 ½ hours) did not allow a detailed presentation of the model. The value of a lengthy, detailed presentation of all model assumptions and parameters (akin to chapters four and five) is questionable. It is unlikely that a formal presentation of these details (over the course of a couple of hours) would do much more to increase interviewee understanding of the model, and if a more detailed understanding of the model was desired, interviewees would need to use and 'play' with the model and its parameter values for different scenarios, in much the same way as the modeller does during model development (see sections 4.6 and 5.5). Because of the high computational demands of the model developed here, such an interactive presentation of the model was not practical.

The model proved itself useful, or warrantable, as a form of discursive tool. Interviewees were able to compare and contrast model output with their thoughts about future change, think about the consequences of individual action over wider extents, and use the model as a jumping-off point for a discussion on what innovation in human activity they envisaged occurring that the model did not account for. However, some interviewee's fatalistic attitude regarding change and the causes of it did inhibit discussion at times. In some ways this approach is similar to the idea forwarded by Morrison and Morgan (1999) of models as mediators between theory and the real world. In this case, the simulation model has provided a means by which to manifest the modeller's mental model of LUCC in a form that the interviewee could interact with in a tangible manner. In turn, through discussion, the interviewee's mental model becomes more apparent and better understood by the modeller, providing opportunities to improve and develop the model. Thus, the simulation model acts a mediator between the mental models of the modeller and interviewee, between the 'certified' expert and the 'experience-based' expert (sensu Collins and Evans 2002, see section 7.3.3). Given the credibility the model has achieved here, it could seemingly act as a mediator between the mental models (i.e. understandings) of stakeholders and decision-makers in the study area (see van den Belt 2004). However, group model evaluation and associated discussion, or the interaction of stakeholders with the model itself (e.g. running the model for different parameter sets or scenarios), would probably improve upon the one-to-one interview-type discussions used here.

8.5 SUMMARY

This chapter has demonstrated some of the advantages of stakeholder evaluation over more conventional pixel-by-pixel pattern-matching techniques. A model assessment exercise that engages with stakeholders offers greater opportunities to interpret and assess model results in terms of processes of change. For example, here the consideration of 'commercial' and 'traditional' agricultural decision-making agents was confirmed as an appropriate representation, but the omission of the representation of urban growth and change was highlighted as an important shortcoming. Further rounds of the iterative model development processes will be able to build on this criticism to improve the model based on information that a simple pixel-by-pixel pattern-matching approach would not provide. The stakeholder evaluation process indicates that the model has been somewhat successful in representing LUCC in SPA 56, but also highlighted clear areas for improvement in future model iterations across all four model assessment criteria used. Stakeholders generally accepted model output as representative of anticipated future change, but suggested the model lacks a consideration of urban change, that market mechanisms are under-represented, and that certain of the chosen spatial decision-making rules are largely irrelevant in contemporary agricultural landscapes.

The results suggest that simulation models can act as a muse to inspire reflection about processes causing change and provide a common focus around which debate about processes of change can proceed (as others have suggested van den Belt 2004). Simulation models can offer stakeholders and decision-makers an insight into states of potential future environmental systems given different scenarios of human activity. By attempting to 'open up' the model for scrutiny by non-modellers, this chapter has gone further than other studies examining issues of trust and expertise (e.g. Yearley 1999, 2000). Incorporating local knowledge throughout the model assessment and evaluation phases – increasing the level of 'upstream' engagement and understanding between modeller and non-modeller – will be a useful strategy to ensure appropriate models of socio-ecological change are developed. Furthermore, the 'non-certified expertise' of local stakeholders provides a means to clearly identify shortcomings and uncertainties in socio-ecological simulation model structure and results. However, such an approach is not without its problems. Some stakeholders had problems distinguishing the model from the scenario and others with thinking about LUCC spatially, highlighting issues of communication and understanding. Fatalistic worldviews and over-eagerness to accept results that fitted with expectations also suggests criticism from an 'extended peer community' (section 7.3.2) will not necessarily be as rigorous as would be expected from the scientific peer community. These issues will need to be considered in future stakeholder evaluation exercises.

CHAPTER NINE

DISCUSSION AND CONCLUSIONS

9.1 INTRODUCTION

This socio-ecological modelling project has taken an interdisciplinary approach (i.e. has used theory and methods from ecology and sociology) to represent the interaction of LUCC and wildfire. Brewer (1999) discussed several opportunities an interdisciplinary approach offers, including the emphases of the approach on the orientation of problem formulation and the context of research practice. Funtowicz and Ravetz (1993) have suggested the task of scientific quality assurance should not stop at product (the model and its results) but must also consider the process (the modelling), particularly with regards to the social aspects of the process. In this concluding chapter therefore, a reflexive approach is taken which considers some of the more contextual aspects of the research in order to summarise what has been learned from the modelling process of this thesis. This reflexivity has been more common in the social sciences than the natural sciences but, as Brown (2004) has suggested, modellers of physical environmental processes might also learn from examining the process by which they gain knowledge. Socio-ecological modelling lies at the interface between the social and natural sciences and remains a relatively immature research area. As such, it seems pertinent to reflect on what has happened in this modelling project so far. This chapter presents the research in chronological order, taking what is termed a 'narrative approach'.

9.2 NARRATIVE APPROACHES TO REFLECT ON THE MODELLING PROCESS

A narrative approach is used here in an attempt to 'open up' the process of model construction for better scrutiny, and to allow better evaluation and understanding of the modelling process and its results. In recent years a recognition of the historical nature of many forms of earth and environmental science (e.g. Frodeman 1995) have led to suggestions that 'narratives' would be useful tools to foster better understanding in subjects like geology (Gould 1987), environmental history (Cronon 1992) and science in general (Allen *et al.* 2001, Zellmer *et al.* 2006). In concluding a review on complexity science, human geography and modelling, O'Sullivan (2004) suggested the possible use of narratives to examine the modelling process to reveal the 'story' behind the decisions that were made during model construction. The presence of equifinality highlights the importance of observer dependencies in simulation modelling and the decisions modellers must make regarding the location of model boundaries (model closure – section 7.2.2). Simulation modelling is undertaken in situations in which data on the system being examined are sparse and logical transformations (approximations) are required to represent the system *in silico*. The justification of these transformations is made on a variety of fronts including theoretical knowledge, empirical generalisations, and experience from modelling similar phenomena (Winsberg 2001). Making explicit the implicit decisions and understanding made in the development of a simulation model is important if it is to be adequately evaluated.

The modelling process is not linear from conceptualisation through model execution and testing to analysis and interpretation of results. Rather, it is an iterative process that demands movement back and forth through these various stages, with modification and refinement, as a continual process (e.g. Figure 8.1). Iteration is required as the modeller(s) learn more about both the system they aim to represent and the adequacy of the model structures that have been put in place to do so. 'Dead-ends' are often encountered in the research process, but current academic and publishing conventions regularly inhibit the full story to be told of what was learnt from the (iterative) modelling processes (O'Sullivan 2004). Furthermore, if science, and in particular socio-ecological simulation modelling, is to move 'upstream' (sections 7.3.2 and 8.2), a more creative mix of formal and informal methods will be required to 'democratise science' (Wilsdon and Willis 2004) and reveal its inner workings. A narrative approach to presentation of the modelling process might help improve communication about both what was learnt and how it was learnt.

A narrative in the sense used here is a formal, but non-technical, representation of the history of events that "organises all representations of time into a configured sequence of completed actions" (Cronon 1992, footnote 5). Narratives provide meaning and understanding about events by representing these events as causal sequences, thereby ordering and simplifying. Thus, this chapter attempts to discuss not only what SPASIMv1, its results and the process of constructing it tell us, but also how this knowledge was gained. The success of a storytelling can only be evaluated by the audience – hopefully this narrative will be useful to provide insight into the messiness of the research process that this thesis presented as a formal and orderly progression.

9.3 SPASIMv1 Modelling Narrative

9.3.1 Research Proposal

The research presented in this thesis was funded by a joint UK Economic and Social Research Council/Natural Environment Research Council (ESRC/NERC) Interdisciplinary Research Studentship and undertaken within the Department of Geography at King's College, London (KCL). The interdisciplinary ESRC/NERC studentships are designed to encourage "postgraduate research on environmental issues which are of interest to both Councils and which require the combined approaches of both the environmental and social sciences" (ESRC 2007). The application for the studentship arose from my interest in studying wildfire/vegetation-dynamics modelling with Dr George Perry (then at KCL – currently at University of Auckland, New Zealand). I had been supervised by Dr Perry during my undergraduate degree dissertation, and we strengthened our relationship during my Master's studies of environmental modelling and management. At the time Dr Perry was completing a manuscript on the role of land abandonment in the landscape dynamics of SPA 56 with Dr Raul Romero Calcerrada (then at Universidad de Alcalá, Spain – currently at Universidad Rey Juan Carlos, Spain) that was later published as Romero-Calcerrada and Perry (2004). With data available, a willing collaborator, and an academically established research question in the early stages of study (i.e. the question of the importance of agricultural land abandonment on wildfire dynamics), the application was made in May 2003. The original research objectives stated in this research proposal were to:

(i) develop a spatially explicit model of vegetation dynamics that considers both wildfire and human activity in the Mediterranean;
(ii) produce scenarios of future land-use change based on expert input from local collaborators, surveys of and discussion with local stakeholders, and informed judgement based on the literature;
(iii) use the model to explore the implications of these independently generated scenarios for future fire regimes;
(iv) use model outputs to assess, in collaboration with local stakeholders, the management implications of potential landscape changes to maintain future biodiversity and sustainability.

Whilst these objectives bear some resemblance to the stated research aims in chapter one (section 1.4), the emphasis on model evaluation made in chapters seven and eight is clearly absent. Furthermore, there is a much stronger emphasis on scenarios of land-use change and their generation with the aid of local stakeholders than was realised here. Much of this shift was a result of the encouragement of my ‘social science supervisor’ Dr David Demeritt, whose research interests lie in cross-cutting areas of nature-society expertise and the articulation of environmental knowledges (particularly scientific and technical). ESRC/NERC studentships require a supervisor from each of the environmental and social sciences – initially, Dr Geoff Wilson was proposed as the social science supervisor. However, soon after the success of the ESRC/NERC application, Dr Wilson moved from KCL to the University of Plymouth and Dr Demeritt replaced him. Other changing ideas and influences are discussed below.

9.3.2 Initial Exploratory Research

In early October 2003, soon after completion of my MSc thesis (Millington 2003), I set about a more detailed literature review of Spanish landscape change and wildfire (chapter two) and some initial statistical modelling and data exploration (chapter three). These two areas of research took me up to, and composed, my interim ‘upgrade’ report in November 2004. The report takes this name as its completion and acceptance following an oral defence ‘upgrades’ the student from MPhil status to PhD status – thus it is seen as a formal stage of the PhD research process in the Department of Geography at KCL. During this time I also made a trip to SPA 56 with Dr Perry and Dr Romero Calcerrada to get a better ‘picture’ (i.e. intuitive understanding) of the study area.

