Managing complex systems: an interdisciplinary approach to modelling the effect of social and ecological interactions on carbon storage in blanket peatlands

Dylan Martin Young

Submitted in accordance with the requirements for the degree of Doctor of Philosophy

The University of Leeds
School of Geography

March 2016

Acknowledgements

Firstly I would like to thank my research supervisors, Paul Chatterton and Joe Holden, who gave me the opportunity to study and Leeds, for their advice, support, and guidance throughout this project. A special thanks goes to Andy Baird for his tireless help with DigiBog, and for providing me with encouragement to get involved with peatland modelling. I would like to thank Paul Morris who has provided both Digibog and thesis advice and help. Thanks also go to Duncan Quincey and Jon Lovett who were members of my research support group.

This project was funded by an ESRC/NERC interdisciplinary studentship, which I would like to gratefully acknowledge.

I wish thank the Moors for the Future Partnership for supporting this research, and would like to offer a special thanks to Sharon Davies, Mike Pilkington, and Jon Walker who provided me with a base for the participatory work, and helped to organise and set up the workshop activities. They also tested my thinking about the various stages of the participatory process, helping me to see what needed further thought. These sessions invariably helped the workshops run smoother. Mike and Jon also helped me to identify and recruit people to join workshops. I am especially grateful to everyone from the peatland community who participated in workshops over the course of the project – without their effort, time and patience, this work would not have been possible.

I would like to thank David Lloyd of the University of Surrey for discussing the modelling of fuzzy cognitive maps and for providing me with the software to calculate the driver nodes that were used in one of my participatory workshops. At Leeds, Dan Olner from the School of Geography, John Stell from the School of Computing, and Jonathan Ward from the School of Maths, all provided assistance with graphs and networks, and Richard Rigby helped me begin to write scripts in Linux. Thanks also to my current and past office companions and fellow PhD students; Kisandra Bynoe, Gemma Dooling, Greta Dragie, Freddie Draper, Tom Kelly, Adriane Muelbert, and Liz Watson. Thanks to Antony Blundell for bike rides and peatland chats.

Jane has been by my side through this journey, she encouraged me to take up this opportunity and has supported me continuously throughout the whole process, and so my biggest thanks go to her.

Finally, I would like to dedicate this thesis to the memory of my father.

Abstract

Peatlands are globally important for carbon storage, water quality and biodiversity. However, many have been degraded by land use, and efforts to conserve or restore them are often contested by stakeholders with different objectives. Peat accumulates as a result of a complex network of interactions, which makes it challenging to predict the impact of climate and land use. Stakeholders' knowledge may help to provide insights into these interactions and into the issues that underpin conflict. To investigate the impact of social and ecological factors on blanket peatland carbon storage, an interdisciplinary approach was used to couple cognitive and peatland development models.

Blanket peatland stakeholders developed fuzzy cognitive maps based on their perceptions of peatland interactions, which they validated to agree on the structure of an aggregate network. To explore the impact of land–use objectives on carbon stocks, stakeholders proposed changes to a set of factors that controlled the state of the network. The changes identified to improve carbon storage and water quality had a positive effect on carbon stored, but those that were proposed to support local livelihoods had no effect on carbon. This was partly because some stakeholders perceived that supporting livelihoods was incompatible with measures that were likely to result in shallower water tables. However, further discussions between stakeholders suggested that land–use objectives could complement each other.

To enrich the results of the network model, the DigiBog peatland model was modified to simulate blanket peat accumulation. Using two factors from the cognitive model, climate change and gully blocking, two novel modelling studies were produced. The first showed that existing peatland development models may overestimate peat accumulation because they aggregate climate variables into annual rather than weekly inputs; the second, that gully blocking is needed to arrest peat losses from oxidation in gullied systems, but that these losses would not be recovered 200 years after gully blocking.

The combination of both cognitive and process–based modelling provides an example of how stakeholder knowledge can be incorporated into simulations of complex ecosystems which is likely to be applicable to other social–ecological systems where land use is contested. In this case, doing so provided holistic insights into how stakeholders' perceptions, and the impacts of climatic forcing and restoration, affect carbon storage in blanket peatlands.

Contents

List of Figures

List of Tables

Chapter 1

Introduction

1.1 Introduction

Land-use pressure on natural systems has resulted in widespread landscape degradation. The majority of ecosystems have been affected by anthropogenic activities to the extent that the future persistence of earth systems, landscapes and biomes has become a concern (Barnosky et al. 2012; MacDougall et al. 2013). Global challenges such as climate change are interconnected with land-use at national and local scales. For example, it has long been suggested that the degradation of large areas of carbon-rich landscapes, such as the Amazon rainforest, tropical peat-swamp forests and northern peatlands, for the provision of timber and land for agriculture is an important feedback to the global climate system (Pan et al. 2011; Moore et al. 2013; Petrescu et al. 2015). Exploitation of these landscapes for land use has also generated concerns about biodiversity loss and the loss of ecosystem function such as flood risk mitigation and subsequent negative impacts on human well-being (Millennium Ecosystem Assessment 2005; Wheater and Evans 2009).

Land-use conflict often occurs where the use of resources competes with efforts to conserve or repair ecosystems and livelihoods are threatened (e.g. Saarikoski et al. 2013b; Tollefson 2015). The challenge is, therefore, to understand how to deliver improvements to the condition of ecosystems whilst there are growing and competing demands for greater use of the provisioning and regulating services provided by these interconnected systems (Lambin and Meyfroidt 2011; Chapin et al. 2012). This is a significant challenge because many degraded landscapes are co-evolved social–ecological systems (e.g. Holden et al. 2007) that exhibit complex interactions, and sometimes surprising emergent behaviours, as a result of feedback loops and self-organisation nested across and within spatial and temporal scales (Liu et al. 2007; Folke et al. 2011). Characterised as complex adaptive systems (Levin 1998) (referred to hereafter as complex systems), their dynamics are difficult to understand and predict and their complexity can become overwhelming. To add to this complexity, an understanding of how such a system might respond to autogenic or allogenic change is only a first step: finding solutions to land–use conflicts, if they can be found at all, requires the bottom–up participation of the relevant land-users themselves if they are to be successfully

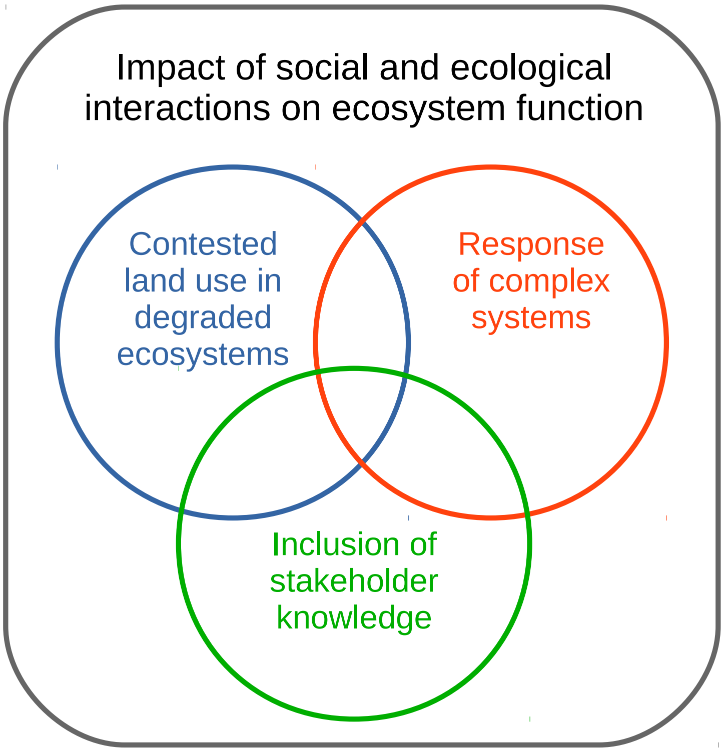


Figure 1.1. Conceptual framework for understanding the impact of social and ecological interactions on ecosystem function.

implemented (Kates et al. 2001; Reed 2008; Danielsen et al. 2010; Raymond et al. 2010; Davies et al. 2015; Corral-Quintana et al. 2016).

These characteristics form three converging themes that provide a conceptual framework in this thesis for understanding the impact of social and ecological interactions on important ecosystem functions such as carbon storage; (1) the conservation or repair of degraded semi–natural systems often results in conflict over land use; (2) many semi–natural systems are the result of complex interactions between humans and nature that are difficult to understand and predict; and (3) decisions about future land use should incorporate knowledge from all relevant stakeholders (Figure 1.1). Both Reynolds et al. (2009) and Whitfield and Reed (2012) report on similar bottom–up frameworks to address issues of dryland use that integrate a wide variety of stakeholder knowledge, and explicitly recognise that drylands are complex coupled systems (discussed in Chapter 2).

The conceptual framework shown in Figure 1.1 can be applied to many peatlands throughout the world: peatlands continue to be degraded by land use, are complex systems, and important stores of carbon (C). It is estimated that peatlands store ≈33% of global soil carbon (Holden 2005a) with accumulation rates in the order of 18.6 g C m^{-2} y^{-1} (Yu et al. 2010). Northern peatlands account for ≈547 (473–621) Gt C with additional quantities of C stored as peat in tropical and southern regions estimated to be ≈50 Gt and ≈15 Gt C respectively (Yu et al. 2010). Peatlands are thought to have an impact on the global climate system and previous studies have proposed that during the late Holocene (≈5000 yrs BP), the accumulation of peat had a net cooling effect on the climate (Frolking and Roulet 2007). The process of peat accumulation involves the exchange of carbon dioxide and methane with the climate system: both are greenhouse gases and have a radiative effect on the atmosphere. Peatlands are therefore important stores, and potential sources, of carbon and their protection and restoration is of global concern (Bonn et al. 2014).

Much recent research on peatlands has focussed on the interactions between land use and the climate system (Beetz et al. 2013; Moore et al. 2013; Helfter et al. 2015; Petrescu et al. 2015). Upland and lowland peatlands throughout the globe have been degraded as a result of afforestation for timber products, deforestation for agriculture, rearing of livestock and game–birds, extraction for fuel and horticulture, and as a result of airborne pollution (such as acid rain) (Bullock et al. 2012; Strack and Zuback 2013; Allen et al. 2013; Law et al. 2015; Busch et al. 2015). Land use can cause accumulated peat to be converted to atmospheric or fluvial sources of carbon (e.g. Moore et al. 2013), and has resulted in losses of terrestrial and aquatic biodiversity (e.g. Anderson and Radford 1993; Brown et al. 2013). Actions to reduce or reverse losses of carbon and biodiversity often compete with multiple demands for the use of peatlands (Venter et al. 2013; Law et al. 2015; Noordwijk et al. 2014) making these landscapes an important example of a degraded complex social–ecological system where there is high reliance on the provision of ecosystem services over global, national and local scales, and where land-use is often contested.

Degraded peatlands around the globe have been the subject of restoration activities (e.g. González et al. 2013; Schimelpfenig et al. 2013) which are promoted by agreements such as the Convention on Biological Diversity in order to reduce the impact of climate change and improve biodiversity (Bonn et al. 2014). Restoration of UK peatlands has been widespread (Parry et al. 2014). Initially most projects focussed on protecting or improving habitats and biodiversity (Holden et al. 2008), whilst latterly there has been a shift to improving ecosystem service provision because of the negative impact of degraded peatlands on human well-being as well as on biodiversity (Maltby 2010; Bonn et al. 2014).

There are several different types of peatlands that occur throughout the world (e.g. raised bogs and peat swamp forests). Blanket peatlands are found in hyperoceanic regions such as Norway, Iceland, Patagonia, and in the UK (Gallego-Sala and Prentice 2012) where they are the most common peatland type. Blanket peatlands in the UK provide a number of important ecosystem services over local, national and global scales such as livelihoods in horticulture, agriculture, game keeping, water provisioning, and climate regulation (Bonn et al. 2009; Parry et al. 2014). Although management of these peatlands has occurred over centuries (Bragg and Tallis 2001), land-use has often been a source of conflict (Reed et al. 2009a; Maltby 2010) (for example the impact of managed burning on carbon stocks and water quality; Harris et al. 2011; Dixon et al. 2015); with some arguing that in light of climate change, management to improve the persistence of carbon stocks should be a high priority for land-users (e.g. Billett et al. 2010; Maltby 2010). However, growing awareness of the importance of peatlands as stores of carbon and rising costs of water processing (Blundell and Holden 2015), has increased conflict between those who seek to repair hydrological and vegetation condition to improve ecosystem functions, and those whose livelihoods are embedded in current land–use regimes. Given the importance of peatland carbon stores and the contested nature of land–use objectives, in this thesis I use the conceptual framework in Figure 1.1 to explore the impact of land use and climate variables on blanket peatland carbon storage.

1.2 Research aims

The overall aim of this thesis is to develop new knowledge of blanket peatlands as a complex system, and to enhance current understanding of the impact of social and ecological interactions on carbon storage. To achieve this aim, four research questions were defined;

1. What are the key interacting social and ecological factors that are needed to represent blanket peatlands as a complex system?
2. How can these factors be used to evaluate the impact of land–use objectives on blanket peatland carbon storage?
3. How should blanket peatlands be managed to achieve the objectives of (a) maintaining or increasing carbon storage, (b) improving the quality of water supplied and, (c) supporting local livelihoods: what are the implications for current land uses?
4. What is the predicted impact of social and ecological factors on the centennial to millennial storage of carbon in blanket peatlands when conceptualised as a complex system?

It is envisaged that the answers to these questions will inform the often intensely contested debate between peatland stakeholders about how blanket peatlands could be managed in relation to both local and wider societal objectives for peatlands, and provide a framework for engagement that can be used to explore new questions about how long–term carbon storage in peatlands is affected by land use. More broadly, it is anticipated that this research will provide an example, applicable to many degraded ecosystems, of how groups of stakeholders with conflicting viewpoints about land use, can collaborate to inform the management of complex systems.

1.3 Thesis structure

The thesis structure is shown in Figure 1.2. Research chapters (3–6) start with a summary and diagram of the work carried out, and includes an introduction that incorporates the literature related to the specific topic and methods used. In Chapter 2, I review; the literature related to the impact land use in peatlands and its contested nature; the evidence for peatlands as complex systems; and the potential approaches for modelling peatlands as complex systems that can incorporate a wide range of stakeholder knowledge, and be used to explore the impact of land use on carbon storage. I also outline the methods selected, and study area: detailed descriptions of methods and models are included in the relevant research chapters. I develop a four stage integrated approach to answer each of the research questions. In Chapter 3, I explore the use of participatory workshops to co–develop a cognitive (network) model of blanket peatlands, and to identify the perceived importance of social and ecological factors and their network of interactions. Chapter 4 includes three main stages; (1) network analysis of the structure of the co–developed model to identify the factors that determine the structural resilience of the peatland, and those that can be used to 'drive' the future development of the model peatland; (2) a participatory workshop to identify the actions to achieve three land–use objectives related to livelihoods, water quality, and carbon storage; and (3) modelling with the cognitive (network) model to evaluate the impact of land–use objectives on carbon storage. In Chapter 5, I develop a new version of the **`DigiBog`** peatland model suitable for simulating blanket peat accumulation in 2D or 3D. Chapters 5 and 6 use the new **`DigiBog`** model to investigate the centennial to millennial impact of climate variables and land use (identified in Chapter 4) on carbon storage in order to explore the long–term response of blanket peatlands to external forcing. Chapter 7 provides a synthesises of the results of Chapters 3–6, and assesses how the approach integrates stakeholder knowledge with models of complex systems in social–ecological systems where land use is contested; it also outlines the contributions of this thesis to understanding blanket peatlands as a complex systems.

Systems that are the product of the interactions between people and the natural environment have been termed as coupled human and natural systems (CHANS) (e.g. Liu et al. 2007) or human–environment systems (e.g. Reynolds et al. 2009). In this thesis, I use the term social–ecological system or social and ecological interactions to encompass the many human and natural interactions that are entwined in blanket peatland carbon accumulation, land use, and restoration.

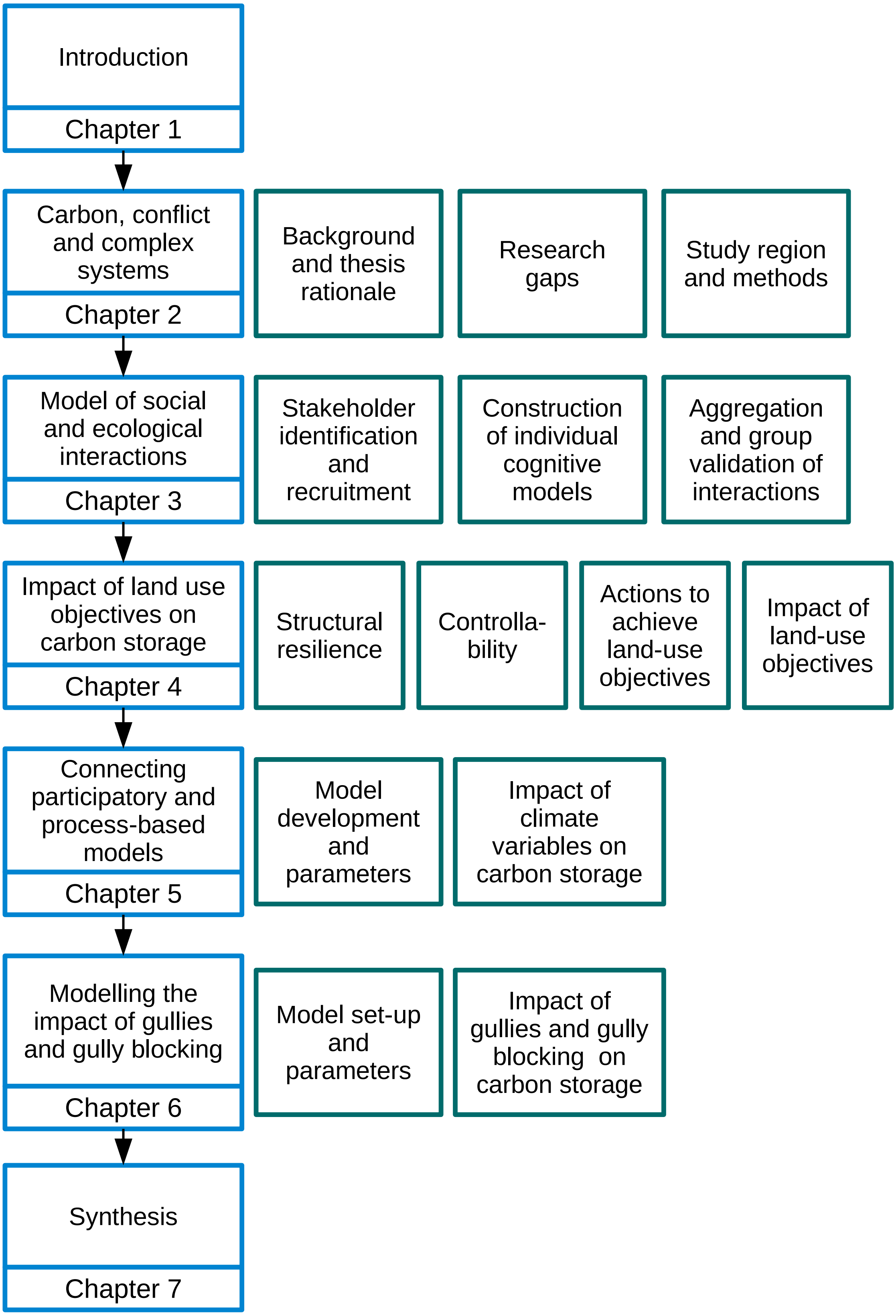


Figure 1.2. Outline of thesis chapters. Green boxes represent the main research activities undertaken during this thesis.

Chapter 2

Carbon, conflict and complex systems

2.1 Chapter summary

This chapter develops the three themes outlined in Chapter 1: (1) global land–use has resulted in degraded semi–natural systems where land use is often contested; (2) many ecosystems are complex systems that are the result of human and natural interactions, and the autogenic response of these systems to land use or climate make them difficult to understand and predict; and (3) decisions about future land use and the repair of damaged ecosystems should be informed by local, practitioner, and scientific knowledge. I review these themes within a broad context of peatland use, with a specific focus on UK blanket peatlands.

The chapter begins with the challenge of repairing damaged ecosystems and the contested nature of land use (Sections 2.2.1 and 2.2.2), and outlines the importance of carbon stored in peatlands (Section 2.2.3). By reviewing the background to contested land use in peatlands (Section 2.3), the evidence for peatlands as complex systems (Section 2.4), participatory approaches to land–use conflict (Section 2.5), and the impact of land use on blanket peatland carbon storage (Section 2.6), I bring the three main themes together to identify a rationale for the approach used in this thesis. Section 2.7 summarises the relevant gaps in the literature and identifies an approach, to address these gaps, that incorporates participatory modelling based on the use of mental models, and a peatland development model. Finally Section 2.8 describes the study region where participatory workshops were held, and describes the selection of two modelling approaches that can be incorporated into a participatory framework to develop new knowledge about carbon storage in blanket peatlands. At several points, cognitive maps, fuzzy cognitive maps, and networks are discussed. All of these terms refer to models comprising a number of nodes connected by links, in some cases the links may be directed and weighted. The difference in terminology derives from their method of development. The cognitive and fuzzy cognitive maps are derived directly or indirectly from people's mental models.

2.2 The challenge

2.2.1 Degradation of ecosystems on a global scale

The majority of global ecosystems have been affected by humans over millennial timescales (Turner et al. 2007). From 1950 to 2000, there was unprecedented and, to a large extent, irreversible negative impact by humans on ecosystems (Millennium Ecosystem Assessment 2005; Turner et al. 2007). There is widespread concern that this degradation will reduce well-being for future generations as humans continue to deplete ecosystem resources at regional and global scales (Foley et al. 2005; Ceballos et al. 2015). The unsustainable use of global resources may jeopardise the earth-systems that support humans such as the regulation of climate and freshwater resources. Planetary boundaries were conceptualised to identify the safe operating limits for the ecosystem process on which humans rely (Rockstrom et al. 2009). Nine boundaries have been defined as climate change, biodiversity loss, nitrogen and phosphorous cycles, ozone depletion, ocean acidification, freshwater use, land-use change, atmospheric aerosol loading and chemical pollution (Rockstrom et al. 2009).

Continued exploitation of ecosystems that push beyond these nine boundaries could lead to collapses in resources, such as those previously reported in some ocean fisheries as a result of overfishing such as Newfoundland cod (Biggs et al. 2009). Although the existence of planetary scale tipping points is debated (Brook et al. 2013), the avoidance of local and regional collapses that reduce ecosystem services essential to well-being has resulted in pressure to reform land use in a call for sustainable use of resources, and for stewardship of earth's "life support" systems (Rockstrom et al. 2009; Folke et al. 2011; Biggs et al. 2012; Chapin et al. 2012). Griggs et al. (2013) suggested that these life-support systems should become integral to the new sustainable development goals, proposed at Rio+20 for the post-2015 follow on from the Millennium Development Goals, because ongoing environmental degradation will constrain improvements in well-being. The overwhelming, and substantial, challenge is to repair or improve degraded ecosystem processes and stem or reverse losses in biodiversity, whilst at the same time global population growth creates increased pressure on the provisioning and regulating services provided by ecosystems (Millennium Ecosystem Assessment 2005). As a consequence, land use is central to this challenge.

Land use often results in intense and intensely contested debates between stakeholders with competing interests for delivery of the goods and services that ecosystems provide to people; and when livelihoods are negatively affected (Young et al. 2005). Land–use decisions have been increasingly determined by national and international factors that often outweigh the local decision-making processes that would have previously prioritised the needs of local people (Reenberg and Fenger 2011). Global programmes have been developed to provide the impetus to reduce or eliminate processes that lead to ecosystem degradation and to restore damage caused by land use. These programmes also bring together the twin goals of sustainable use of ecosystems and improvement of well-being. The UN Environment Programme (UNEP) initiated the The Convention for Biological Diversity (CBD) which came into force at the end of 1993. Its aims are: (1) the conservation of biological diversity; (2) sustainable use of the components of biological diversity; and (3) fair and equitable sharing of the benefits arising out of the utilisation of genetic resources (www.cbd.int/intro/default.shtml). The strategic vision of the CBD is that, *"By 2050, biodiversity is valued, conserved, restored and widely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people"* (Secretariat of the CBD 2010). This vision is disseminated through five strategic goals and twenty targets agreed at Nagoya, Japan in 2010 (known as the Aichi targets), that include references to carbon storage, water and livelihoods. For example, target 15 states, *"By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification"* (Secretariat of the CBD 2010).