The initial literature review of the issues facing Mediterranean landscapes, with particular regard to the potential effects of contemporary socio-economic change on ecological systems, provided the context for the study and made the case for the need to study the questions required to achieve the first research aim (section 1.4 – though these questions were refined and finalised during the work described in section 9.3.3). The literature review suggested that in traditional Spanish pine-oak woodland landscapes (of which SPA 56 is a prime example) vegetation dynamics are strongly influenced by disturbance, predominantly human activity and wildfire (section 2.2). High spatial heterogeneity in Mediterranean environmental resources (e.g. water) and the spatial nature of wildfire ignition and spread mean the consideration of spatial dynamics is vital to the study of ecosystems in Mediterranean regions (section 2.4.4). Furthermore, lengthy human occupation and the high proportion of wildfire ignition by humans make humans a direct and important driver of land-cover and wildfire dynamics (section 2.3). Changes in human activity recently have led to decreasing land use for agriculture and increasing scrub-like covers (as found by Romero-Calcerrada and Perry 2004), with potential implications for wildfire occurrence and spread.

The literature review and the case made for the research questions regarding the interaction of LUCC and wildfire are justified in their own academic and scientific right. However, two points can be made regarding the context of the framing of this particular literature review and the research questions. First, the literature review and research questions highlight the importance of human activity as a component of ecological processes in the landscape, in keeping with the interdisciplinary remit of the research funding. Second, the research questions focus on the importance of wildfire rather than any of the other outlined environmental consequences of land abandonment. There are currently many similar ecological questions being addressed in Mediterranean environments that focus on problems that are not directly influenced by humans (e.g. Keeley *et al.* 2006, Pausas *et al.* 2006, Pons and Pausas 2006), and the alternative environmental questions posed by agricultural land abandonment could be claimed to be equally important. The particular issues examined in this thesis were chosen because the context of the research project made them most relevant in terms of the project funding and my interest and expertise. There is no scientific or academic problem with this selection process and stressing that the context of the selection of research focus was based on academic rather than political or commercial grounds lends credibility to the process.

My initial literature review also examined the methods used to model LUCC and highlighted the reasons to model rather than examine landscape change using other techniques. Specifically, the large time and space extents involved in landscape study make empirical experimentation virtually impossible because of logistical, political and financial constraints. The use of empirical regression-based models is one approach which has been used frequently to examine LUCC (section 3.2). However, when assessed on a pixel-by-pixel basis the performance of these models has regularly been found to be poorer than that of using no model at all (i.e. a map at time t1 is often a better predictor of change at a later time t2 than a model based on empirical data from time t1, Pontius *et al.* 2004). To examine both this issue and the data available for

SPA 56, we (I with the aid of Dr Perry) developed logistic regression models using both socio-economic and biophysical data (section 3.3.3). The predictive inadequacies detailed by Pontius *et al.* (2004) were also observed in our models for SPA 56. Furthermore, issues regarding differing levels of data resolution between these data types were found to hinder model performance (section 3.3.4). Problems of stationarity mean that these models are likely to perform poorly in regions where processes of change are dynamic, for example in human-dominated areas where socio-economic (i.e. institutional) structures are being modified (like in SPA 56 – section 3.3.4). In light of this empirical modelling exercise, it was suggested that a simulation model approach would prove more useful for representing spatially dynamic processes of change (such as land abandonment and wildfire) and improving understanding about these dynamics (section 3.4).

At the time of the work presented in chapter three, several other aspects of empirical modelling were explored that are not presented in the thesis. These omissions are because the later shift in emphasis toward stakeholder validation and assessment of models restricted what could be included. For example, an empirical GIS-model approach attempted to incorporate aspects of the temporal and spatial dynamics of wildfire regimes into static wildfire risk framework using the results of the logistic regression models presented in chapter three (Millington 2005). I felt that this modelling approach had limited success and that a dynamic simulation approach remained the most pertinent, and as an attitude is representative of the operator dependencies described by Beven (2004). Thus this GIS-model research was somewhat of a 'dead end' (despite being published!), of the type that shapes the researcher's thinking but is never reported as such (O'Sullivan 2004). Other research on explanatory forms of empirical modelling (Millington *et al.* 2007) was not presented in the thesis because of the later shifting emphasis toward work on SESM evaluation. This explanatory empirical modelling was not so much a 'dead end' as it was research that did not fit with the resulting emphasis of the thesis. Again, this process helped to shape my resolve that a more explicitly process-based model was the best way to proceed with the modelling project. Whilst the regression models produced (section 3.3) performed as well as similar previously applied regression models (e.g. Carmel *et al.* 2001), it was felt that such a static and deterministic approach did not reveal much more than was already known from direct empirical observation. It was hoped that a more process-based modelling exercise would provide greater insights into potential future landscapes.

The intended structure of the integrated Socio-Ecological Simulation Model (SESM) at this time was a Landscape Fire Succession Model (LFSM) much the same as the current version, but with human activity imposed in a top-down fashion by broad scenarios of socio-economic change. This attitude, emphasising representation of the ecological processes, was based upon my expertise and interests arising from my physical geography background. During the 'upgrade report' defence however, questions regarding whether the representation of human activity via large-scale scenarios was appropriate and whether an agent-based approach might be pursued. Initially, I was not keen on this idea of incorporating an agent-based approach – I saw problems relating to data availability (i.e. lack of land-tenure data at that point in time), computational requirements, and, indeed, my own technical (computer programming) skills.

9.3.3 Model Conceptualisation and Initial Construction

Following the upgrade report, I started development of the LFSM (chapter four). Work proceeded quickly (relative to development of the ABM/LUCC) as I had a good understanding of the literature on this type of model, of how I thought the model should be constructed, and the raster/cellular-automata approach suited my programming skills well. The literature on LFSM-type modelling (section 4.2) highlighted that the large spatial and temporal extent of the proposed research (1×10^3 km^2 over decades) and the limits of data available for model parameterisation in Mediterranean-type environments demand trade-offs between processing power and model complexity (section 4.2.4). These tradeoffs have been satisfied most frequently in the past, and were satisfied in the LFSM here, by using a spatially-explicit landscape model-type approach utilising conceptual Plant Functional Types (PFTs). The two PFTs that best describe the life-history strategies adopted by Mediterranean-type vegetation to survive in the face of frequent disturbance are 'resprouters' and 'seeders' (representing oak and pine species respectively – section 4.3.2). The Rule-Based Community-Level Modelling (RBCLM) was used to represent changes in 11 land-use categories through time via conditional rules regarding environmental conditions (mainly moisture and light) and vegetation succession-pathways (secondary or regeneration – section 4.4). Land-cover state transitions were modelled using a time-based approach based on a look-up table specifying expected directions and times to transition for pixels under given environmental conditions (section 4.4.1 and Appendix I). Wildfire was represented in the model using a cellular automata approach that explicitly considered ignition frequency and location, particularly with regard to human activity as a cause (section 4.5). The model considers land-cover flammability, slope, wind and human activity as factors influencing wildfire spread.

At this point it should be noted that my supervisory panel had changed once again – Dr Perry had moved to the University of Auckland and was replaced by Prof John Wainwright (then at KCL – currently at the University of Sheffield) whose research focuses mainly on geomorphic modelling but who also has experience in modelling human-environment interactions in Mediterranean environments. During this time I successfully applied for an ESRC travel award to visit the School of Geography and Environmental Science at the University of Auckland for three months to work with Dr Perry and Dr David O'Sullivan. In Auckland (May – August 2005) I reviewed the literature on agricultural location theory and agricultural decision-making (section 5.2) which highlighted recent advances in ABM approaches (section 5.3). The highly spatially heterogeneous and temporally-dynamic nature of environmental resources and land-tenure in SPA 56 suggested that land-use decision-making would be highly dependent upon individual farmers' circumstances. The literature review also suggested very few previous ABMs had explicitly represented interactions between human decision-making and the biophysical environment, and none seem to consider spatial land-tenure structure explicitly as a factor influencing LUCC (section 5.3). An ABM approach therefore appeared useful both to ensure adequate representation of agricultural decision-making and because it was a novel approach that was at the cutting edge of LUCC modelling. However, the literature also highlighted the drawbacks of an ABM approach (and supported my own concerns), including data demands, difficulties in agent representation and model validation issues.

Nevertheless, it had become apparent that an agent-based approach would be required to ensure a properly integrated and novel SESM. The opportunity to work with Dr O'Sullivan at this time was particularly influential (and enlightening) given his expertise in simulation – particularly agent-based – modelling of geographic phenomena. During my time at the University of Auckland I began playing with the NetLogo agent-based modelling environment (Wilensky 1999) and started working on prototype models of agent-based decision making for SPA 56 based on an understanding I had developed via the literature and my previous visit to the study area. Furthermore, it was during this time that we (Drs Perry, O'Sullivan and I) discussed issues regarding modelling the 'real world', which provided the foundation for much of discussion presented in chapter seven.

On return from Auckland I felt that I needed a more secure empirical base on which to further develop the ABM/LUCC. Thus, interviews with local stakeholders (November 2005) were undertaken to supplement the understanding gleaned from the literature (section 5.4). As a result, two different 'farmer types' were conceptualised to represent the attitudes toward land-use decision-making in SPA 56 (section 5.4.2). 'Commercial' agents were based on the frequently used, but fictional, perfectly economically rational decision-making agent *Homo economicus* (section 4.5.3.1). 'Traditional' agents represented the part-time or 'hobby' farmers that continued tending their land regardless of economic profitability, because it was a part of their culture (section 5.5.3.2). Depending upon their type, agents made decisions based upon market conditions, their own personal attributes (e.g. age) and spatial rules regarding their land tenure (section 5.4.3). The conceptualisation of these farmer 'types' was not straight-forward as it was evident that numerous attitudes toward agriculture were present in SPA 56. However, upon reflection of the interviews and discussions these farmer types offered the most divergent worldviews whilst also representing the differing attitudes of older and younger generations of farmers.

9.3.4 Model Construction, Testing and Sensitivity Analyses

The majority of 2006 (January – September) was spent coding the agent-based model structure (developed in NetLogo) in the object-oriented programming language C++, and testing, running sensitivity analyses on, and integrating the two models (LFSM and ABM/LUCC) into the current SESM (SPASIMv1). By this time, land tenure data (i.e. maps of land ownership boundaries) of a quality high enough to represent individual agricultural decision-making agents had become available. The opening sentence of this section does little to describe the many hours of de-bugging, wrestling with understanding what (if anything) the ABM/LUCC model was doing, how best to undertake model analysis and 'validation', and despairing about whether the model would actually work or be of use for anything or anyone. From my experiences modelling LUCC in SPA 56, this confusion, uncertainty and insecurity seems likely to be an inherent part of the process of modelling an open, middle-numbered system for which knowledge about the important (and importance of) processes of change is poor.