In Europe, Natura 2000 forms the mainstay of nature and biodiversity conservation and encompasses Aichi targets to improve biodiversity and ecosystem service provision through restoration of 15% of degraded ecosystems by 2020 (Egoh et al. 2014). The UN-REDD programme (reduction in emissions from deforestation and forest degradation) and subsequently the REDD+ and REDD–ALERT (Reducing emissions from Deforestation and Degradation from Alternative Land Uses in the Tropics) programmes provide incentives for sustainable management of forests. Since 2005 Brazil has decreased rates of deforestation by 75% whilst food production has increased (Tollefson 2015). REDD+ has also been used as a driver to halt logging operations and spread of palm oil plantations in Indonesia, much of which degrades peatland carbon stocks (Busch et al. 2015). In the UK, the UK Post-2010 Biodiversity Framework (JNCC and Defra 2012) sets out the structure, up to 2020, to meet the Aichi targets. This framework outlines how the five Aichi strategic goals will be applied at a UK scale, including best practice and common approaches for ecosystem restoration (strategic goal D; JNCC and Defra 2012, p.7).

2.2.2 The 'wicked' nature of land use conflict

Notwithstanding the success of some initiatives to address the growing demand for ecosystem services alongside the restoration and conservation of ecosystems, concerns have been raised about the failure of conservation planning to deliver approaches that can address gaps between research and implementation; and to impact policy decisions that enable transformative change (e.g. Biggs et al. 2011; Rudd 2011; Wiek et al. 2012; Young et al. 2016b). McShane et al. (2011) argued that many previous attempts seek 'win-win' outcomes for both conservation and improvements to well-being, but have largely failed to achieve their aims. Some of the reasons given for this failure include a conflict with the livelihoods of local people, lack of viable alternative livelihood choices, a lack of willingness to share knowledge and decision–making, and economic solutions that do not benefit local users of natural resources (McShane et al. 2011; Noordwijk et al. 2014; Shoreman-Ouimet and Kopnina 2015; Young et al. 2016a). Trade-offs between the conservation of natural resources and well-being are seemingly inevitable and could avoid the, perhaps, unrealistic aims and likelihood of dashed expectations of 'win-win' approaches, that can lead to failed outcomes (e.g McShane et al. 2011; Biggs et al. 2012; Redpath et al. 2013; Tolvanen et al. 2013): although Reed et al. (2013b) demonstrated that stakeholder groups with conflicting objectives can help to identify complementary outcomes that could enhance multiple ecosystem benefits.

Realising improvements to ecosystem condition whilst utilising natural resources to improve well-being is highly challenging because of the complex network of biophysical, social and policy interconnections that are characteristic of social–ecological systems (Noordwijk et al. 2014; Liu et al. 2015a). This complexity encompasses a multitude of stakeholders at global, national and local scales, that are likely to have diverse values and differing objectives. As a result land–use problems are laden with opportunities for conflict and often appear intractable. Commonly referred to as "wicked" problems, the term is used to convey the difficulties in solving these problems (if indeed they can be truly solved) (e.g. Ludwig 2001; Fairweather 2010; Polk 2014). Rittel and Webber (1973) coined the terms "tame" and "wicked" in relation to the type of problem to be solved in complex societal planning problems, such as policing and healthcare, that featured nested interconnections over multiple scales. These are also the characteristics of current challenges to reverse ecosystem degradation, indeed perhaps more so because social systems are also highly interconnected with natural systems. Tame problems are those traditionally addressed by science and engineering that can be clearly defined, have a point at which a solution is reached, have solutions that can be tested as right or wrong, can be solved in a similar manner to problems in the same class, and have solutions that can by hypothesised, tested and then accepted or rejected (Ritchey 2013). Whereas according to Rittel and Webber (1973), wicked problems display the following ten characteristics:

- There is no definitive problem statement
- There is no stopping rule (i.e. no single correct solution)
- Solutions are not true or false, but good or bad
- There is no immediate and no ultimate test of a solution
- Every solution is a "one-shot operation" (every attempt counts)
- There is no enumerable set of potential solutions
- Every problem is unique
- Every problem can be considered a symptom of another problem
- The existence of a discrepancy representing a problem can be explained in numerous ways
- The planner has no right to be wrong (the aim is to improve some characteristics of the world where people live)

These are indeed challenging problems. Rittel and Webber's list of wickedness helps to at least partly explain the research–implementation gap discussed previously, and the difficulty in achieving sustainable outcomes to land–use conflicts. In this case then, how might challenges related to the sustainable use of earth's resources be addressed? A growing body of scientific discussion has focussed on this challenge (e.g. Kates et al. 2001; Young et al. 2005; Liu et al. 2015a); not only trying to understand the impact of social and ecological interconnections, but also to develop knowledge that can be used to transform current damaging land–use regimes into approaches that can deliver Aichi targets, and Natura 2000 and REDD+ goals for sustainability (Wiek et al. 2012). Much of this work recognises the conflict that surrounds land use and the "wickedness" that makes these goals so difficult to achieve. Compounding the difficulty of this challenge is the process of moving from strategic visions that set goals for ecosystem condition and improvement of well-being (top–down), to delivery at national and local scales where they will be realised (bottom–up). In fact, new sources of conflict can be generated by the top–down imposition of goals without the participation of local stakeholders: a problem seen with the implementation of the EU Habitats Directive (one of the components of Natura 2000) (Paavola et al. 2009).

There has been increased emphasis on describing frameworks and proposing modes of research that are not simply analytical but stress the need to engage stakeholders in collaborations aimed at integrating existing, and co-developing new, knowledge to inform land–use decisions, and to resolve existing or potential conflicts (e.g. Reynolds et al. 2009; Lang et al. 2012; Whitfield and Reed 2012; Polk 2014; Mcpherson et al. 2016). In Section 2.3, I discuss the main drivers of contested land use in blanket peatlands in the UK, and in Section 2.5 I review participatory modelling processes that could be used to integrate stakeholder knowledge into discussions of future peatland uses.

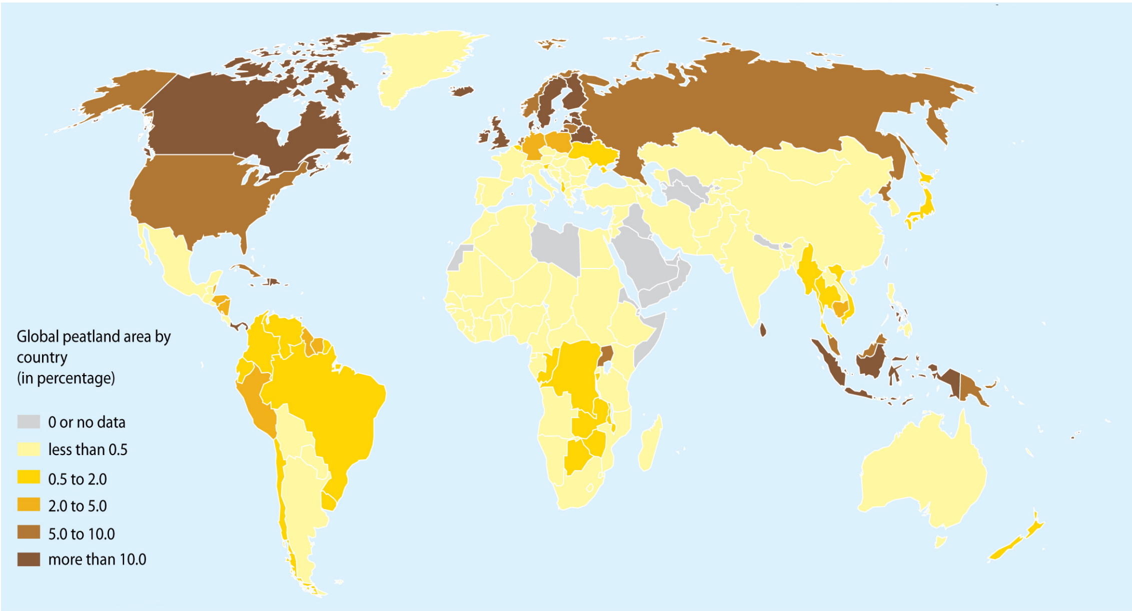


Figure 2.1. Global distribution of peatlands. Darker colours represent a greater proportion of peat by country area. Source: Parish et al. (2008).

2.2.3 Carbon storage in peatlands

Peatlands occur throughout the globe in both upland and lowland locations, most commonly in the high latitudes and the topics (Figure 2.1). Peatlands store atmospheric carbon dioxide (CO_2) as partially decomposed plant litter in waterlogged (anoxic) conditions (Clymo et al. 1984). Low rates of decomposition that are exceeded by plant litter production, cause a build-up of partially decayed organic material which accumulates in small annual increments. Gorham (1991) estimated peat accumulation in subarctic and boreal peatlands as a gain in height of 0.2–0.8 mm y^{-1}, although rates are now usually reported as the amount of carbon (C) gained or lost as g C m^{-2} y^{-1}. Yu et al. (2010)[1] estimated rates of Holocene peat accumulation as 18.6 (northern peatlands), 12.8 (tropical peatlands), and 22.0 (southern peatlands) g C m^{-2} y^{-1}. Peatlands accumulate this partially decomposed plant material over long periods of time (1,000s of years), and can reach depths of up to 15 m (Charman 2002). The mean depth for peatlands in the northern hemisphere varies according to study but was reported as 1–2.3 m by Yu et al. (2010).

Peatland processes and classifications

Hydrology is the dominant physical process in peatlands and plays a significant role relation in both the accumulation and loss of organic carbon. Water transports dissolved gases, particulate material and solutes and creates the anoxic conditions that reduce rates of organic matter decay (Limpens et al. 2008; Belyea 2009). Impeded drainage of mineral soils and the related increase

[1] regions were defined as: northern = latitudes above 30°N, southern = latitudes below 30°S, and tropical = < 30°N and > 30°S.

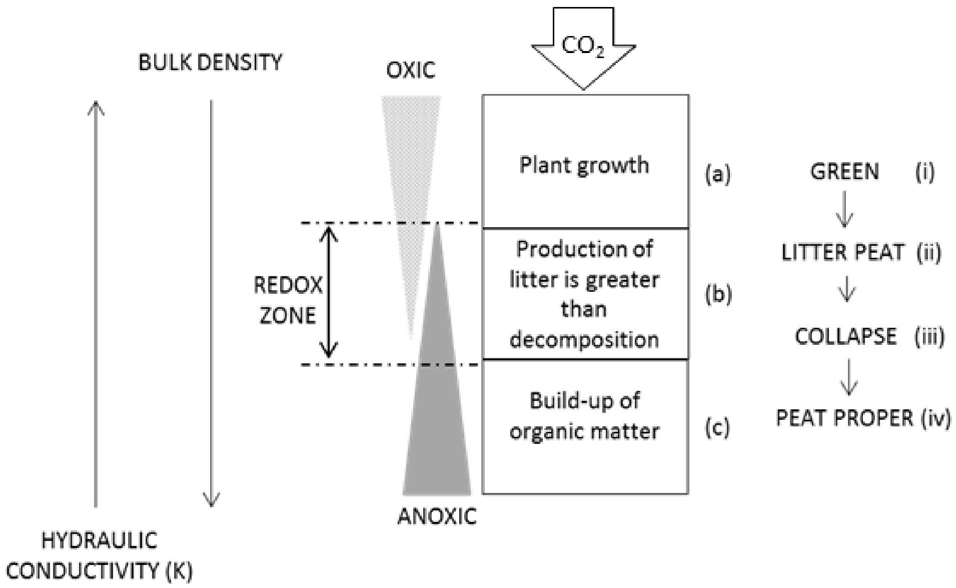


Figure 2.2. Peat structures. (a) Atmospheric carbon dioxide (CO_2) is taken up by plants as part of photosynthesis. (b) Due to anoxic conditions, slow decomposition leads to a greater rate of production than decomposition which in turn leads to (c) a build up of organic matter. The stages (i-iv) are the structural layers proposed by Clymo et al. (1995) that define the different stages of plant decomposition. The oxic zone decreases with depth from the plant growth layer as water is held within the collapsed structures, where bulk density is increased and hydraulic conductivity is reduced, inhibiting microbial decomposition. The redox zone is where hydrological conditions fluctuate so that carbon is oxidised to form CO_2 or reduced to form methane (CH_4).

in waterlogging can be related to; (1) changes in climate (e.g. higher temperatures, increased precipitation); (2) podsolization and acidification of mineral soils; (3) wildfire or; (4) anthropogenic activities such as deforestation (that may be coupled with climate). Waterlogging and leaching of nutrients and minerals can ultimately lead to conditions that favour a build up of organic material and peatland initiation due to paludification (growth of peat directly over mineral soils) (Charman 2002). Water movement generally slows with depth in the peat profile due to the increasing bulk density and decreasing hydraulic conductivity of increasingly humified peat (Figure 2.2).

Peatlands are classified depending on hydrological influxes and effluxes (Moore 1987). On this basis there are two main classifications; (1) ombrotrophic peats (bogs) that receive most water and nutrient input from precipitation and tend to be acidic and nutrient poor; and (2) minerotrophic peats (fens) that in addition to precipitation, receive water and nutrients from ground water and run-off (Charman 2002). Ombrotrophic bogs are further classified according to form, structure and hydrology into “raised” and “blanket” bogs. Raised bogs are typically domed peatlands above a flat or depressed substrate, often with patterned surfaces in the form of concentric or eccentric rings of alternating pools and ridges (Charman 2002). Blanket bog has peat that often follows the underlying topography including covering gently rolling terrain and hillslopes. Blanket bog (blanket peatlands hereafter) is a globally rare ecosystem (Gallego-Sala and Prentice 2012) found in temperate hyperoceanic regions (Charman 2002) where there is high annual rainfall (> 1000 mm), a high number of wet days (> 160 days with > 1 mm of rain), little variation in temperature, and a mean temperature of < 15°C for the warmest month (Lindsay et al. 1988). Charman (2002) suggests that it is the distribution of rainfall that is more important than the total amount. UK blanket peatlands contribute ≈10 %–15 % of the total global extent (Wallage et al. 2006), and are therefore nationally and internationally important. In the UK, peatlands cover ≈15 % of land area and account for 2.3 Gt of carbon (Billett et al. 2010). Blanket peatlands in the UK cover an estimated 1.5 million ha (Scotland ≈1060,000 ha ; England ≈215,000 ha; Wales ≈70,000 ha and Northern Ireland ≈140,000 ha) (Maddock 2011). Blanket peatlands are often a complex mosaic of habitat classes that form to cover large extents of upland landscapes with only rocky outcrops exposed (Rodwell 1991; Charman 2002). Microforms (≈1–10 m in size) classified into hummocks, pools, and hollows which vary in vegetation composition and hydrophysical properties, can be observed as patterns on the surface of blanket peatlands.

Peatlands are valued for their biodiversity and it is known that they are an important part of the global carbon store. However, they also support the livelihoods of local people in a number of activities that have the potential to damage peatland ecosystem function, and alter the balance between peat accumulation and decomposition, as well as negatively impact biodiversity. As a result peatland use is often contested.

2.3 Contested land use in peatlands

2.3.1 Land–use conflict in peatlands around the world

Land use is contested in peatlands around the world. The deforestation of tropical peatlands in Southeast Asia for palm oil production has caused widespread damage to human well–being (nationally and internationally) and has substantially increased emissions of carbon dioxide (Gaveau et al. 2014; Konecny et al. 2016). Reduction in deforestation conflicts with the expansion of commercial agriculture and timber plantations which are the combined main causes of greenhouse gas emissions from peatlands and forests (Busch et al. 2015), but also provide livelihoods in local communities (Chokkalinga et al. 2005; Noordwijk et al. 2014). Mining of oil sands in northen Alberta, Canada, was studied by Rooney et al. (2012) who projected that four of the 10 approved mines would result in the loss of 124.1 km^2 of peatlands which was estimated to be ≈2,400–3,040 t of annual carbon sequestration. This damage and subsequent concerns related to pollution, greenhouse gas emissions, and increased pressure on local infrastructure has resulted in conflict between indigenous and environmental groups, and the Alberta and national government who approved the intensification of oil sands exploitation (Hoberg and Phillips 2011).

In Europe, there has been a growing conflict between the use of peatlands for agriculture, forestry, and fuel, and the increasing awareness of the wider societal benefits associated with peatlands such as climate mitigation and water quality (e.g. Maltby 2010; Bullock and Collier 2011). Conflict has arisen in Ireland because conservationists are perceived to prioritise the environment over local people who favour household scale (mechanical) peat extraction (Bullock and Collier 2011). Whilst in Finland, Tolvanen et al. (2013) found that different stakeholder groups wished to protect and restore peatlands but contested future land uses depending on how they perceived they would benefit personally, and how closely their livelihoods were connected to peatlands (e.g. those who worked in the commercial use of peatlands favoured an increase in the area of production). Peatland use in the UK is also highly heterogeneous and contentious.

2.3.2 Contested land use in UK blanket peatlands

Land use in the UK uplands has had a profound impact on blanket peatlands. Moore (1973; 1975) proposed that human activity was coincident with the initiation of blanket peatlands in Wales and the Peak District and was likely to have been, to some extent, the cause of peat initiation and expansion during the Holocene. But this explanation has been contested by the results of palaeoecological and modelling studies that propose climate as the main driver of blanket peatland development in the UK and in other hyperoceanic regions worldwide (Tipping 2008; Gallego-Sala et al. 2016). The interaction of humans with blanket peatlands has resulted in the development of semi-natural habitats and vegetation types such as moorland and wet heath. The latter is thought to exist mainly as a result of excessive grazing of vegetation on blanket peatlands (Rodwell 1991). And palaeoecological evidence suggests humans have repeatedly modified blanket peatland vegetation over many years following peat initiation (Blundell and Holden 2015).

Peatlands have been managed by people for millennia to create land for agriculture, to grow timber products, to use as fuel, to provide drinking water, and also for horticulture (Bragg and Tallis 2001). Land uses include hills farming (sheep and deer), tourism, field sports, and forestry (Reed et al. 2009a), and, conservation of biodiversity and carbon stocks. Worrall et al. (2011, p.15) estimated that the proportion of UK blanket peatlands under management treatments as; grazed (85%), burnt (18%), afforested (15%) and drained (13%). They also estimated that 3 % of blanket peatlands had very little or no vegetation cover. Often there are a number of combined land uses that take place concurrently; for example grazing, draining and burning can occur in a single location. In some blanket peatlands, these land uses have combined with erosion (caused by natural processes and exacerbated by land use), to produce highly degraded peatlands (e.g. Pilkington et al. 2015) (Figure 2.3). It is against this background of ecological and hydrological damage, which undermines the wider societal benefits of peatland functioning, that the continuation of some

Figure 2.3. Gully networks and erosion at Bleaklow Head, Peak District National Park, UK. The light areas show where peat has eroded to mineral soils. Map data ©2015 Google Imagery ©2015 DigitalGlobe, Getmapping plc, Infoterra Ltd & Bluesky, The Geoinformation Group.

land–use practices are contested.

Agricultural intensification, from just after the Second World War until the mid–1980s, was the main driver for land use in the UK uplands. Government policy had encouraged, with the use of subsidies, increases in livestock, extensive draining, and afforestation of blanket peatlands (Holden et al. 2007; Condliffe 2009). However, by the mid–1980s there was concern that this single focus on production was causing a loss of biodiversity and damage to peat structure and function (e.g. Anderson and Yalden 1981). Although the SSSI (Sites of Special Scientific Interest) designation was introduced in 1949, it was not until 1986 that Environmentally Sensitive Areas (ESA) were introduced for locations where social and environmental considerations needed to be made alongside farming: the Dark Peak area of the Peak District National Park was designated an ESA in 1988 because of degradation of heather moorland as a result of overgrazing (Condliffe 2009). There were significant reductions in flock sizes as farmers were financially encouraged to remove sheep from their farms (Reed et al. 2009a). The switch from primary producers to stewards of the environment, and the subsequent reduction in subsidy for the former, was not always received well by the farming community. There has also been a reduction in upland farms and the need for upland farmers to develop alternative sources of income (Condliffe 2009). There remain concerns about the vulnerability of rural communities in the uplands as a result of this change in policy emphasis (Defra 2011) because upland farming became unsustainable without external funding via agri-environment schemes (Hubacek et al. 2009, p.298).

Rearing grouse for sports shooting is perhaps currently the most contested land use carried out on blanket peatlands (Lee et al. 2013). The process of rearing economically sufficient numbers of red grouse involves two controversial processes; managed burning, and predator control which in some cases is thought to include the illegal persecution of birds of prey (Young et al. 2010; Elston et al. 2014; Young et al. 2016a). The latter has resulted in several public campaigns to raise awareness of the plight of species such as the hen harrier (e.g. www.raptorsalive.co.uk). In 2012, the RSPB (Royal Society for the Protection of Birds) submitted a complaint to the EU commission against Natural England (the UK government's advisory body on the natural environment and part of the Department for Food and Rural Affairs (Defra)) for failure to protect blanket peatland habitats, under the obligations of Natura 2000, against damage caused by burning. The organisation continues to campaign against managed burning on blanket peatlands.

Managed (rotational or prescribed) burning (muirburn in Scotland) has occurred for more than 150 years (Thompson et al. 1995), and is carried out mainly to improve food availability and habitat condition for rearing red grouse *Lagopus lagopus*. A patchwork of heather *Culluna. vulgaris* stands of varying age are created by burning in rotations of ≈10–25 years (Defra 2007) (Figure 2.4). Burning is carried out according to a voluntary code of practice with the aim of removing only the dwarf shrub canopy and without damaging the moss layer on the peat surface (known as cool burns) (Defra 2007). The continued burning of blanket peatlands is also been contested because of concerns about the negative impact on hydrological and ecological processes and their affect on carbon storage, water quality (Harris et al. 2011; Dixon et al. 2015), and the response of rivers to stormflows.

Because ≈70% of UK drinking water is supplied from upland catchments many of which have headwaters in blanket peatlands (Watts et al. 2001), water companies are interested in the impact that managed burning, and areas of bare peat, have on the amount of dissolved organic carbon released into river catchments which increases water colour and processing costs at treatment plants (Evans et al. 2014). Although a review by Holden et al. (2012) found that the the results of plot and catchment studies were contradictory, and often the effects of burning and vegetation cover could not be disentangled. Contradictory evidence could be used by different groups to challenge the effects of land use, and therefore proposals to change land uses could be contested. Many water companies have either supported the work of peatland restoration organisations, or developed their own restoration schemes such as United Utilities' SCaMP (Sustainable Catchment Management Programme). In addition, as a result of flooding, some local communities have become worried that the loss of peat building species, such as *Sphagnum* spp., and an increase in areas of sparse vegetation cover, as a result of burning, has increased flood risk in vulnerable catchments. There

Figure 2.4. Managed burning on Bleaklow, Peak District National Park, UK. Map data ©2015 Google Imagery ©2015 DigitalGlobe, Getmapping plc, Infoterra Ltd & Bluesky, The Geoinformation Group.

have been public campaigns and protests related to concerns that burning has increased flood risk (http://www.hebdenbridge.co.uk/news/2014/045.html).