The testing and sensitivity analysis of the LFSM was more straight-forward than the ABM/LUCC, largely because the parameter space was smaller and there were less, and more homogenous, interactions between elements of the model (i.e. it was less 'complex' than the ABM/LUCC). The sensitivity analysis showed the fire parameters to be most sensitive and critical influential on land-cover composition and wildfire regime behaviour (section 4.6). These wildfire-spread probabilities have been found previously to be highly sensitive, exhibiting critical threshold behaviour (e.g. Ratz 1995 – and see section 6.5.2, McCarthy and Gill 1997, Perry and Enright 2002b). However, ensuring accurate values were used for these parameters was troublesome because of the difficulty of translating data collected for the parameterisation of models such as the Rothermel (1972) semi-empirical model for use in cellular automata (although Berjak and Hearne 2002 did recently attempt this). Thus, a sweep of the parameter space to find parameter values that produced system behaviour similar to that empirically observed (i.e. similar burned areas and frequency-area relationships) was used to derive the baseline parameters. This approach relied little on process knowledge other than that described in sections 4.4 and 4.5. Furthermore, specification of the soil moisture class definitions (section 4.4.2.2), 'random seed dispersal' (section 4.4.3.2), and 'oak mortality burn frequency' (section 4.4.3.1) parameters are examples of observer dependencies in model construction. Whilst based on the best available empirical evidence, insufficient data (for example, unknown locations of viable seeds in SPA 56 at model initialisation – section 4.4.3.2) and understanding at the scale at which these processes are modelled demands decisions be made using the modeller's experience and understanding of the system. These processes of parameterisation also applied to the state-and-transition look-up table, the values of which are based on previous literature (Barbero *et al.* 1990a) and expert understanding about the system being studied (i.e. discussion with Dr. Romero Calcerrada and his colleagues). All aspects of these parameterisations are open to improvement dependent upon increased understanding and data availability at the landscape scales considered.

The 'complex' nature of the ABM/LUCC (i.e. many non-homogenous interacting agents) made testing and sensitivity analyses of this model very difficult. Testing for debugging often involved tracking the behaviour of individual agents through their spatial decision-making activities during a model replicate. At this point I realised that any analysis would need to be taken at the macro, system (i.e. landscape) level as analysis at the micro (i.e. decision-making actor) level would be too demanding in terms of time, data and computation. Later, I came to understand this approach as being 'generative' (Brown *et al.* 2006). I did consider analysis examining individual agents' life-histories and decisions, but decided this approach would be uneconomical (trading off the resources available with the likely knowledge gained). The system-level sensitivity analysis indicated that market conditions were the predominant factor driving decisions and LUCC (section 5.5). Neighbourhood effects between agents were observed (influencing agent attitudes) and land-tenure was also found to influence decision-making as the model specified. I didn't deem any more detailed, rigorous analysis worthwhile at this point in the modelling process given my uncertainty about how accurately the model represented the real-world system. Many of the parameter values used in the ABM/LUCC had been enumerated out of necessity and common sense – more colloquially, they felt like 'fudge factors' (i.e. values were used that ensured 'realistic' model behaviour but were not based on proven empirical fact). Furthermore, representation of the market forces driving LUCC, though found to be the most important, wasn't explicit or dynamic enough – but 'it would have to do for now'. This uncertainty was an important factor leading to the idea of engaging with local stakeholders to assess the model.

My experiences during this model development phase have led me to appreciate fully the suggestions of Matthews and Selman (2006 p.208) that "at their current level of development, ABMs are probably more useful as tools to *explore* options for effecting change in landscapes and rural communities, rather than *predicting* them, and as such, it is important that the structure of the models and the assumptions incorporated into them are transparent, and therefore well-documented, and also that the mechanistic behaviours assumed for the agents are well grounded in actual behaviour patterns". This passage serves to summarise my experiences and understanding developed through the model development, the result of which, in turn, is the final section of this thesis (i.e. chapters seven, eight and nine). What I felt I needed to do was to go to talk with people that might be able to indicate where and why my model was inadequate, but also to assess whether in this state it could be of any use.

Alongside the idea that I might be able to use local knowledge to evaluate and improve my model, I had come to think that if models are to explore rather than predict, many of the established, laboratory-type scientific methods for establishing the veracity of the knowledge produced by models and theories (e.g. the hypothetic-deductive method) become unavailable. In this case, I concluded, alternatives needed to be sought. The thinking (as presented in chapter seven) proceeds as follows. Socio-ecological systems, like the one I was trying to study (i.e. LUCC in SPA 56) are 'open', middle-numbered systems. Open systems are those in which mass and energy (and when humans are considered, information, meaning and value) flows both into and out of them, placing them in a state of disequilibrium (section 7.2.1). Middle-numbered systems have many components whose interactions are heterogeneous, and therefore cannot be studied easily using the methods of Cartesian science (e.g. calculus) or thermodynamics (e.g. statistical mechanics). This nature raises the epistemological problems of equifinality (i.e. there are multiple logical model structures that are able to reproduce empirically observed system behaviour – Beven 2002) and the potential of committing the logical fallacy of 'affirming the consequent' (i.e. rejecting a model if it does not reproduce the observed data, or accepting it if it does – Oreskes *et al.* 1994 – section 7.2.2). Alongside these epistemological problems, through attendance of seminars, wider reading and talking with other geographers in the department at KCL, I had become aware of the recent attitudes toward the issues facing contemporary society and the environment (e.g. Beck 1992, Funtowicz and Ravetz 1993 – section 7.3).

Thus, I have suggested a shift in emphasis in model 'validation' (i.e. evaluation) away from establishing the truth of the model's structure via mimetic accuracy and toward ensuring trust in the model's results via practical adequacy (section 7.3.1). These two criteria (trust and practical adequacy) would be useful alongside the necessity to ensure a model structure is based on sound logical and factual basis and possesses a realistic degree of mimetic accuracy. In this way validation becomes centred on the model user(s) and uses rather than the model, and suggests a shift away from falsification and deduction toward more reflexive approaches. For models of socio-ecological systems these criteria will likely be more useful than establishing the factual accuracy of a model structure or its results, but should be considered as additions to, rather than substitutions for, the criteria more suited to laboratory-based experiments. In turn, these additional criteria mean recent issues regarding expertise (e.g. Collins and Evans 2002), public engagement (Jackson *et al.* 2005) and the democratisation of science (Wilsdon and Willis 2004) become relevant (section 7.3.2).

The argument presented in chapter seven arose partly out of my problems of formally justifying the simulation model I was developing, due largely to insufficient data. But further, the sheer complexity (i.e. open, middle-numbered nature) of the socio-ecological system under study left me realising there was no way the model, given the timeframe and resources available, would ever be able to precisely match observed patterns and trends. Upon exploring the underlying problems it was realised that this was not simply an issue of data and other resources, but rather was an epistemological problem that had been previously explored and debated, particularly within the UK geography literature (e.g. Lane 2001, Brown 2004, Lane *et al.* 2006). As such, I felt that with regard to my experiences constructing this landscape-level SESM I might be able to contribute to the discussion. By this point I had already been confronted by much literature from outside my previous undergraduate and graduate training (e.g. section 5.2) and felt willing to challenge myself to ensure a truly interdisciplinary study. Alongside the epistemological issues, the more socially-oriented issues of 'expertise' and 'public engagement' were also evidently related. The context of an interdisciplinary socio-ecological modelling study provided a good opportunity to test the possibility and utility of 'participatory approaches' in modelling (e.g. Matthews and Selman 2006). With this in mind preparations for a stakeholder model assessment exercise were made (as presented in chapter eight).

9.3.5 Stakeholder Evaluation and Model Use

By October 2006 SPASIMv1 was in what was deemed to be a useable state, worthy of the time needed to be examined and evaluated by local stakeholders in the study area. In November 2006 seven interviews undertaken in SPA 56 with local stakeholders from within the study area, each of whom had knowledge of specific regions of the study area due to their occupation. Interviewees covered a range of institutional contexts from private, individual land owners with no governmental connections, through to the head of one of the subsections of the Autonomous Community of Madrid's department of environment. The semi-structured interviews considered five aspects of the simulation model: drivers of change, (section 8.3.2) model results (section 8.3.3), model assumptions (section 8.3.3), model modification (section 8.3.4), and changes in understanding (section 8.3.5). Using the results of these interviews the model was assessed against the four validation criteria established in chapter seven (accuracy of logical structure, mimetic accuracy, trust and practical adequacy – section 8.4.1). Stakeholders generally accepted the model structure and output were representative decision-making processes and anticipated future change respectively. However, several shortcomings were highlighted by the stakeholders – most importantly the absence of consideration of urban change and an inadequate representation of market mechanisms and subsidies. Furthermore, it was suggested that the emphasis placed on the spatial nature of land-use decision making (arising from my reading of the literature and discussion with other 'experts') was not justified.

Regarding the potential uses of the model, stakeholders were somewhat divided in their opinions. Those with roles in local planning were enthusiastic about the potential use of the model and said they would be willing to work with it. In contrast, the somewhat fatalistic outlook of those directly concerned with agriculture and whose decisions are based on an individual (single farm) basis led to a more sceptical response about the possible uses of the model. Thus, in some quarters the model gained a degree of trust and was deemed to have practical adequacy, whilst in others this was less evident. However, as suggested in section 8.4.1, trust and practical adequacy will only really be proven once people use the model (or more specifically, knowledge gained from it) to make decisions and/or policy. According to stakeholders' comments, and when compared against the four model validation criteria, the model was not a complete failure but had clear room for improvement in future model iterations. Much of this improvement will be based on comments made by the stakeholders, ensuring that their engagement becomes an integral component of the modelling process.

Whilst the stakeholder validation exercise proved useful, two particular shortcomings were evident. First, some stakeholders seemingly found it difficult to make the distinction between the model and the scenarios it was run for. The format of the interviews did not allow the degree to which this was a problem to be established, but any future contact between simulation models and non-modellers will need to consider this issue. Second, as noted above, several of the stakeholders were fatalistic about the change they expected to occur over the coming decades and saw little use of the model to explore alternative scenarios. This attitude was apparent despite accepting the model was generally an appropriate representation of agricultural decision-making and LUCC as it is occurring in SPA 56. However, the model did provide a useful platform from which discussion about potential LUCC was launched and the interviews indicated that the use of the model as a 'mediator' (Morrison and Morgan 1999) or 'muse' will be useful for generating discussion about future environmental policy- and decision-making if so required.

Personally, the stakeholder engagement experience, both during model development (i.e. chapter five) and later during evaluation (i.e. chapter eight), was very useful for conceptualising and understanding the processes of LUCC. Given that the agents to be represented in any simulation model of agricultural decision-making or LUCC will be actual humans able to impart knowledge about their actions, it seems necessary that all models of this type engage with local stakeholders during the model development processes, even if they are not engaged for the model evaluation phase (and as advocated will be good research practice in the future by Moss and Edmonds 2005, Matthews 2006). Whilst there are drawbacks to this approach (as highlighted above and in chapter eight) the benefits of improved system understanding and the potential for ensuring the development of a practically adequate model (if that is required) outweigh them.