Yallop et al. (2006) found that the frequency and extent of burning in the UK uplands had increased between 1970 and 2000 and noted that, in some areas, this increase included the burning (with permission) of sites with SSSI protection. Douglas et al. (2015) reported that managed burning on areas of deep peat, likely to be classified as blanket peat, continued to increase between 2000 and 2011, irrespective of habitat protection status. Nationally, prescribed burning takes place across 8,551 km^2 of uplands ($\approx$16.7 % of this area is burnt), of which 60 % occurs over areas of deep peat (Douglas et al. 2015), which is only likely to increase concerns over the practice. Holden et al. (2015) found that managed burning deepens water tables, changes vegetation, and changes the structure of peat, which utltimately alters the hydrological reponse of blanket peatlands to rainfall events and storms (Section 2.6 this chapter). And the UK Committee on Climate Change has proposed that Natural England should create a programme that reviews consents for burning on protected wetland sites (Committee on Climate Change 2015, p. 16). This evidence suggests that there is the potential for conflict in relation to managed burning to increase.

Peatlands also act as hosts for wind farms because of the exposed nature of many sites (Smith et al. 2014). There have been concerns over the impact of wind farms on bird populations that mirror the concerns of those who oppose offshore wind farms (Percival 2005). Recent evidence suggests that by 2040, there are unlikely to be any net emissions savings from wind farms sited on undegraded blanket peatlands (Smith et al. 2014). In the UK, this conflict has involved the

Scottish Government, Scottish Natural Heritage (the body that advises the Scottish Government on matters of wildlife, habitats, and landscapes), and conservation charities such as The John Muir Trust (www.johnmuirtrust.org), National Trust for Scotland (www.nts.org.uk), and the RSPB. Other opportunities for conflict on blanket peatlands include suggestions to rewild upland areas of the UK (although this may not take place on deep peat), and other users such as walkers and local conservation groups may object to shallower water tables and changes in vegetation that favour mosses rather than heather. I review the evidence for the impact of peatland use on carbon storage in Section 2.6.

Clearly, blanket peatlands provide a number of important ecosystem services over local, national and global scales that relate to climate mitigation (storage of atmospheric carbon dioxide), water provisioning and quality, and livelihoods (Bonn et al. 2009; Parry et al. 2014; Reed et al. 2014a). There has been a shift in the way blanket peatlands are conceptualised: from sources of production to landscapes that provide benefits to society that enhance human well-being and economic activity, but this has increased land–use conflicts (Maltby 2010). Policy changes that began with the development of agri-environment schemes followed by Natura 2000, the Water Framework Directive, and more recently the implementation of the ecosystem services approach (Millennium Ecosystem Assessment 2005) along with blanket peatland restoration projects, have resulted in tension and uncertainty about the future of traditional livelihoods and ways of life. A demonstration of the conflict between environmental issues and farming and game–keeping, was given with the release of Natural England's vision for the future of the English Uplands called Vital Uplands, published in 2009. The document set out to bring together the threads of societal benefits such as carbon, water and recreation with sustainable upland communities but was widely criticised by land–owners and –managers and was withdrawn in 2012 (http://www.theguardian.com/environment/2012/jun/06/attacks-on-landowners-tax-land).

Land–use decisions in blanket peatlands can involve a wide variety of stakeholders. In addition to land owners and their tenants, there are a number of government departments, non—governmental organisations, and advisory bodies. These include national park authorities, Forestry Commision, Defra, Natural England, Scottish Natural Herritage, and the Environment Agency. Added to this there are a number of individuals such as land agents, and organisations who represent land users and conservationists: for example; the National Farmers Union, Moorland Association, Game and Wildlife Conservation Trust, and The Heather Trust, who can act as important lobby groups (e.g. Bullock et al. 2012). Previously, the various representative bodies of farmers and gamekeepers (such as the National Farmers Union and Moorland Association) have highlighted the fact that the UK uplands are a product of past management and that continued management is needed to prevent further loss of heather, and the aesthetic value of the uplands enjoyed by many for recreational purposes. A model by Chapman et al. (2009b) supports this view: they showed that simulations where managed burning was banned resulted in reductions in dwarf shrub cover. However, the objectives of traditional peatland users often conflicts with those of water companies and conservationists who have aims based around carbon, water, and biodiversity. These contested objectives extend to the government agencies who are responsible ensuring that the aims of the legislation that underpins Natura 2000, and obligations of the Aichi targets (Section 2.2.1) are accomplished, such as Natural England.

The conflicting viewpoints of stakeholders bring together two key issues: land use; (1) provides support for local livelihoods in blanket peatlands, but has changed the ecological and hydrological functioning of peat soils; and (2) can negatively affect the wider national and international aims of improved water quality and carbon storage (for climate change mitigation). These three objectives (livelihoods, water, and carbon), are an integral part of the Aichi targets where the focus in on the 'fair and equitable' (Secretariat of the CBD 2010) use of ecosystem resources. However, further opportunities for conflict could arise because landowners may feel that they are not adequately compensated for providing benefits to society whilst being restricted from carrying out their own objectives (Reed et al. 2014a): but changes to agri–environment schemes and payment for ecosystem services schemes, such as the Peatland Code (Reed et al. 2013a; Bonn et al. 2014), may provide opportunities to address these issues. Although there is an emphasis on returning ecosystem function to blanket peatlands, conflict resulting from land use could prevent or limit how the Aichi targets are achieved (Young et al. 2016b). Some in land–use communities have questioned the indirect effect that peatland restoration will have on livelihoods, and have called for the development of shared objectives and the inclusion of local knowledge to inform restoration decisions (*Peatland restoration - what's in it for me?* 2015). This co–development of land–use objectives could reduce conflict if decision–makers are committed to embedding the knowledge of land managers into the decision–making process (Young et al. 2010), and also provides an important opportunity to investigate how stakeholder knowledge could be brought together to achieve this aim in blanket peatlands.

In light of the differences between land–use objectives for blanket peatlands and the resulting conflict, the question remains; how can these groups with conflicting viewpoints on managing blanket peatlands be brought together to achieve equitable solutions for local communities and wider society, rather than win–lose or lose–lose outcomes? (*sensu* Maltby 2010; Redpath et al. 2013) (Figure 2.5). The Convention on Biological Diversity specifically calls for the invlovement of local communities in decision–making along with the consideration of all relevant knowledge (including scientific, local and traditional sources) (Secretariat of the CBD 2010). In Europe, the right of stakeholders to participate in environmental decisions is a matter of legislation, specified in

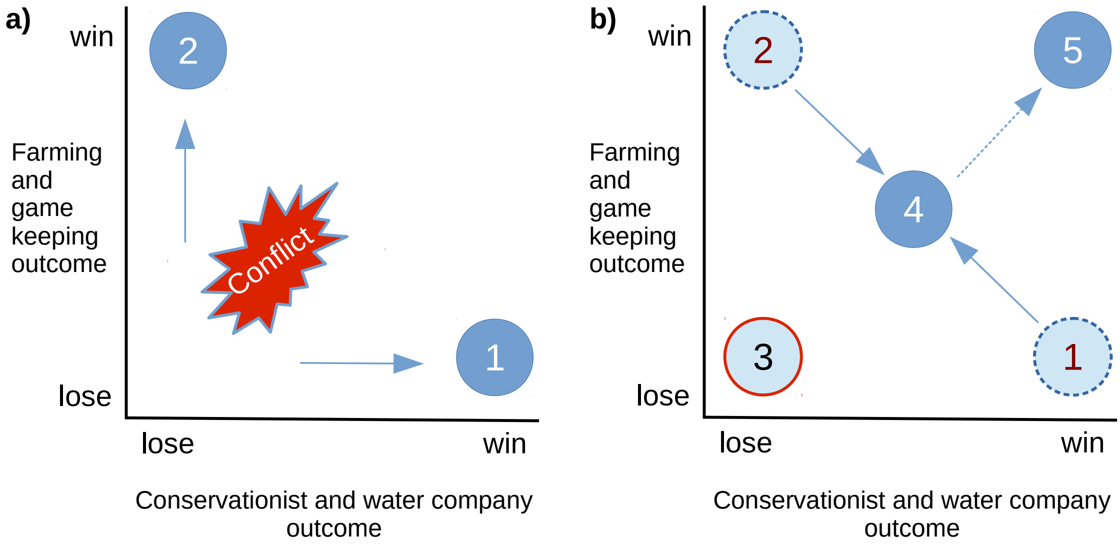


Figure 2.5. Blanket peatland contested land use. a) The approach of both groups (1) & (2) is to try to win the conflict with little compromise. **b)** During the conflict resolution process both parties recognise the conflict as a shared problem by identifying lose–lose outcomes (3) and potential trade-offs (4) if a win–win outcome (5) is not possible. The two conflicting groups shown here were identified from stakeholder network analysis in the Peak District National Park (Prell et al. 2008; Reed et al. 2013b). Redrawn and adapted from Redpath et al. (2013).

the Public Participation Directive (2003/35/EU) to improve both public participation in environmental matters and the effectiveness of environmental policy implementation (Newig and Fritsch 2009). Maltby (2010) related the 12 principles of the ecosystem approach from the Convention on Biological Diversity to blanket peatland land use and states that ecosystem management "must involve" the relevant communities. I suggest that four aspects of this approach are key to stakeholder participation: (1) decision–making processes and the knowledge of all relevant stakeholders should be coupled for the purpose of; (2) the conservation of peatland structure and function; (3) the management of peatlands within functional limits (i.e. rather than over–exploitation); and (4) to balance appropriately the conservation and economic use of peatlands (Maltby 2010). Although these conflicts may never be fully resolved (Redpath et al. 2013), Maltby (2010) argues that there should be four aims for peatland use:

1. Enhance the vitality of the rural economy
2. Maintain or enhance the contribution to environmental security (the mitigation of climate change and supply of high-quality water)
3. Support the well–being of the wider local and national communities
4. Safeguard important biodiversity and cultural heritage

Translating these aims to action on the ground will require the collaboration of land owners, farmers, gamekeepers, conservationists, water companies, non–governmental organisations, government agencies, and local communities in participatory processes to; (1) understand the impact of social and ecological interactions on blanket peatland carbon storage; (2) explore how to achieve land–use objectives whilst repairing damaged peatland function; and (3) determine the impact of land use and restoration on long–term carbon storage.

Studies continue to investigate and evaluate the impact of land use and restoration on blanket peatlands using field observations and modelling (e.g. Douglas et al. 2015; Holden et al. 2015; Pilkington et al. 2015). However, given that the restoration of peatlands is a priority (Bonn et al. 2014; Committee on Climate Change 2015), and the engagement of stakeholders in land–use decision–making processes is seen as essential (Maltby 2010), it is notable that there are few studies that combine stakeholder knowledge in collaborative processes to co–develop an understanding of blanket peatlands as complex systems, and of how the network of social and ecological interactions affect carbon storage. Reed et al. (2013b) reported on a process used to engage blanket peatland stakeholders that combined a conceptual model of blanket peatlands with process–based models to determine the biophyscial outcomes of two scenarios (discussed in more detail in Section 2.5.4). However, although the conceptual model was used to develop scenarios, the knowledge embedded in the model was not used directly to determine how stakeholders' combined understanding of the complex web of interactions might affect carbon storage. There are also a lack of studies that identify those interactions that blanket peatland stakeholders perceive as most important, and determine, in collaborative processes, how those interactions could be managed to achieve land–use objectives. Such an approach could be used to identify where there are differences in perceptions, build a common understanding of interactions, and highlight the implications of land–use objectives on carbon storage, based on stakeholder knowledge.

Since the work of Reed et al. (2013b), studies that engage blanket peatland stakeholders have often focussed on ecosystem service assessments (e.g. Drew et al. 2013; Pilkington et al. 2015). These studies are important because they provide a link between ecosystem system services, the impact of management practices, and the agri–environment schemes that often pay for management on peatlands. However, in light of the continued conflict that characterises some management practices such as managed burning (which has continued to increase; Douglas et al. 2015), I propose that an approach that incorporates stakeholder knowledge about social and ecological interactions that could be integrated into participatory processes, to support decision–making alongside ecosystem services assessments, may provide insights into how stakeholders could work together collaboratively to achieve land–use objectives.

Communities throughout the globe are being challenged by the drive to restore function to degraded ecosystems. Restoration can lead to conflict if communities become disadvantaged; conservation appears to threaten local livelihoods or traditions; or if suitable, viable, alternatives are not provided (Chokkalinga et al. 2005; Redpath et al. 2013; Noordwijk et al. 2014; *Peatland restoration - what's in it for me?* 2015). Although top–down policy can set the context and targets for restoration, previous attempts have failed to halt continued degradation (e.g. JNCC and Defra 2012). Therefore bottom–up approaches that connect disparate knowledge sources about the web of interactions that characterise these complex systems, can provide a basis for collaborative working, trust building (e.g. Biggs et al. 2011; Corral-Quintana et al. 2016; Young et al. 2016a), and be used to inform land-use and restoration decisions in peatlands.

2.4 Peatlands as complex systems

2.4.1 Introduction

Many ecological, biological and human systems are complex systems characterised by the low–level interaction of components that determines the emergent properties of the system (Levin 1998). Some of the interactions are the positive and negative feedbacks that take place within and across spatial and temporal scales. Complex systems exhibit self–organisation that leads to spatial heterogeneity, and respond to autogenic (internal) processes and allogenic (external) forcing in a non-linear, and sometimes hysteretic (Section 2.4.2) manner so that the future development of the system emerges from these past interactions (Levin 1998; Liu et al. 2007; Gao et al. 2016). Belyea and Baird (2006) identified four concepts from Levin's work (1998) on complex systems to be characteristic of peatlands: (1) spatial heterogeneity – differences in microform properties such as vegetation composition and hydraulic conductivity become part of the ecological memory of the peat mass and are sometimes evident as patterns on the peatland surface; (2) localised flows – microforms interact across small spatial scales through the flow of water, energy and nutrients; (3) self-organising structure – local interaction aggregates microforms and drives the structure of the whole peatland; and (4) non-linear interactions – depending on past development trajectory and the dominance of positive or negative feedback processes, there may be abrupt response to climate, nutrient enrichment or land–use. Several studies have suggested that this complex adaptive behaviour is an inherent feature of peatlands (Lamers et al. 2000; Belyea et al. 2004; Eppinga et al. 2009; Baird et al. 2011; Morris et al. 2011b; Ireland and Booth 2012; Ramirez et al. 2015).

2.4.2 Non–linearity

Belyea (2009) proposed a number of non-linear dynamics in peatlands where slow processes dominate and minimise the impact of external forcing and fast processes occasionally dominate and force abrupt change (Table 2.1). Slow and fast dynamics are concepts of resilience theory

Table 2.1. Examples of proposed peatland stabilising and destabilising processes*

Stabilising processes (slow)	Destabilising (fast)
Self regulating water table via the mechanism of transmissivity (depth integrated K_{sat})†	Removal of water and concentration of resources by vascular plants on ridges
Pool expansion and contraction to modulate water losses on an ecosystem scale	Unidirectional pool expansion into large ponds and lakes (long-term impact with increased climate wetness) that can lead to collapses in ridge structures and abrupt changes in hydrological regimes
Expansion and contraction of microforms reduces the magnitude of increased climate wetness. Hummocks reduce runoff losses. Hollows increase runoff losses	

* Source: Belyea (2009)
† K_{sat} = saturated hydraulic conductivity

(Holling 1973; May 1977) that relate to stabilising (slow; negative feedback) and destabilising (fast; positive feedback) processes (Ludwig et al. 1997; Gunderson and Holling 2002; Belyea 2009). The continued gradual loss of resilience of slow processes can result in an abrupt shift to an alternative attractor or state (Ludwig et al. 1997; Scheffer et al. 2001). Examples of shifts to alternative states in peatlands include; between a moss dominated and vascular dominated peatland (Lamers et al. 2000; Eppinga et al. 2009); a fen–bog transition (Tahvanainen 2011); or between a permafrost peatland and arctic fen (Swindles et al. 2015). Shifts to alternative states may persist even after the conditions that were present before the shift are restored because of hysteresis (Scheffer et al. 2001; Kéfi et al. 2012).

Hysteresis means that a system may not return to its pre–degraded state even after the disturbance regime has been reduced or removed because a new system of negative feedbacks reinforces the degraded state (i.e. the new state is resilient). For example, Isbell et al. (2013) reported that nutrient enrichment had resulted in a low biodiversity grassland, but the loss of biodiversity was not reversed even after 20 years of reductions in nitrogen. Similar relationships that lead to the collapse of resources have also been proposed for highly interconnected social systems (Helbing 2013). Shifts can occur following a period of gradual change so that a relatively small disturbance causes a sudden shift; or following a large shock to the system (Hastings and Wysham 2010; Scheffer et al. 2012). Because such switches are often associated with a collapse in resources, they have negative consequences. But shifts can take place in positive direction, leading to the restoration of ecosystem function (Eppinga et al. 2009; Kéfi et al. 2012). Evans et al. (2014) suggested that blanket peat erosion in the South Pennines is likely to be irreversible without intervention to shift degraded peatlands into a functioning state by revegetating areas of bare peat, and blocking eroding gullies.

2.4.3 Feedback mechanisms

There are a number of important negative feedback mechanisms in peatlands (Waddington et al. 2015). Kettridge and Waddington (2014) identified a negative feedback mechanism that reduces evaporative losses by increasing the surface resistance of mosses with increasing water-table depth; effectively disconnecting the peatland from changes in climate. Others have identified important negative feedback processes such as the interaction of peat formation and water-table depth that maintains a relatively stable oxic layer thickness which helps to explain the steady long-term rates of peat formation under varying climate (Belyea and Clymo 2001). Wang et al. (2015) have proposed a negative feedback mechanism in boreal peatlands (pocosins in the southwestern United States) causes an increase in high–phenolic shrubs when water tables deepen, and concluded that increasing shrub cover is an adaptive response to drought or warming that increases peat accumulation rates, and protects current carbon stocks from decomposition. But because pocosin peatlands have developed in a location that often experiences drought, it is not known if these results apply more widely to other peatland types.

Waddington et al. (2015) developed a conceptual model that describes seven, mainly hydrological autogenic feedback routines comprised of a number of positive and negative feedback mechanisms and their interactions on a grand scale (Figure 2.6). The authors called for similar conceptual models of other peatland feedback mechanisms to be developed in order to improve understanding of peatland response to climate and land use. There seems to be no reason why the development of these feedback networks cannot incorporate the experiential knowledge of practitioners and land users captured during participatory processes. The effect of these negative feedbacks between internal structure and autogenic processes can isolate the peatlands from external forcing (Swindles et al. 2012; Morris et al. 2015a). In a modelling study, Ramirez et al. (2015) found that peat pore structure could decouple the emission of methane from the enviromental forcing events that caused the bubble production. Because less porous peat captured bubbles of methane, they did not escape the surface of the peat at the same time as the forcing, effectively decoupling the signal from the event. This demonstrates the importance of taking peat structure into account in models, and can explain some of the site–to–site differences in observed ebullition, which makes it difficult to successfully make statistical predictions about these events (Ramirez et al. 2015).

The response of peatlands to climate is likely to vary from location to location. In future, climatic conditions in the mid northern latitudes are likely to be too warm for peat growth (Gallego-Sala and Prentice 2012; Charman et al. 2015), which may increase the vulnerability of blanket peatlands in the south of their UK distribution (e.g. Li et al. 2015). And whereas warming

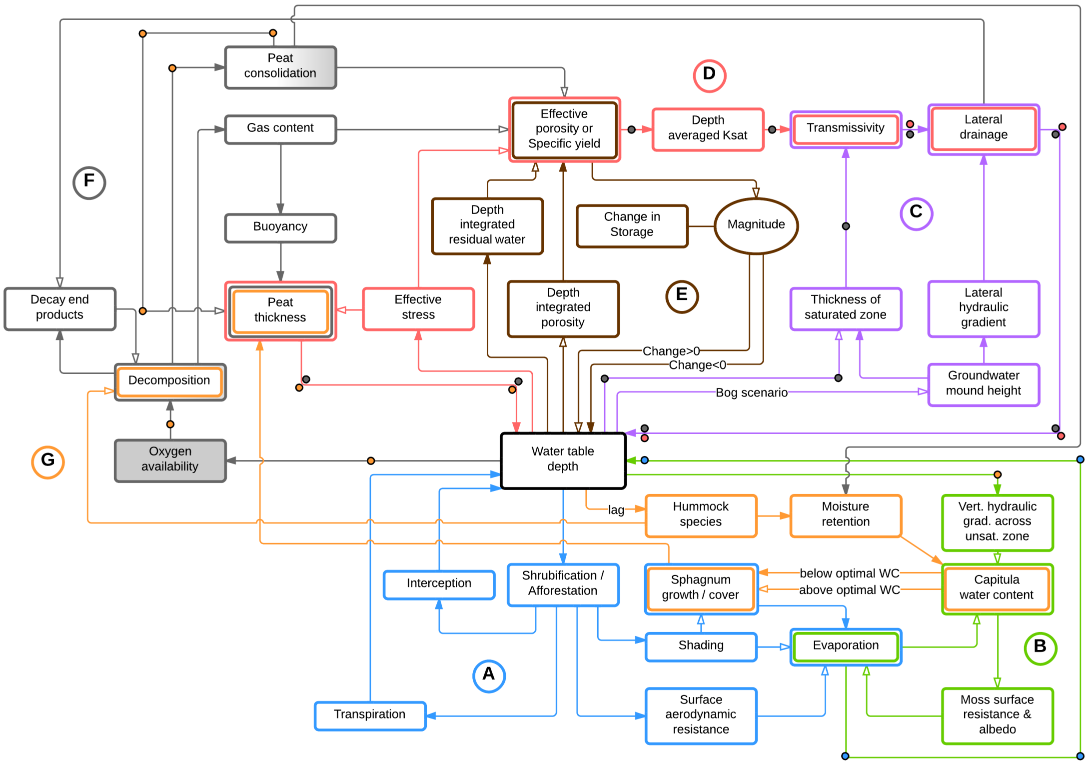


Figure 2.6. Hydrological feedbacks in northern peatlands. Feedback A: Water-table depth–afforesstation/increase in shrubs. Feedback B: Water-table depth–moss surface resitance and albedo. Feedback C: Water-table depth–transmissivity. Feedback D: Water-table depth–peat deformation. Feedback E: Water-table depth–peat formation. Feedback F: Water-table depth–peat decomposition. Feedback G: Water-table depth–moss productivity. Solid and outline arrow heads represent positive and negative relationships (from Waddington et al. 2015).

may cause some northern latitude peatlands to increase rates of peat accumulation, the carbon stores of high northern latitude permafrost peatlands could be destabilised as a result of dominant positive feedback mechanisms (Swindles et al. 2015). Fenner and Freeman (2011) proposed that warming induced drought events and subsequent rewetting would degrade peatland carbon stocks by increasing the decomposition of organic matter. A peatland model developed by Ise et al. (2008) suggested large quantities of carbon (≈86% for deep peat) would be lost to the atmosphere because an of an increase in temperature However, Laiho (2006) argued that these effects were likely to dominate only in the short–term until the autogenic processes that drive carbon accumulation have adapted, but with the caveat that this will vary depending on the severity of forcing, type of peatland, and location.

Contrary to the conclusions of Ise et al. (2008), in a study that used used peat core data from ninety sites in the northern hemisphere, Charman et al. (2013) concluded that peatlands have previously exhibited a small negative climatic feedback when temperature increased due to an increase in net peat accumulation. Model simulations by Morris et al. (2011b) that investigated ecological and hydrological feedbacks, found that peatlands may respond only weakly to allogenic forcing driven by increases in rainfall; a conclusion supported by Swindles et al. (2012). But the impact of external forcing on peatland carbon stores is also dependant on the nature and magnitude of the forcing event. Using a version of the **`Digibog`** peatland development model, Morris et al. (2015a) showed that changes in rainfall prompted a largely homeostatic recovery to pre–disturbance water–table regimes in raised bogs, but changes in temperature caused the peatland to permanently shift to a drier state. They concluded that the autogenic mechanisms that enable peatlands to respond homeostatically to rainfall do not enable water tables to adapt in the same way in the case of temperature. They also demonstrated that the speed and magnitude of these changes plays an important role in the resilience of peatlands.

Even though climate is one of the key factors in peatland initiation and development, the current peatland development models reviewed in this chapter (see Section 2.8.3), including the model of Morris et al. (2015a), use climate inputs that have been lumped into annual values. As yet, the impact of this averaging on modelled peat accumulation is not known, but may be important especially when simulating blanket peatland development because the annual distribution of rainfall and temperature have been identified as fundamental conditions (Lindsay et al. 1988). In addition, I could find no evidence of a peatland development model being used to explore the impact of land use or restoration on blanket peatland carbon storage.