Given the modelling experiences I have gained during the completion of the research presented in this thesis I would suggest that currently ABM/LUCC approaches will be useful in two distinctly contrasting situations. First, the agent-based approach will be useful for examination of essential system-level understanding of the processes of LUCC, when used in a 'metaphor model' approach (section 7.2.1 – Perry and Millington 2007). In this situation, representation of a specific real-world system or place will not be the aim – instead the multi-agent system itself will be the object of enquiry. This approach would contribute to the descriptive modelling at a low level of representation that Moss and Edmonds (2005) suggest will be necessary before more general theory can be developed. Alternatively, if explicit representation of LUCC in the context of a specific place is intended, the modeller(s) will need to ensure that iterative engagement with the actors being represented is possible from the outset and throughout the modelling process. This engagement will be vital to ensure models are based on the best available evidence and understanding. Social enquiry will be as important as systems enquiry in these modelling projects (e.g. Oxley and Lemon 2003). Furthermore, this approach will demand detailed data for parameterisation, time for development and the services of a skilled and experienced object-oriented programmer (in contrast to the initial situation in this modelling thesis). With contemporary levels of widely available computing power as they are, a less computationally intensive modelling approach would be more suited to ‘decision-support’ regarding the environmental impacts of LUCC (Oxley *et al.* 2004). This less intensive approach might be achieved by taking a systems dynamics approach (instead of an agent-based approach) or by vastly simplifying agent representation, and would allow stakeholders to ‘use’ and interact with the model (interface at least) themselves.

Having established a degree of confidence in SPASIMv1, it was now (i.e. December 2006 – February 2007) used with scenarios of economic and demographic change to simulate interactions between LUCC and wildfire regimes (section 6.3). Results of simulations not considering human activity, but over longer temporal extents (i.e. centuries rather than decades), were also examined (section 6.4 and 6.5). To characterise wildfire regimes for comparison between effects of different scenarios and parameters, the power-law wildfire frequency-area scaling exponent β was proposed (section 6.2). Other state variables including mean total burned area, land-cover composition and various land-cover spatial configuration metrics were used to examine the results of LUCC-wildfire interactions. Simulation results indicted that mean largest wildfire and mean total burned area will increase if agricultural activity declines and that changes in land-cover composition are driven more by human activity than wildfire. The implication of these results is that wildfire and environmental managers, both locally to SPA 56 and across other regions of the Mediterranean experiencing agricultural decline and abandonment, will need to consider social change in their wildfire and environmental management plans. Landscape configuration varied little across the scenarios of human activity (section 6.3), due to the importance of small-scale human activity on LUCC.

The investigation of the use of β (a measure of the ratio of large to medium to small events) stemmed from my previous work (Millington 2003) that was developed subsequently and concurrently with the work in this thesis (resulting in Malamud *et al.* 2005, Millington *et al.* 2006). For scenarios of human activity the power-law frequency-area scaling exponent β was not found to vary significantly in response to human activity. In contrast, simulations over longer periods did indicate variation in β values as a function of total and maximum land-cover flammability probabilities (section 6.5). These results indicate that the power-law scaling exponent β is not the most useful measure of the wildfire frequency-area relationship over smaller, regional (i.e. 1×10^3 km^2) extents. Future investigation into the β scaling exponent and its implications should remain at larger spatial and temporal extents.

One of the most novel aspects of SPASIMv1, afforded by the integration of an ABM/LUCC with a LFSM, is the attempt to represent explicitly the impact of human activity on wildfire ignition frequency and location (section 4.5.2). Whilst it is known that the majority of fires in Spain are ignited by human activity (Moreno *et al.* 1998), there are limited data and understanding regarding the reasons people start fires or the locations in which fires occur. The model was therefore based on scant previous research (e.g. Chuvieco and Salas 1996) and 'expert knowledge', which in this case meant 'common sense' assumptions (such as fire ignition frequency is likely to be greater nearer areas of greater outdoor human activity). The difficulties in establishing formal justifications for the modelling approach taken to represent humans as a source of wildfire ignition highlight the need for more (sociological) research in this area. However, such research is likely to be contentious and unbiased evidence may be difficult to come by (e.g. Millington 2006 and associated comments – but see discussion in section 9.3.9 below). The explicit simulation of wildfire ignition cause allows the behaviour of wildfire due to these different causes to be examined. Fires ignited by human causes were found to burn greater areas of scrub than would be expected at random, and lightning fires burned greater areas of forest than would be expected at random. As other authors have found (e.g. Mouillot *et al.* 2005) this suggests some areas will be burned more frequently than others due to human activity (this follows from the model assumptions that human ignitions will be more frequent where greater levels of outdoor human activity are present). No effect on the wildfire regime was found due to 'types' of people in the landscape (section 4.5.2). Further work needs to examine the human wildfire ignition representation of the model (especially a more formal justification of the assumptions) and its potential results. Furthermore, whilst not evaluated in the stakeholder model evaluation, the interaction and feedback between land burning and the human response is not well integrated currently (section 6.6). With the model assumptions in mind, and given the empirical evidence found elsewhere, these results suggest wildfire managers will need to locate their energies on areas of increased human activity.

9.3.6 Self-reflection

This chapter has attempted to do two things. First is has tried to summarise what this thesis has told us and reflect on not only what was learnt but how it was learnt. Lane *et al.* (2006 p.251) recently suggested that reflexivity often leads modellers to question "our assumptions about what the problems are that are being modelled". One of the additional model validation criteria suggested in chapter seven was that of 'practical adequacy' – ensuring that model structure allows the examination of problems the model user(s) wants to examine. As such, the work in chapter eight found that the model developed here was only partially practically adequate and that some of the key concerns for stakeholders in SPA 56 were not represented by the model, the primary omission being urban change and the pressures of a nearby growing capital city. Whilst this might be interpreted as a failure of the model, when taken as part of an iterative process of model development this is but another stage in the ongoing modelling process, at all points along which both modeller and model user can learn. Furthermore, had stakeholder engagement been even further 'upstream' and more iterative, this problem would be diminished. The thesis may end here but the modelling and learning should continue in the future.

Second, this chapter has presented events as a narrative. The chronological orderings of events a narrative demands has highlighted areas of this research that did not occur in the order presented in the main body of the thesis (e.g. SPASIMv1 results – chapter six). Thus, the narrative presented here has illustrated the route to discovery as well as describing what has been learnt. For example, the causes of realisation that an agent-based approach was required to ensure a cutting-edge model of socio-ecological change and wildfire were discussed, and the path leading to engagement with non-modellers for model evaluation was mapped. This narrative approach to 'open up' the modelling process may not be so relevant in an academic setting where work is read and reviewed by other modellers (and who therefore understand the subtleties and vagaries of the modelling process), but it will be important for engaging models with non-modellers. One tool that may help modellers to construct and disseminate these narratives to non-modellers in the future (at least in more developed countries) is that of the weblog. Weblogs have grown in recent years as informal platforms for dissemination of material or media on the World Wide Web. Generally these user-publishing platforms are unedited and not subject to any form of peer review *before* publication, and therefore do not match the formal requirements of academic or scientific research for publication (thus comments such as those cited above – section 9.3.5 – should be treated with caution). However, given the diary-style, (reverse) chronological nature of their presentation, such a platform may provide a means to by 'open-up' the modelling process and, particularly, be a useful tool for generating model narratives. Thought processes underlying assumptions and the dead ends of research might be revealed if a rigorous log is kept. This log might just as easily be a traditional laboratory log book as a weblog, but a particular advantage of the weblog over more traditional research diaries is that it is freely available on the web and therefore will be able to aid public engagement, public understanding and the democratisation of scientific models and the process of modelling (as recent policy research has suggested is required, e.g. Wilsdon and Willis 2004). In this manner project stakeholders can keep up-to-date with the thoughts and work of the modeller(s) and the iterative nature of stakeholder engagement can become even more integrated. Whether issues will then arise then about some kind of 'over engagement' will remain to be seen. Regardless of the tool used to record the narrative of the modelling process, hopefully the attempt as it has been presented here indicates some promise for this form of model dissemination.

9.4 SUMMARY

This thesis aimed to examine the interactions between human land use/cover change (LUCC) and wildfire regimes in a Mediterranean landscape and to explore and evaluate novel methods to 'validate' simulation models (and processes of modelling) of environmental change considering human activity. Chapter two introduced the research context and highlighted the importance of both human activity and wildfire in Mediterranean-type ecosystems. The literature review also showed how ongoing socio-economic change is driving agricultural LUCC (predominantly abandonment), with potential impacts for wildfire and vegetation dynamics. Chapter three examined the alternatives for modelling LUCC in Mediterranean-type landscapes such as SPA 56, and presented a regression-based modelling approach that demonstrated the shortcomings of empirical modelling. These shortcomings included issues such as non-stationarity in the driving forces of change, reconciliation of different 'types' of data that are aggregated at different resolutions (i.e. biophysical versus socio-economic data), and the limited process understanding an empirical approach provides. Experiences from this empirical modelling indicated that a simulation modelling approach would best serve the first aim of the thesis.

Chapters four and five presented the structure of the simulation model. A time-based, state-and–transition vegetation-dynamics model with a wildfire cellular automata component (a Landscape Fire Succession Model) was integrated with an agent-based model of agricultural decision-making (an ABM/LUCC). The integrated socio-ecological simulation model (SESM) was a novel approach to directly consider the influence of human activity on wildfire ignition frequency and location. One of the main findings from the use of this model was that if agricultural change (i.e. abandonment) continues to follow recent trends, the risk of large fires will increase and a greater total area will be burned. Such a finding implies that future wildfire management and planning will need to consider the nature of contemporary land use change. The SESM also suggested that there will be biases in the areas of land-cover burned according to ignition cause (i.e. whether caused by human agents or otherwise). However, the experience from this modelling work shows that explicitly representing human fire risk in simulation models is currently challenging because of poor understanding regarding the causes of ignition and locations of wildfire, due to factors such as arson. Increased empirical research on the human causes and locations of fire ignition would aid and improve future simulation modelling of this type. The use of the model for both scenarios that considered human activity and those that did not revealed limitations in the use of the β power-law scaling exponent in the fire frequency-area relationship that has been suggested as a useful measure by which to compare and contrast wildfire regimes. Results suggested that β is not a useful measure by which to contrast wildfire regimes at the shorter time scales that were considered over the regional extent. However, β did become useful over longer time scales when human activity was not considered in model scenarios. The use of the LFSM to examine influences on wildfire frequency-area scaling and pattern-process feedbacks (independent of human activity) is an area that provides potential contributions to understanding these aspects of wildfire regimes.

The experiences of the work presented in chapter five showed the resource intensiveness of this agent-based modelling approach at the landscape level (i.e. 1×10^3 km^2 over decades). Furthermore, these experiences lead to the recommendation that ABM/LUCC approaches will be useful in two distinctly contrasting situations. First, an agent-based approach will be useful for examining essential system-level understanding of the processes of LUCC independent of a specific real world study area. In this case the model system itself becomes the object of study (rather than the real world system), contributing to the development of more general theory via low-level descriptive modelling. Alternatively, if the aim of the model is to represent explicitly a specific landscape in the real world, modeller(s) will need to ensure that iterative engagement with the relevant actors is possible throughout the modelling process. In this case the understanding gained by both modeller(s) and stakeholders from the modelling arises as much out of the process of model building as its application.

Chapter seven discussed the problems of modelling open, middle numbered systems and the associated issues of model 'validation' and assessment. The problems of equifinality (presence of multiple logical structures able to reproduce empirical behaviour) and the consequent logical fallacy of (incorrectly) accepting a model as valid if it reproduces observed data, mean that deduction and falsification are not useful methods for model evaluation. As a response, when simulating socio-ecological systems the model validation criteria of trust and practical adequacy were suggested as useful additions to the more traditional structural and mimetic accuracy criteria. In the case of modelling LUCC in a real world landscape it was suggested that these added evaluation criteria could be applied by local stakeholders, a suggestion which was tested for the SESM developed in this thesis in chapter eight. Whilst the approach taken in this thesis did not allow comprehensive participation throughout the project, when non-modellers were engaged with the model several potential drawbacks were found. Whilst the model structure was broadly accepted as adequate several shortcomings were identified, including the lack of representation of urban change and an emphasis on spatial configuration in agricultural decision-making. Subsequently, enthusiasm regarding the potential uses of the model varied according to the institutional setting of the stakeholder, who tended to be more positive if involved directly in the planning process. The insights provided by the stakeholder model evaluation suggest there is a need to engage with local stakeholders at all stages throughout the iterative modelling process. It is important to highlight that post-modelling stakeholder consultation undertaken here was not as useful as fully integrated participatory modelling might have been. Not involving participants throughout the process, and the engagement of an integrated simulation model with non-modellers, highlighted potential problems during this engagement process. Specifically, several participants conflated scenarios the model was run for with the actual model itself and fatalistic attitudes toward change combined with limited participant-modeller interaction prevented full understanding of the model structure and behaviour. The difficulties of engaging non-modellers with models and the potential learning benefits of participatory modelling are highlighted by this research as areas for potential future investigation.

Finally, this concluding chapter has presented reflections on the modelling process in a chronological, narrative fashion. The narrative has highlighted the vagaries of the modelling process and demonstrated a reflexive approach that this thesis has suggested would be more informative than falsification and deduction for the evaluation of models of socio-ecological systems. Greater use of reflexive approaches and engagement with key actors in LUCC will ensure improved construction, assessment and utility of these integrated models, but at the same time raises many methodological issues.

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APPENDIX I

Look-up table of pixel physical attributes to define *D∆* and *T∆* in the rule-based community level model of Mediterranean vegetation-dynamics. Use of this table by SPASIMv1 is detailed in section 4.4. Codes for each column are defined at the bottom the table.

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
1	1	0	1	0	0	0	1	0
1	1	0	1	0	0	1	1	0
1	1	0	1	0	0	2	1	0
1	1	1	1	0	0	0	1	0
1	1	1	1	0	0	1	1	0
1	1	1	1	0	0	2	1	0
1	1	0	1	0	1	0	1	0
1	1	0	1	0	1	1	1	0
1	1	1	1	0	1	0	1	0
1	1	1	1	0	1	1	1	0
1	1	0	1	1	0	2	1	0
1	1	1	1	1	0	0	1	0
1	1	1	1	1	0	2	1	0
1	1	1	1	1	1	0	1	0
1	1	0	0	1	0	0	2	40
1	1	1	0	1	0	1	2	40
1	1	0	0	1	0	1	2	30
1	1	0	0	1	1	0	2	40
1	1	1	0	1	1	1	2	40
1	1	0	0	1	1	1	2	30
1	1	0	1	1	0	0	2	40
1	1	1	1	1	0	1	2	40
1	1	0	1	1	0	1	2	30
1	1	0	1	1	1	0	2	40
1	1	1	1	1	1	1	2	40
1	1	0	1	1	1	1	2	30
1	1	0	0	0	1	2	4	20
1	1	1	0	0	1	2	4	20
1	1	0	0	0	1	1	4	20
1	1	0	0	1	1	2	4	20
1	1	1	0	1	1	2	4	20
1	1	0	1	0	1	2	4	20
1	1	1	1	0	1	2	4	20
1	1	0	1	1	1	2	4	20
1	1	1	1	1	1	2	4	20
1	0	1	1	0	1	0	1	0
1	0	1	1	0	0	0	1	0
1	0	1	1	1	1	0	1	0
1	0	1	1	1	0	0	1	0
1	0	0	0	0	1	0	2	20
1	0	0	0	0	0	0	2	20
1	0	0	0	0	1	1	2	15
1	0	0	0	0	0	1	2	15
1	0	0	0	0	0	2	2	20
1	0	0	0	1	1	0	2	20
1	0	0	0	1	0	0	2	20
1	0	0	0	1	1	1	2	15
1	0	0	0	1	0	1	2	15

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
1	0	0	0	1	0	2	2	15
1	0	0	1	0	1	0	2	20
1	0	0	1	0	0	0	2	20
1	0	0	1	0	1	1	2	15
1	0	0	1	0	0	1	2	15
1	0	0	1	0	0	2	2	15
1	0	0	1	1	1	0	2	20
1	0	0	1	1	0	0	2	20
1	0	0	1	1	1	1	2	15
1	0	0	1	1	0	1	2	15
1	0	0	1	1	0	2	2	20
1	0	1	0	0	1	1	2	20
1	0	1	0	0	0	1	2	20
1	0	1	0	0	0	2	2	20
1	0	1	0	0	1	2	2	15
1	0	1	0	1	1	1	2	20
1	0	1	0	1	0	1	2	20
1	0	1	0	1	0	2	2	20
1	0	1	0	1	1	2	2	15
1	0	1	1	0	1	1	2	20
1	0	1	1	0	0	1	2	20
1	0	1	1	0	0	2	2	20
1	0	1	1	0	1	2	2	15
1	0	1	1	1	1	1	2	20
1	0	1	1	1	0	1	2	20
1	0	1	1	1	0	2	2	20
1	0	1	1	1	1	2	2	15
1	0	0	0	0	1	2	4	20
1	0	0	0	1	1	2	4	20
1	0	0	1	0	1	2	4	20
1	0	0	1	1	1	2	4	20
2	1	0	0	0	0	0	1	20
2	1	1	0	0	0	0	1	20
2	1	1	0	0	1	0	1	20
2	1	1	0	1	0	0	1	20
2	1	1	0	1	1	0	1	20
2	1	1	1	0	0	0	1	20
2	1	1	1	0	1	0	1	20
2	1	0	1	0	0	0	1	25
2	1	0	1	0	1	0	1	25
2	1	1	1	1	0	0	1	20
2	1	1	1	1	1	0	1	20
2	1	1	0	0	0	1	1	25
2	1	0	0	0	0	1	2	0
2	1	0	0	0	0	2	2	0
2	1	1	0	0	0	2	2	0
2	1	0	0	0	1	0	1	20
2	1	0	0	1	0	0	6	20
2	1	0	0	1	1	0	6	20
2	1	0	1	1	0	0	6	20
2	1	0	1	1	1	0	6	20
2	1	0	0	0	1	1	4	25
2	1	1	0	0	1	1	1	30
2	1	0	0	1	0	1	6	25
2	1	0	0	1	1	1	6	25
2	1	1	0	1	0	1	6	30
2	1	1	0	1	1	1	6	30
2	1	0	1	0	0	1	1	25
2	1	0	1	0	1	1	1	25
2	1	1	1	0	0	1	1	30
2	1	1	1	0	1	1	1	30

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
2	1	0	1	1	1	1	6	25
2	1	1	1	1	0	1	6	30
2	1	0	1	1	0	1	6	25
2	1	1	1	1	1	1	6	30
2	1	1	0	1	0	2	6	40
2	1	0	0	1	0	2	6	50
2	1	1	1	0	0	2	1	30
2	1	0	1	0	0	2	1	30
2	1	1	1	1	0	2	1	30
2	1	0	1	1	0	2	1	30
2	1	0	0	0	1	2	4	20
2	1	1	0	0	1	2	4	20
2	1	0	0	1	1	2	4	20
2	1	1	0	1	1	2	4	20
2	1	0	1	0	1	2	4	20
2	1	1	1	0	1	2	4	20
2	1	0	1	1	1	2	4	20
2	1	1	1	1	1	2	4	20
2	0	1	0	0	1	0	2	0
2	0	1	0	0	0	0	2	0
2	0	1	0	1	1	0	2	0
2	0	1	0	1	0	0	2	0
2	0	1	1	0	1	0	1	20
2	0	1	1	0	0	0	1	20
2	0	1	1	1	1	0	1	20
2	0	1	1	1	0	0	1	20
2	0	0	0	0	1	0	6	40
2	0	0	0	0	0	0	6	40
2	0	0	0	1	1	0	6	40
2	0	0	0	1	0	0	6	40
2	0	0	1	0	1	0	6	45
2	0	0	1	0	0	0	6	45
2	0	0	1	1	1	0	6	40
2	0	0	1	1	0	0	6	40
2	0	1	0	0	1	1	6	35
2	0	1	0	0	0	1	6	35
2	0	0	0	0	1	1	6	25
2	0	0	0	0	0	1	6	25
2	0	1	0	1	1	1	6	30
2	0	1	0	1	0	1	6	30
2	0	0	0	1	1	1	6	20
2	0	0	0	1	0	1	6	20
2	0	1	1	0	1	1	6	35
2	0	1	1	0	0	1	6	35
2	0	0	1	0	1	1	6	25
2	0	0	1	0	0	1	6	25
2	0	1	1	1	1	1	6	35
2	0	1	1	1	0	1	6	35
2	0	0	1	1	1	1	6	25
2	0	0	1	1	0	1	6	25
2	0	0	0	0	0	2	6	40
2	0	1	0	0	0	2	6	30
2	0	0	0	1	0	2	6	40
2	0	1	0	1	0	2	6	30
2	0	0	1	0	0	2	6	45
2	0	1	1	0	0	2	6	35
2	0	0	1	1	0	2	6	40
2	0	1	1	1	0	2	6	30
2	0	1	0	0	1	2	4	20
2	0	0	0	0	1	2	4	20
2	0	1	0	1	1	2	4	20

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
2	0	0	0	1	1	2	4	20
2	0	1	1	0	1	2	4	20
2	0	0	1	0	1	2	4	20
2	0	1	1	1	1	2	4	20
2	0	0	1	1	1	2	4	20
3	1	0	0	0	0	0	5	3
3	1	0	0	0	1	0	5	3
3	1	0	0	1	0	0	5	3
3	1	0	0	1	1	0	5	3
3	1	0	1	0	0	0	5	3
3	1	0	1	0	1	0	5	3
3	1	0	1	1	0	0	5	3
3	1	0	1	1	1	0	5	3
3	1	1	0	0	0	0	5	3
3	1	1	0	0	1	0	5	3
3	1	1	0	1	0	0	5	3
3	1	1	0	1	1	0	5	3
3	1	1	1	0	0	0	5	3
3	1	1	1	0	1	0	5	3
3	1	1	1	1	0	0	5	3
3	1	1	1	1	1	0	5	3
3	1	0	0	0	0	1	5	3
3	1	0	0	0	1	1	5	3
3	1	0	0	1	0	1	5	3
3	1	0	0	1	1	1	5	3
3	1	0	1	0	0	1	5	3
3	1	0	1	0	1	1	5	3
3	1	0	1	1	1	1	5	3
3	1	0	1	1	0	1	5	3
3	1	1	0	0	0	1	5	3
3	1	1	0	0	1	1	5	3
3	1	1	0	1	0	1	5	3
3	1	1	0	1	1	1	5	3
3	1	1	1	0	0	1	5	3
3	1	1	1	0	1	1	5	3
3	1	1	1	1	0	1	5	3
3	1	1	1	1	1	1	5	3
3	1	0	0	0	0	2	5	3
3	1	0	0	1	0	2	5	3
3	1	0	0	0	1	2	5	3
3	1	0	0	1	1	2	5	3
3	1	0	1	0	0	2	5	3
3	1	0	1	1	0	2	5	3
3	1	0	1	0	1	2	5	3
3	1	0	1	1	1	2	5	3
3	1	1	0	0	0	2	5	3
3	1	1	0	1	0	2	5	3
3	1	1	0	0	1	2	5	3
3	1	1	0	1	1	2	5	3
3	1	1	1	0	0	2	5	3
3	1	1	1	1	0	2	5	3
3	1	1	1	0	1	2	5	3
3	1	1	1	1	1	2	5	3
3	0	0	0	0	1	0	5	3
3	0	0	0	0	0	0	5	3
3	0	0	0	1	1	0	5	3
3	0	0	0	1	0	0	5	3
3	0	0	1	0	1	0	5	3
3	0	0	1	0	0	0	5	3
3	0	0	1	1	1	0	5	3
3	0	0	1	1	0	0	5	3