2.4.4 Process–based models of complex systems

Process–based models of complex systems are built from the bottom–up interactions between low level components (Evans et al. 2012). They do not rely on statistical projections but on an understanding of how system processes take place (Evans et al. 2012; Cuddington et al. 2013). Process–based models can therefore be used to predict how the system might develop under futures that have not yet been observed. Since Box (1976) famously said *"..all models are wrong.."*, there is clearly a need to identify the relationships that are important to overall system behaviour whilst at the same time being mindful that the model itself is a simplification of reality (Grimm et al. 2005). Box (1976) also cautions against overly elaborate models because they will not necessarily be more "right". Here I discuss the use of process–based models to understand peatland dynamics.

Models of peatlands as complex systems can help develop an understanding of responses to autogenic processes and allogenic forcing and help tackle questions about the impact of long-term (centennial and millennial timescales) climate–induced feedbacks and the impact of land uses or restoration on future peatland carbon storage. Surface patterns, such as the distribution of water–table depth across the slope of a blanket peatland can be measured, but are difficult to use as a predictor of future water tables, or water tables in similar slope locations, because they are the product of below surface peat properties (hydraulic conductivity and drainable porosity) that may have been developed under different conditions (Baird 2014). These properties are the result of low–level interactions that decompose peat and affect water movement (Holden 2005b). Therefore, predicting the outcome of how past and future peatland carbon stores will respond to changes land use regimes, or restoration approaches is challenging: but many studies do not take into account these low–level properties. For example, the study of gullies and gully blocking by Allott et al. (2009) who calculated a wetness index based on topography and water table drawdown at gully–edges but did not take peat structure into account.

Process based models have been used to simulate peatland response to climate and land use (Ise et al. 2008; Frolking et al. 2010; Heinemeyer et al. 2010; Morris et al. 2015a). Modellers seek to represent the autogenic dynamics of litter production, litter and peat decomposition, water-table position, hydraulic conductivity, peat depth, and the effect of external variables such as net rainfall (rainfall minus evapotranspiration), temperature, and land use. Although some recent peatland development models explicitly represent the four aspects of complex systems identified by Belyea and Baird (2006) (Section 2.4.1), including the key ecological and hydrological feedback between decomposition and hydraulic conductivity (e.g. Frolking et al. 2010; Baird et al. 2011; Morris et al. 2011b), in others some of these relationships are missing (e.g. Ise et al. 2008; Heinemeyer et al. 2010). Models that do not include feedback between decomposition and hydraulic conductivity can show bistable behaviour (e.g. Hilbert et al. 2000) that disappears if the feedback is added to the model (Baird et al. 2011; Morris et al. 2011b).

There are additional gaps in the literature in relation to simulating peatland development, especially in relation to blanket peatlands. There has been limited research into peatland development using 2D or 3D process–based models that incorporate autogenic behaviour (e.g. Morris et al. 2011a). For example, although the model of Morris et al. (2015a) explicitly accounted for vertical heterogeneity in a peat column (1D), lateral water movement was implied. As a result, a spatially heterogeneous structure was not developed across the peatland, a feature which is apparent in real peatlands and likely to impact peat accumulation (Belyea and Baird 2006). Other models are also based on a single column of peat (1D) (e.g. Frolking et al. 2010) or on a series of hydrologically disconnected columns (Heinemeyer et al. 2010). Blanket peat accumulation has not yet been simulated on slopes as a series of 2D or 3D hydrologically connected columns that incorporate the feedback between water–table depth, decomposition, and hydraulic conductivity. This feedback is missing in previous blanket peat accumulation models (e.g. Heinemeyer et al. 2010), which is likely to alter how the modelled peatland responds to external forcing. I review three current peatland development models in Section 2.8.3 .

2.5 Addressing land-use conflict through participatory modelling

2.5.1 Background

Stakeholders often take part in participatory processes that aim to explore potential solutions to land–use and conservation conflicts (e.g. Tolvanen et al. 2013; Elston et al. 2014; Butler et al. 2015). There are large number of studies that suggest that collaboration between stakeholders groups can; improve the uptake and implementation of research results; build trust, help to reduce conflict, improve the legitimacy, quality, and ownership of decisions; enhance social learning; and reduce the costs of policy implementation (e.g. Reed 2008; Danielsen et al. 2010; Chapin et al. 2012; Wiek et al. 2012; Elston et al. 2014; Gramberger et al. 2014; Madden and McQuinn 2014; Young et al. 2016a).

Reed (2008) reviewed the development of, and evidence for, stakeholder participation in environmental decision–making. He found that whilst many claims had not been tested, there was evidence to suggest stakeholder participation can lead to improved decisions, but that it should be viewed as a process that enables stakeholders to fairly contribute their knowledge to the decision–making process. Integrating local qualitative knowledge into the decision–making process can increase the acceptance of results by participants, but conflicts are likely to remain unresolved if the agencies responsible for decision–making are unwilling to incorporate the outputs of participation into implementation plans (Young et al. 2016a). One approach to the bottom–up integration of social and ecological knowledge within stakeholder engagement processes is participatory modelling.

It is common for modellers to engage with stakeholders. In many cases modellers talk to experts who can help specify parameters for model inputs or develop scenarios for model testing and experimentation (e.g. Chapman et al. 2009b; Elston et al. 2014; Wood et al. 2015). Often the aim of these models is to inform management or policy decisions in relation to conservation or land use. However, process–based simulation models can be seen as ‘black boxes’, which can make acceptance of model outputs difficult especially when used to communicate results to a group with varied backgrounds (most of who are unlikely to be modellers) (Lorscheid et al. 2012). This could be a major obstacle to using these models to help determine how to achieve land–use objectives with groups of stakeholders who contest land use; and so alternative approaches to modelling complex systems with stakeholders may be needed that can, if needed, be coupled with process–based models as suggested by Prell et al. (2007).

Land–use decisions could negatively affect the livelihoods of some stakeholders, and, as a result they may contest the evidence used in the decision–making process (Saarikoski et al.

2013a). Incorporating social and ecological knowledge from stakeholders when conceptualising and building models may help to maintain transparency, and the credibility of results (Durance and Godet 2010; Voinov and Bousquet 2010). Voinov and Bousquet (2010) proposed that *"a truly participatory effort would engage stakeholders in an interactive and iterative mode, where the flow of information is arranged in both directions: from the stakeholders to researchers (modellers) and vice versa, from modellers back to stakeholders in a process of learning (co-learning)"*. Model building in this way also provides the opportunity for information to move between different stakeholder groups to build an understanding of the activities and motivations of various groups so the process is multi– and not just bidirectional.

Although the nature of the engagement with stakeholders will differ from project to project, and hence the modelling process itself, there are a number of iterative stages that can be used to co-develop a model with stakeholders during a participatory process (Figure 2.7) (Voinov and Bousquet 2010). Proponents of participatory modelling argue that the outputs of the models themselves need not be the main goal of the process, and perhaps the most significant outcome is that stakeholders work together which my improve collaborative decision–making processes (Penn et al. 2013; Bommel et al. 2014). However, combinations of highly uncertain futures, competing interests between stakeholders and the complexity of social–ecological systems (wicked problems – *sensu* Rittel and Webber 1973) mean that land–use decisions are unlikely to be based on model outputs alone (Bommel et al. 2014; Wood et al. 2015).

2.5.2 Participatory modelling approaches

Clearly the main goal of participatory modelling is the engagement of stakeholders in the modelling process. Workshops are commonly used to gather information from groups, but interviews with individuals are also used for model building (e.g. Dougill et al. 2006; Scott et al. 2013; Christen et al. 2015), whereas workshops are obviously needed where the aim is for stakeholders to collaborate to build models, develop scenarios or assess results.

Most models could be used to some degree in a participatory process, where stakeholder help develop model parameters, or make an assessment of model outputs. For example, Chapman et al. (2009b) developed a model to simulate the coupled dynamics of vegetation and management on moorlands, the model included management regimes that were developed by stakeholders as part of a wider participatory process. However, in the context of conflicts there is a need for transparency, and so it may be beneficial for stakeholders to be involved in model building (Reed et al. 2009b) as well as selecting parameter sets for experiments. In the UK, there is an ongoing conflict between grouse moor managers and conservationists about the potential threat to hen harrier populations as a result of predator control. Elston et al. (2014) worked with a group of stakeholders to develop

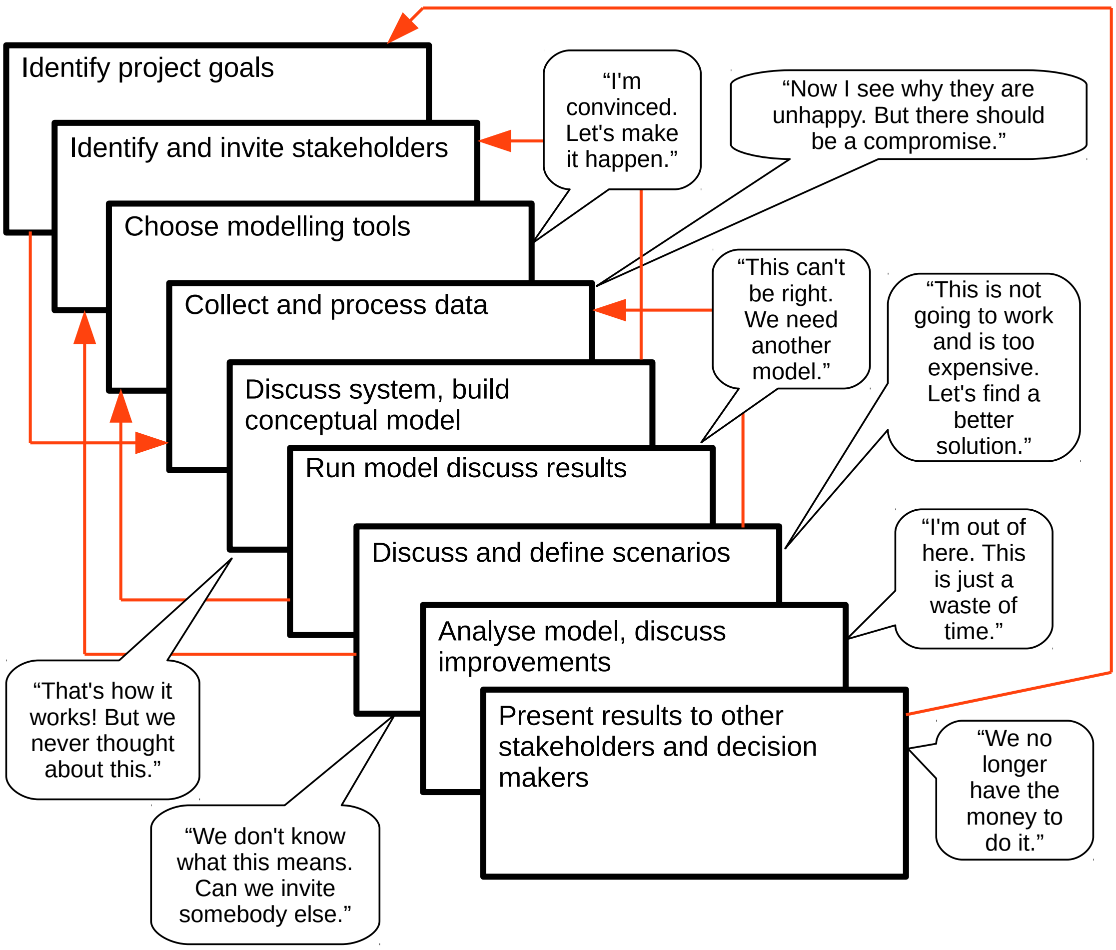


Figure 2.7. Generic stages of a participatory modelling process. The process can proceed in any order and there may be several, perhaps repeated iterations between stages. Where appropriate, stages can be combined or missed completely, for example if goals are achieved or management decisions agreed. (Redrawn from Voinov and Bousquet 2010).

a model and agree on data sets so that model outputs would be accepted by all parties. Next I briefly review four approaches to modelling complex systems that can enable stakeholders to have a significant input to the model building process.