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
3	0	1	0	0	1	0	5	3
3	0	1	0	0	0	0	5	3
3	0	1	0	1	1	0	5	3
3	0	1	0	1	0	0	5	3
3	0	1	1	0	1	0	5	3
3	0	1	1	0	0	0	5	3
3	0	1	1	1	1	0	5	3
3	0	1	1	1	0	0	5	3
3	0	0	0	0	1	1	5	3
3	0	0	0	0	0	1	5	3
3	0	0	0	1	1	1	5	3
3	0	0	0	1	0	1	5	3
3	0	0	1	0	1	1	5	3
3	0	0	1	0	0	1	5	3
3	0	0	1	1	1	1	5	3
3	0	0	1	1	0	1	5	3
3	0	1	0	0	1	1	5	3
3	0	1	0	0	0	1	5	3
3	0	1	0	1	1	1	5	3
3	0	1	0	1	0	1	5	3
3	0	1	1	0	1	1	5	3
3	0	1	1	0	0	1	5	3
3	0	1	1	1	1	1	5	3
3	0	1	1	1	0	1	5	3
3	0	0	0	0	0	2	5	3
3	0	0	0	1	0	2	5	3
3	0	0	0	0	1	2	5	3
3	0	0	0	1	1	2	5	3
3	0	0	1	0	0	2	5	3
3	0	0	1	1	0	2	5	3
3	0	0	1	0	1	2	5	3
3	0	0	1	1	1	2	5	3
3	0	1	0	0	0	2	5	3
3	0	1	0	1	0	2	5	3
3	0	1	0	0	1	2	5	3
3	0	1	0	1	1	2	5	3
3	0	1	1	0	0	2	5	3
3	0	1	1	1	0	2	5	3
3	0	1	1	0	1	2	5	3
3	0	1	1	1	1	2	5	3
4	1	0	1	0	0	0	1	25
4	1	0	1	0	1	0	1	25
4	1	0	1	1	0	0	1	25
4	1	0	1	1	1	0	1	25
4	1	1	1	0	0	0	1	20
4	1	1	1	0	1	0	1	20
4	1	1	1	1	0	0	1	20
4	1	1	1	1	1	0	1	20
4	1	1	1	0	0	1	1	30
4	1	1	1	1	0	1	1	30
4	1	0	0	1	0	0	2	35
4	1	1	0	1	0	0	2	40
4	1	1	0	1	0	1	2	30
4	1	0	0	0	0	0	4	0
4	1	0	0	0	1	0	4	0
4	1	0	0	1	1	0	4	0
4	1	1	0	0	0	0	4	0
4	1	1	0	0	1	0	4	0
4	1	1	0	1	1	0	4	0
4	1	0	0	0	0	1	4	0
4	1	0	0	0	1	1	4	0

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
4	1	0	0	1	0	1	4	0
4	1	0	0	1	1	1	4	0
4	1	0	1	0	0	1	4	0
4	1	0	1	0	1	1	4	0
4	1	0	1	1	0	1	4	0
4	1	0	1	1	1	1	4	0
4	1	1	0	0	0	1	4	0
4	1	1	0	0	1	1	4	0
4	1	1	0	1	1	1	4	0
4	1	1	1	0	1	1	4	0
4	1	1	1	1	1	1	4	0
4	1	0	0	0	0	2	4	0
4	1	0	0	0	1	2	4	0
4	1	0	0	1	0	2	4	0
4	1	0	0	1	1	2	4	0
4	1	0	1	0	0	2	4	0
4	1	0	1	0	1	2	4	0
4	1	0	1	1	0	2	4	0
4	1	0	1	1	1	2	4	0
4	1	1	0	0	0	2	4	0
4	1	1	0	0	1	2	4	0
4	1	1	0	1	0	2	4	0
4	1	1	0	1	1	2	4	0
4	1	1	1	0	0	2	4	0
4	1	1	1	0	1	2	4	0
4	1	1	1	1	0	2	4	0
4	1	1	1	1	1	2	4	0
4	0	0	1	0	0	0	1	25
4	0	0	1	0	1	0	1	25
4	0	0	1	1	0	0	1	25
4	0	0	1	1	1	0	1	25
4	0	1	1	0	0	0	1	20
4	0	1	1	0	1	0	1	20
4	0	1	1	1	0	0	1	20
4	0	1	1	1	1	0	1	20
4	0	0	0	0	0	0	2	30
4	0	0	0	0	1	0	2	30
4	0	0	0	1	0	0	2	30
4	0	0	0	1	1	0	2	30
4	0	1	0	0	0	0	2	35
4	0	1	0	0	1	0	2	35
4	0	1	0	1	0	0	2	35
4	0	1	0	1	1	0	2	35
4	0	0	0	0	0	1	2	40
4	0	0	0	1	0	1	2	40
4	0	0	1	0	0	1	2	40
4	0	0	1	1	0	1	2	40
4	0	1	0	0	0	1	2	35
4	0	1	0	0	1	1	2	35
4	0	1	0	1	0	1	2	35
4	0	1	0	1	1	1	2	35
4	0	1	1	0	0	1	2	35
4	0	1	1	0	1	1	2	35
4	0	1	1	1	0	1	2	35
4	0	1	1	1	1	1	2	35
4	0	0	0	0	1	1	4	0
4	0	0	0	1	1	1	4	0
4	0	0	1	0	1	1	4	0
4	0	0	1	1	1	1	4	0
4	0	0	0	0	0	2	4	0
4	0	0	0	0	1	2	4	0

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
4	0	0	0	1	0	2	4	0
4	0	0	0	1	1	2	4	0
4	0	0	1	0	0	2	4	0
4	0	0	1	0	1	2	4	0
4	0	0	1	1	0	2	4	0
4	0	0	1	1	1	2	4	0
4	0	1	0	0	0	2	4	0
4	0	1	0	0	1	2	4	0
4	0	1	0	1	0	2	4	0
4	0	1	0	1	1	2	4	0
4	0	1	1	0	0	2	4	0
4	0	1	1	0	1	2	4	0
4	0	1	1	1	0	2	4	0
4	0	1	1	1	1	2	4	0
5	1	0	1	0	0	0	1	10
5	1	0	1	0	1	0	1	10
5	1	0	1	1	0	0	1	10
5	1	0	1	1	1	0	1	10
5	1	1	1	0	0	0	1	10
5	1	1	1	0	1	0	1	10
5	1	1	1	1	0	0	1	10
5	1	1	1	1	1	0	1	10
5	1	0	1	0	0	1	1	15
5	1	0	1	0	1	1	1	15
5	1	0	1	1	0	1	1	15
5	1	0	1	1	1	1	1	15
5	1	1	1	0	0	1	1	10
5	1	1	1	0	1	1	1	10
5	1	1	1	1	0	1	1	10
5	1	1	1	1	1	1	1	10
5	1	1	1	0	0	2	1	15
5	1	1	1	1	0	2	1	15
5	1	0	0	0	0	0	5	0
5	1	0	0	0	1	0	5	0
5	1	0	0	1	0	0	6	40
5	1	0	0	1	1	0	6	40
5	1	1	0	0	0	0	5	0
5	1	1	0	0	1	0	5	0
5	1	1	0	1	1	0	5	0
5	1	0	0	0	0	1	5	0
5	1	1	0	0	0	1	5	0
5	1	0	0	0	0	2	5	0
5	1	1	0	0	0	2	5	0
5	1	0	1	0	0	2	5	0
5	1	0	1	1	0	2	5	0
5	1	1	0	1	0	0	6	40
5	1	0	0	1	0	1	6	30
5	1	1	0	1	0	1	6	30
5	1	0	0	1	0	2	6	30
5	1	1	0	1	0	2	6	30
5	1	0	0	0	1	1	4	15
5	1	0	0	1	1	1	4	15
5	1	1	0	0	1	1	4	20
5	1	1	0	1	1	1	4	20
5	1	0	0	0	1	2	4	15
5	1	0	0	1	1	2	4	15
5	1	0	1	0	1	2	4	15
5	1	0	1	1	1	2	4	15
5	1	1	0	0	1	2	4	15
5	1	1	0	1	1	2	4	15
5	1	1	1	0	1	2	4	15

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
5	1	1	1	1	1	2	4	15
5	0	1	1	0	0	0	1	15
5	0	1	1	0	1	0	1	15
5	0	1	1	1	0	0	1	15
5	0	1	1	1	1	0	1	15
5	0	0	1	0	0	0	2	15
5	0	0	1	0	1	0	2	15
5	0	0	1	1	0	0	2	15
5	0	0	1	1	1	0	2	15
5	0	0	1	0	0	1	2	15
5	0	0	1	0	1	1	2	15
5	0	0	1	1	0	1	2	15
5	0	0	1	1	1	1	2	15
5	0	1	1	0	0	1	2	15
5	0	1	1	0	1	1	2	15
5	0	1	1	1	0	1	2	15
5	0	1	1	1	1	1	2	15
5	0	1	1	0	0	2	2	15
5	0	1	1	1	0	2	2	15
5	0	0	0	0	0	0	6	45
5	0	0	0	0	1	0	6	45
5	0	0	0	1	0	0	6	45
5	0	0	0	1	1	0	6	45
5	0	1	0	0	0	0	6	50
5	0	1	0	0	1	0	6	50
5	0	1	0	1	0	0	6	50
5	0	1	0	1	1	0	6	50
5	0	0	0	0	0	1	6	35
5	0	0	0	0	1	1	6	35
5	0	0	0	1	0	1	6	35
5	0	0	0	1	1	1	6	35
5	0	1	0	0	0	1	6	40
5	0	1	0	0	1	1	6	40
5	0	1	0	1	0	1	6	40
5	0	1	0	1	1	1	6	40
5	0	0	0	0	0	2	6	40
5	0	0	0	1	0	2	6	40
5	0	0	1	0	0	2	6	40
5	0	0	1	1	0	2	6	40
5	0	1	0	0	0	2	6	35
5	0	1	0	1	0	2	6	35
5	0	0	0	0	1	2	4	15
5	0	0	0	1	1	2	4	15
5	0	0	1	0	1	2	4	15
5	0	0	1	1	1	2	4	15
5	0	1	0	0	1	2	4	20
5	0	1	0	1	1	2	4	20
5	0	1	1	0	1	2	4	20
5	0	1	1	1	1	2	4	20
6	0	1	1	0	0	0	2	30
6	0	1	1	0	1	0	2	30
6	0	1	1	1	0	0	2	30
6	0	1	1	1	1	0	2	30
6	0	0	0	0	1	2	2	30
6	0	0	0	1	1	2	2	30
6	0	0	1	0	1	2	2	30
6	0	0	1	1	1	2	2	30
6	0	0	0	0	0	0	6	0
6	0	0	0	0	1	0	6	0
6	0	0	0	1	0	0	6	0
6	0	0	0	1	1	0	6	0

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
6	0	0	1	0	0	0	6	0
6	0	0	1	0	1	0	6	0
6	0	0	1	1	0	0	6	0
6	0	0	1	1	1	0	6	0
6	0	1	0	0	0	0	6	0
6	0	1	0	0	1	0	6	0
6	0	1	0	1	0	0	6	0
6	0	1	0	1	1	0	6	0
6	0	0	0	0	0	1	6	0
6	0	0	0	0	1	1	6	0
6	0	0	0	1	0	1	6	0
6	0	0	0	1	1	1	6	0
6	0	0	1	0	0	1	6	0
6	0	0	1	0	1	1	6	0
6	0	0	1	1	0	1	6	0
6	0	0	1	1	1	1	6	0
6	0	1	0	0	0	1	6	0
6	0	1	0	0	1	1	6	0
6	0	1	0	1	0	1	6	0
6	0	1	0	1	1	1	6	0
6	0	1	1	0	0	1	6	0
6	0	1	1	0	1	1	6	0
6	0	1	1	1	0	1	6	0
6	0	1	1	1	1	1	6	0
6	0	0	0	0	0	2	6	0
6	0	0	0	1	0	2	6	0
6	0	0	1	0	0	2	6	0
6	0	0	1	1	0	2	6	0
6	0	1	0	0	0	2	6	0
6	0	1	0	0	1	2	6	0
6	0	1	0	1	0	2	6	0
6	0	1	0	1	1	2	6	0
6	0	1	1	0	0	2	6	0
6	0	1	1	0	1	2	6	0
6	0	1	1	1	0	2	6	0
6	0	1	1	1	1	2	6	0
7	0	0	0	0	0	0	5	3
7	0	0	0	0	1	0	5	3
7	0	0	0	1	0	0	5	3
7	0	0	0	1	1	0	5	3
7	0	0	1	0	0	0	5	3
7	0	0	1	0	1	0	5	3
7	0	0	1	1	0	0	5	3
7	0	0	1	1	1	0	5	3
7	0	1	0	0	0	0	5	3
7	0	1	0	0	1	0	5	3
7	0	1	0	1	0	0	5	3
7	0	1	0	1	1	0	5	3
7	0	1	1	0	0	0	5	3
7	0	1	1	0	1	0	5	3
7	0	1	1	1	0	0	5	3
7	0	1	1	1	1	0	5	3
7	0	0	0	0	0	1	5	3
7	0	0	0	0	1	1	5	3
7	0	0	0	1	0	1	5	3
7	0	0	0	1	1	1	5	3
7	0	0	1	0	0	1	5	3
7	0	0	1	0	1	1	5	3
7	0	0	1	1	0	1	5	3
7	0	0	1	1	1	1	5	3
7	0	1	0	0	0	1	5	3

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	D∆	T∆
7	0	1	0	0	1	1	5	3
7	0	1	0	1	0	1	5	3
7	0	1	0	1	1	1	5	3
7	0	1	1	0	0	1	5	3
7	0	1	1	0	1	1	5	3
7	0	1	1	1	0	1	5	3
7	0	1	1	1	1	1	5	3
7	0	0	0	0	0	2	5	3
7	0	0	0	0	1	2	5	3
7	0	0	0	1	0	2	5	3
7	0	0	0	1	1	2	5	3
7	0	0	1	0	0	2	5	3
7	0	0	1	0	1	2	5	3
7	0	0	1	1	0	2	5	3
7	0	0	1	1	1	2	5	3
7	0	1	0	0	0	2	5	3
7	0	1	0	0	1	2	5	3
7	0	1	0	1	0	2	5	3
7	0	1	0	1	1	2	5	3
7	0	1	1	0	0	2	5	3
7	0	1	1	0	1	2	5	3
7	0	1	1	1	0	2	5	3
7	0	1	1	1	1	2	5	3
8	1	0	0	0	0	0	5	3
8	1	0	0	0	1	0	5	3
8	1	0	0	1	0	0	5	3
8	1	0	0	1	1	0	5	3
8	1	0	1	0	0	0	5	3
8	1	0	1	0	1	0	5	3
8	1	0	1	1	0	0	5	3
8	1	0	1	1	1	0	5	3
8	1	1	0	0	0	0	5	3
8	1	1	0	0	1	0	5	3
8	1	1	0	1	0	0	5	3
8	1	1	0	1	1	0	5	3
8	1	1	1	0	0	0	5	3
8	1	1	1	0	1	0	5	3
8	1	1	1	1	0	0	5	3
8	1	1	1	1	1	0	5	3
8	1	0	0	0	0	1	5	3
8	1	0	0	0	1	1	5	3
8	1	0	0	1	0	1	5	3
8	1	0	0	1	1	1	5	3
8	1	0	1	0	0	1	5	3
8	1	0	1	0	1	1	5	3
8	1	0	1	1	0	1	5	3
8	1	0	1	1	1	1	5	3
8	1	1	0	0	0	1	5	3
8	1	1	0	0	1	1	5	3
8	1	1	0	1	0	1	5	3
8	1	1	0	1	1	1	5	3
8	1	1	1	0	0	1	5	3
8	1	1	1	0	1	1	5	3
8	1	1	1	1	0	1	5	3
8	1	1	1	1	1	1	5	3
8	1	0	0	0	0	2	5	3
8	1	0	0	0	1	2	5	3
8	1	0	0	1	0	2	5	3
8	1	0	0	1	1	2	5	3
8	1	0	1	0	0	2	5	3
8	1	0	1	0	1	2	5	3

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
8	1	0	1	1	0	2	5	3
8	1	0	1	1	1	2	5	3
8	1	1	0	0	0	2	5	3
8	1	1	0	0	1	2	5	3
8	1	1	0	1	0	2	5	3
8	1	1	0	1	1	2	5	3
8	1	1	1	0	0	2	5	3
8	1	1	1	0	1	2	5	3
8	1	1	1	1	0	2	5	3
8	1	1	1	1	1	2	5	3
10	1	0	0	0	0	0	5	10
10	1	0	0	0	1	0	5	10
10	1	0	0	1	0	0	5	10
10	1	0	0	1	1	0	5	10
10	1	0	1	0	0	0	5	10
10	1	0	1	0	1	0	5	10
10	1	0	1	1	0	0	5	10
10	1	0	1	1	1	0	5	10
10	1	1	0	0	0	0	5	10
10	1	1	0	0	1	0	5	10
10	1	1	0	1	0	0	5	10
10	1	1	0	1	1	0	5	10
10	1	1	1	0	0	0	5	10
10	1	1	1	0	1	0	5	10
10	1	1	1	1	0	0	5	10
10	1	1	1	1	1	0	5	10
10	1	0	0	0	0	1	5	10
10	1	0	0	0	1	1	5	10
10	1	0	0	1	0	1	5	10
10	1	0	0	1	1	1	5	10
10	1	0	1	0	0	1	5	10
10	1	0	1	0	1	1	5	10
10	1	0	1	1	0	1	5	10
10	1	0	1	1	1	1	5	10
10	1	1	0	0	0	1	5	10
10	1	1	0	0	1	1	5	10
10	1	1	0	1	0	1	5	10
10	1	1	0	1	1	1	5	10
10	1	1	1	0	0	1	5	10
10	1	1	1	0	1	1	5	10
10	1	1	1	1	0	1	5	10
10	1	1	1	1	1	1	5	10
10	1	0	0	0	0	2	5	10
10	1	0	0	0	1	2	5	10
10	1	0	0	1	0	2	5	10
10	1	0	0	1	1	2	5	10
10	1	0	1	0	0	2	5	10
10	1	0	1	0	1	2	5	10
10	1	0	1	1	0	2	5	10
10	1	0	1	1	1	2	5	10
10	1	1	0	0	0	2	5	10
10	1	1	0	0	1	2	5	10
10	1	1	0	1	0	2	5	10
10	1	1	0	1	1	2	5	10
10	1	1	1	0	0	2	5	10
10	1	1	1	0	1	2	5	10
10	1	1	1	1	0	2	5	10
10	1	1	1	1	1	2	5	10
11	1	0	0	0	0	0	5	2
11	1	0	0	0	1	0	5	2
11	1	0	0	1	0	0	5	2

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
11	1	0	0	1	1	0	5	2
11	1	0	1	0	0	0	5	2
11	1	0	1	0	1	0	5	2
11	1	0	1	1	0	0	5	2
11	1	0	1	1	1	0	5	2
11	1	1	0	0	0	0	5	2
11	1	1	0	0	1	0	5	2
11	1	1	0	1	0	0	5	2
11	1	1	0	1	1	0	5	2
11	1	1	1	0	0	0	5	2
11	1	1	1	0	1	0	5	2
11	1	1	1	1	0	0	5	2
11	1	1	1	1	1	0	5	2
11	1	0	0	0	0	1	5	2
11	1	0	0	0	1	1	5	2
11	1	0	0	1	0	1	5	2
11	1	0	0	1	1	1	5	2
11	1	0	1	0	0	1	5	2
11	1	0	1	0	1	1	5	2
11	1	0	1	1	0	1	5	2
11	1	0	1	1	1	1	5	2
11	1	1	0	0	0	1	5	2
11	1	1	0	0	1	1	5	2
11	1	1	0	1	0	1	5	2
11	1	1	0	1	1	1	5	2
11	1	1	1	0	0	1	5	2
11	1	1	1	0	1	1	5	2
11	1	1	1	1	0	1	5	2
11	1	1	1	1	1	1	5	2
11	1	0	0	0	0	2	5	2
11	1	0	0	0	1	2	5	2
11	1	0	0	1	0	2	5	2
11	1	0	0	1	1	2	5	2
11	1	0	1	0	0	2	5	2
11	1	0	1	0	1	2	5	2
11	1	0	1	1	0	2	5	2
11	1	0	1	1	1	2	5	2
11	1	1	0	0	0	2	5	2
11	1	1	0	0	1	2	5	2
11	1	1	0	1	0	2	5	2
11	1	1	0	1	1	2	5	2
11	1	1	1	0	0	2	5	2
11	1	1	1	0	1	2	5	2
11	1	1	1	1	0	2	5	2
11	1	1	1	1	1	2	5	2
11	0	0	0	0	0	0	5	2
11	0	0	0	0	1	0	5	2
11	0	0	0	1	0	0	5	2
11	0	0	0	1	1	0	5	2
11	0	0	1	0	0	0	5	2
11	0	0	1	0	1	0	5	2
11	0	0	1	1	0	0	5	2
11	0	0	1	1	1	0	5	2
11	0	1	0	0	0	0	5	2
11	0	1	0	0	1	0	5	2
11	0	1	0	1	0	0	5	2
11	0	1	0	1	1	0	5	2
11	0	1	1	0	0	0	5	2
11	0	1	1	0	1	0	5	2
11	0	1	1	1	0	0	5	2
11	0	1	1	1	1	0	5	2

Start (S)	Succession	Aspect	Pine	Oak	Deciduous	Water	DΔ	TΔ
11	0	0	0	0	0	1	5	2
11	0	0	0	0	1	1	5	2
11	0	0	0	1	0	1	5	2
11	0	0	0	1	1	1	5	2
11	0	0	1	0	0	1	5	2
11	0	0	1	0	1	1	5	2
11	0	0	1	1	0	1	5	2
11	0	0	1	1	1	1	5	2
11	0	1	0	0	0	1	5	2
11	0	1	0	0	1	1	5	2
11	0	1	0	1	0	1	5	2
11	0	1	0	1	1	1	5	2
11	0	1	1	0	0	1	5	2
11	0	1	1	0	1	1	5	2
11	0	1	1	1	0	1	5	2
11	0	1	1	1	1	1	5	2
11	0	0	0	0	0	2	5	2
11	0	0	0	0	1	2	5	2
11	0	0	0	1	0	2	5	2
11	0	0	0	1	1	2	5	2
11	0	0	1	0	0	2	5	2
11	0	0	1	0	1	2	5	2
11	0	0	1	1	0	2	5	2
11	0	0	1	1	1	2	5	2
11	0	1	0	0	0	2	5	2
11	0	1	0	0	1	2	5	2
11	0	1	0	1	0	2	5	2
11	0	1	0	1	1	2	5	2
11	0	1	1	0	0	2	5	2
11	0	1	1	0	1	2	5	2
11	0	1	1	1	0	2	5	2
11	0	1	1	1	1	2	5	2

Start (S): Land-cover given by code in Table 4.1

Succession: 0 = regeneration pathway, 1 = secondary pathway

Aspect: 0 = North, 1 = South

Pine: Presence of Pine seeds, 0 = False, 1 = True

Oak: Presence of Oak seeds, 0 = False, 1 = True

Deciduous: Presence of Deciduous seeds, 0 = False, 1 = True

Water: Soil moisture, 0 = xeric, 1 = mesic, 2 = hydric

DΔ: Land-cover given by code in Table 4.1

TΔ: Total time (years) required to complete transition from S to DΔ

APPENDIX II

Curve numbers used in Equation 4.3. Use of the SCS-CN method (SCS 1985) is described in section 4.4.2.2. Infiltration capacity decreases from soil A to soil D. Sources: Ferrér *et al.* (1995), Symeonakis *et al.* (2004).

Soil	Pine	Transition Forest	Pasture	Deciduous	Scrub	Holm Oak	HOP	Crops	Urban	Burnt
				Slope < 3%						
A	35	35	71	35	46	35	56	62	93	91
B	54	54	78	54	68	54	75	72	93	91
C	69	69	82	69	78	69	86	78	93	91
D	77	77	86	77	83	77	91	82	93	91
				Slope ≥ 3%						
A	39	39	76	39	46	39	56	65	96	94
B	60	60	82	60	68	60	75	76	96	94
C	73	73	88	73	78	73	86	84	96	94
D	78	78	91	78	83	78	91	87	96	94

APPENDIX III

Stakeholder Model Evaluation Interview Translations. These transcriptions were made by Dr Raul Romero Calcerrada following the stakeholder interviews and presented in chapter eight.

Español	English
"Cuando se produce (especialmente con sello ecológico) y se transforma lo producido, sí es rentable la actividad agraria. Aquellos agricultores que son empresarios pueden vivir bien y se tienden a ser competitivos. La agricultura tradicional va a desaparecer por falta de rentabilidad y los precios de los productos… Si tienes una explotación ganadera (p.e. La "maríquesos".), en la cual produces quesos que además comercializas, entonces tienes una explotación rentable. Hay que asociar una actividad industrial a la actividad productiva primaria. Hay si es rentable y se asocia toda la familia (padre, mujer e hijos). Mientras que en otro tipo de explotación ganadera se abandona."	"Where production has the organic stamp; yes, agricultural activity is profitable. There are a few organic farmers here who are competitive businessmen and live well. But traditional [forms of] agriculture are disappearing because of a lack of profitability and low product prices. … If you have a livestock exploitation producing cheese, you need to have commercial links with supermarkets to be profitable. Or, you need to associate your agricultural activity with other commercial activities on your land. Then you can be profitable and all the family is involved. If not, livestock exploitations are abandoned."
"Yo no creo que en San Martín existan actividades que por si solas sean sostenibles. El viñedo ya está mal (abandonado o los que no está abandonado no se recoge las cosecha), y, en el futuro, con la nueva subvención por arranque va a estar peor o desaparecerá más del 50%."	"I don't believe agricultural activities in San Martín are [economically] sustainable. Vineyards are in decline – either abandoned or those that are not often remain unharvested – and, in the future, the removal of subsidies is going to make the situation worse. I'd say more than half [of vineyards] will disappear."
"…los agricultores son a tiempo parcial, tipo hobby o por mantener la tradición. Los hijos o nietos de esos no tienen interés porque no son rentables y necesitan mucha dedicación. Los jóvenes se van o buscan otro trabajo."	"…most farmers are part-time, maintaining the traditional agriculture. The children or grandchildren of those [farmers] do not have interest [in agriculture] because is it not profitable and requires a lot of dedication. The youths go or they seek other work."
"El albañil es el tipo de trabajo que interesa. ¿Por qué no tienes gente en las huertas? Porque los albañiles tienen libres los fines de semana. En la huerta trabajas muchas horas, fines de semana y, en muchos casos tienes	"Building is the type of work that interests people here. Why don't people want to work in the greenhouses? Because the bricklayers and builders have the weekends free. In the greenhouses you work many hours, and

Español	English
que llevar la producción al mercado por las noches.”	weekends, and in many cases you must take products to market at night.”
“Ahora 2 de 24 agricultores quieren seguir la dedicación de sus padres. En cuanto el padre se jubila los hijos se pasan a la construcción o otros dejan la explotación y se dedican a la distribución de productos hortícolas. Es un trabajo muy esclavo”	“Now, two of 24 farmers want to follow the footsteps [occupation] of their parents. As soon as the father retires the children go into construction or others leave agricultural production and move into [agricultural] product distribution. It [farming] is very hard work.”
“En el NE de Villa del Prado va a desaparecer la actividad agrícola. Hay grandes fincas que se cultivaban de cereales. Ahora se pasaría a caza. Ya han pasado en otras fincas.”	“In the north east of Villa of the Prado agricultural activity is going to disappear. There are large properties that were cultivated of cereals or Dehesa. Now one will become hunting for sure. This has already happened in one other property.”
“Si a la hora de hacer el análisis, dejas fuera una variable como el crecimiento urbanístico, se pierde una parte de información que explica porque se pierden los cultivos. Porque está muy vinculado, sobre todo en la zona próxima a los cascos urbanos. La rentabilidad inmediata prima que la gente venda esas propiedades.”	“If, when making your model, you leave out variables like land prices, you will be missing data that explain why crops are lost [i.e. abandoned]. These [variables] are tied – abandonment is mainly in the zone next to the urban areas. The immediate profit [of selling the land for urban development] means that people sell those agricultural properties.”
“Los que trabajan y puede vivir realmente de la viña son tres o cuatro. Los demás es una dedicación a tiempo parcial (renta añadida) o como una ocupación cultural (Heredado de padres o abuelos).”	“Those that work and can live on the land really are only three or four persons. The others work part time, for additional income, or do it as cultural occupation, inherited from parents or grandparents.”
“El viñedo no se abandono por cuestiones de lejanía, es una cuestión de que es productiva o no. Además no hay un centro… bueno el pueblo. Entonces puede generarse un efecto inverso. Cuanto más cerca esté del pueblo antes se abandona (cuestiones urbanísticas), mientras que cuando más alejado se mantiene en producción.”	“Vineyard abandonment is not a distance question, it is a question of whether it is productive or not. In addition the town is not a market centre … in fact, there could be a reverse effect. The closer areas to town will be abandoned first [due to higher land prices for urban development], whereas land further away will be maintained production.”
“Creo que si funciona el modelo. Me gusta el modelo. Solo tengo una duda con el tema de la distancia. Aquí no es importante ese aspecto.”	“I do not think the distance rules are very useful. The subject of distance does not influence decisions here. Everything is so close; it’s a maximum of 7 km [from field to town]. That’s no distance.”
“Los que tienen muchas viñas por economía	“Those that have big properties can maintain

Español	English
de escala pueden mantener la actividad.”	the activity through scales of economy.”
[¿Qué no hemos considerado en el modelo?] “La explotación urbanística. La especulación urbanística. Ha subido el precio de las tierras. A mi me hubiese gustado comprar. Sin embargo, se están vendiendo las tierras a un precio que no puedo comprar. Si alguien quiere tierras para cultivar no puede comprar, porque se están vendiendo con las expectativas de suelo urbano.”	[What have you not considered in the model that is important?] “The urban aspect. The urban speculation. The price of land has risen. I would like to buy land but it is very expensive and I can’t afford to. If somebody wants land to cultivate often they cannot buy, because it is being sold with urban speculation in mind.”
“Estos modelos nos complementan la percepción que tenemos. Tiene que llegar a los usuarios para la toma de decisiones … Con estos modelos conocemos previsiones de futuro, podemos realizar una planificación y asignar fondos a determinados aspectos. Imagina que nosotros queremos trabajar en el tema de turismo, pero no tenemos en cuenta el efecto del abandono del desarrollo agrícola. Este tipo de modelo nos ayuda a tomar decisiones sabiendo los que va a pasar.”	“These [types of] models complement the perceptions of change we have. But it must be available for end-users for decision-making. … With forecasts of the future, we can make plans and assign funds for certain aspects. For example, we want to develop our tourism sector in the future, but we haven’t considered the effects of the abandonment of the agricultural sector. This type of model helps us to make decisions knowing how this is going to happen.”
“El observar estos modelos te ayuda a ver la situación. Cuando lo ves en un soporte cartográfico lo ves mejor o de forma más clara. De cara a una planificación o actuaciones en una zona es relevante.”	“Observing these models helps you to see the [overall] situation. When you see it in a map you see it [LUCC] clearer. The model is excellent for planning or managing a zone [of the municipality]”.
“[Son útiles estos modelos] Sí. Aunque cuando estás muy metidos en estos temas ya tienes una idea de lo que está pasando. Te esperas las evoluciones.”	“[These models are useful,] yes. But when you are the subject of the model, already you have an idea of what is happening. You are already anticipating the change yourself.”

APPENDIX IV

Enclosed Compact Disc. The enclosed CD contains larger versions of figures from chapters two, three, four, and six, materials used in stakeholder interviews, source code and executable model files, digital offprints of papers written by the author during the research process, and a digital copy of this thesis. Also see http://landscapemodelling.net net for other related materials.