Agent–based models

Agent–based models can be used to represent an array of complex systems by incorporating the network of bottom–up interactions between system components, processes, or individuals (including environmental components) in order to predict system behaviour (Grimm et al. 2005). Agent–based models have been used to model many different complex systems in relation to, for example, grazing, habitat loss, wildfire regimes, changes in land–use policy (Millington et al. 2011; Bommel et al. 2014; Wood et al. 2015), or the dynamics of individuals in cities (Heppenstall et al. 2016). Individual components are autonomous and exhibit adaptive behaviour that affects the overall development and trajectory of the system (North 2014), and so display properties of complex systems (Levin 1998; Grimm et al. 2005). Furthermore, the interactions between individual components can include feedback loops and the nonlinearity often observed in complex systems (Levin 1998).

Agent–based models provide significant opportunity for stakeholder participation in, for example model conceptualisation, development, data identification and parameter selection (Voinov and Bousquet 2010; Wood et al. 2015). Stakeholders can be involved in determining how agents interact, and in assessing the outputs of models (e.g. Parker et al. 2003; Millington et al. 2011; Bommel et al. 2014). Although it has also been suggested that agent–based models can be 'black boxes' (Topping et al. 2010), Wood et al. (2015) suggested that this need not be the case if reporting is clear, and model outputs are presented in ways that are easy for stakeholders to visualise.

Bayesian belief networks

Bayesian belief networks are comprised of a causal network and a set of combined probabilities that describe the interactions of the network (e.g. Chan et al. 2010). Both the development of the causal network and details of the interactions between components is ideally suited to participatory approaches such as workshops (Chan et al. 2010; Maxwell et al. 2015). However, the causal network used is a directed acyclic graph (Bashari et al. 2008) which means that none of the available paths from a node can be followed to return to that node (i.e. feedback). This rules out the inclusion of feedback loops (Figure 2.8) which are an important class of interaction within complex systems (Levin1998), and known to be found in peatlands in particular (Belyea and Baird 2006; Waddington et al. 2015).

Systems dynamics

Systems dynamics promotes thinking of systems as a whole and is used to understand the development of a system's structure over time. *"The aim of systems dynamics modelling is to explain behaviour by providing a causal theory, and then to use that theory as the basis for designing policy interventions into the system structure which then change the resulting behaviour and improve performance"* (Lane 2008). Often used within (but not restricted to) corporations, stakeholders are frequently involved in group model building: the process starts with a problem that needs to be solved, and with the assumption that at least some of the information required to solve the problem will be held by stakeholders as mental models (Doyle and Ford 1998; Scott et al. 2013). Systems dynamics research uses causal loop diagrams that incorporate stocks and flows diagrams to communicate the behaviour of the system of interest, and to identify assumptions for mathematical

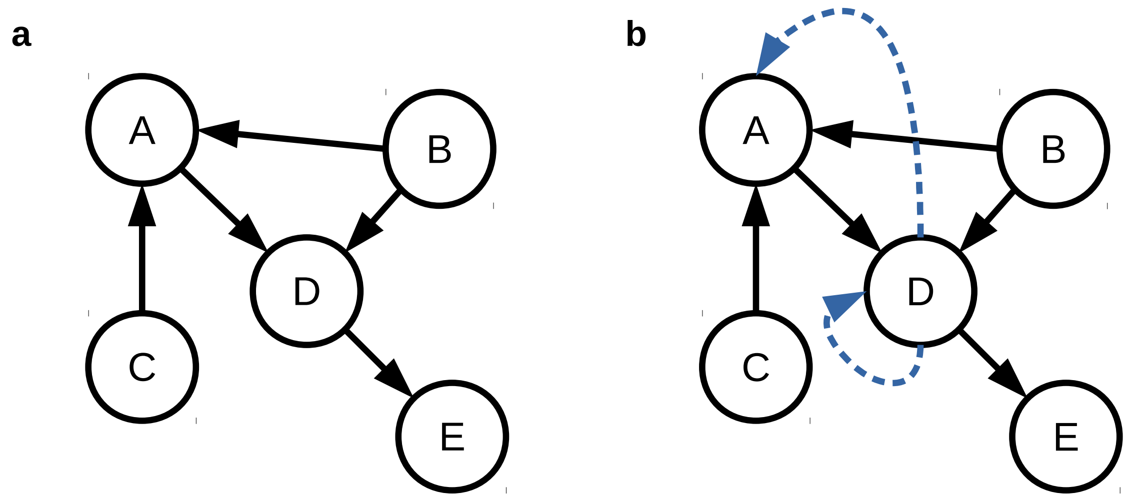


Figure 2.8. Difference between directed acylic graphs and directed graphs. **a** Directed acyclic graph. Nodes and links are arranged without returning loops (e.g. A $\Rightarrow$ D $\not\Rightarrow$ A). **b** Directed graph. Nodes can be returned to either directly or indirectly (e.g. A $\Rightarrow$ D $\Rightarrow$ A). Self–loops are also permitted (e.g. D $\Rightarrow$ D), which could represent certain types of vegetation that engineer their own environment (Eppinga et al. 2009).

modelling of the system (Lane 2008). However, although systems dynamics can be used to model many systems and processes, increased complexity increases the time, effort and difficulty involved in using this process (Voinov and Bousquet 2010).

Network analysis

Many social, ecological, biological and anthropogenic systems can be represented by networks. Networks have been used to map the communication links between different groups of stakeholders (Bodin and Crona 2011), the interactions between two proteins (Barabási and Oltvai 2004), the mutualistic dependencies of plant and animal communities (Suweis et al. 2013), and food webs (Dunne et al. 2002). Models of networks can contain many thousands of nodes and hundreds of thousands of links such as the world–wide–web (Albert et al. 1999). Both the dynamics and structure of networks can be studied depending on the information associated with nodes and links (Strogatz 2001; Boccaletti et al. 2006; Gao et al. 2016) (Figure 2.9). Network structure and processes can be used to understand many different system properties such as the rate at which a disease is likely to spread, the robustness of communications networks (Barabási 2009; Morone and Makse 2015), or the dynamics of a traffic system (Li et al. 2014). Furthermore, networks are transparent and can be easily visualised: they are therefore suited to participatory processes so that; stakeholders can develop models; for the communication of results; and to discuss how system components interact (Krzywinski et al. 2012; Pocock et al. 2016).

Prager and Pfeifer (2015) used network structure to understand if rainwater management in Ethiopia could be improved. They developed networks based on surveys of smallholder farmers,

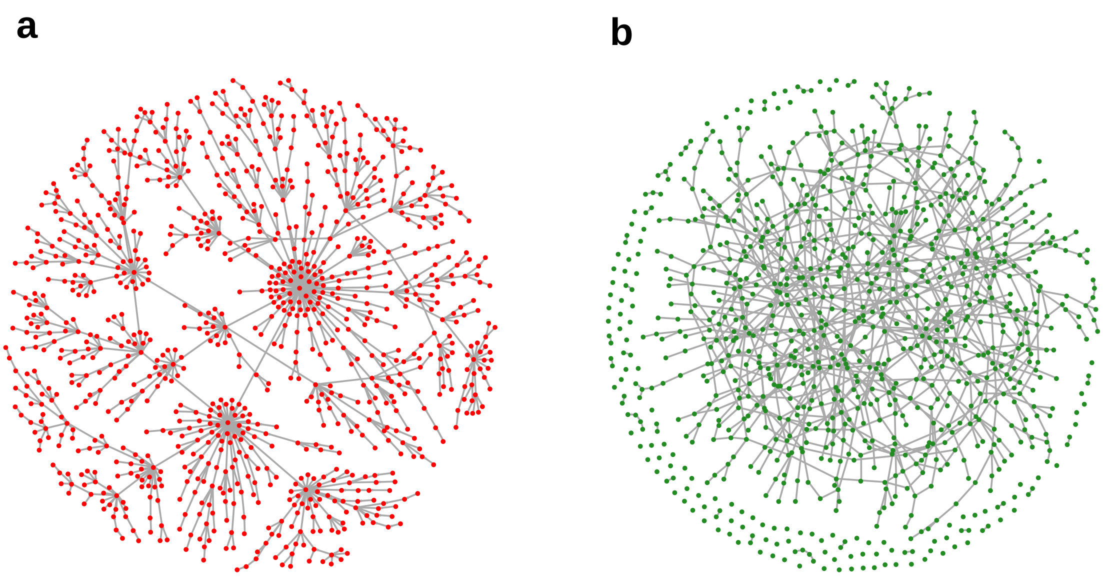


Figure 2.9. Examples of network structure. **a** Scale–free. New nodes are preferentially attached to existing nodes. A few nodes (hubs) have many connections whilst the majority of nodes have a few connections. Scale–free networks are found in the real–world. In this example, the maximum number of connections for a single node=82 and the mean number of connections per node=1.99. **b** Erdös-Réyni random network model. Isolated nodes are connected by the probability (p) that two nodes are linked. In this example, p=0.035, the maximum number of connections for a single node=10 and the mean number of connections per node=2.07. The number of nodes in both networks=1000. Networks generated using the igraph package (Csardi and Nepusz 2006) in R (R Core Team 2015).

vegetation condition, and the physical characteristics of each surveyed plot. They coupled ecological and plot information to develop a social–ecological network of smallholders which identified the influence of different smallholders on the rainwater supply for agriculture. The study demonstrated that network structure can be a useful approach to understand better the management of complex social–ecological systems. Fuzzy cognitive maps (FCM) are a type of cognitive map (directed weighted network; Section 2.8.2) that have been used to develop an understanding of system interactions and dynamics based on stakeholder knowledge (e.g. Kontogianni et al. 2012; Christen et al. 2015). In a participatory study about a bio–based economy in the UK, Penn et al. (2014) incorporated the structure of a FCM. They first developed a FCM with stakeholders, and then identified those nodes that provided structural control of the network (driver nodes), by extending the work of Liu et al. (2011). In theory the driver nodes can be manipulated to 'steer' the network to a desired state. They then asked stakeholders to determine which driver nodes could be classed as controllable and generated a series of control configurations for the FCM. However, the approach was not further applied to determine how the driver nodes, within control configurations, should be managed to achieve specific objectives for the system.

2.5.3 *Mental and cognitive models of complex systems*

Those involved in land–use conflicts may have different mental models of how the social and ecological components of an ecosystem interact (Biggs et al. 2011). These interactions can be seen as causal assertions or beliefs, *'A causes D'* (Figure 2.8a), where an individual's overall knowledge of a system is about how they understand these components to be organised and interconnected (Doyle and Ford 1998). Participants' mental models can be used to aid the development of simulation models for each of the approaches in Section 2.5.2. Doyle and Ford (1998) developed the following definition of mental models of dynamic systems in order to remove some of the ambiguity that had developed around the subject: *"a mental model of a dynamic system is a relatively enduring and accessible, but limited, internal conceptual representation of an external system whose structure maintains the perceived structure of that system"*. They differentiate this internal (mental) model of a system to the externalised model that is elicited during participatory processes which they identify as a cognitive structure (map) (e.g. Axelrod 1976).

Although mental models can continue to be developed and refined with experience and expertise, evidence indicates that when externalised, descriptions of system functions used for modelling purposes are likely to contain errors and be incomplete (Doyle and Ford 1998; Lynam and Brown 2011). Even so, mental models are useful in situations of conflict between stakeholders because working together on representations of how components of a system interact encourages improved communication, brings together different sources of knowledge and develops shared ownership (Biggs et al. 2011). Of course, there may be instances where differences between the competing aims, values or beliefs of stakeholders are so great that even partial resolution of a conflict may not be possible (Redpath et al. 2013). That said, the benefits of working with mental models in situations of land–use conflict appear to be worth pursuing (Jones et al. 2011), and a number of different approaches have been developed to elicit them (e.g. Axelrod 1976; Kosko 1988; Hodgson 1992; Rouwette and Vennix 2006; Etienne et al. 2011) (refer to Section 2.8).

Group building of conceptual models can lead to alignment of participants' mental models, even for short–term engagements, (Rouwette and Vennix 2006; Scott et al. 2013); which is highly desirable in the case of stakeholders working on problems of contested land use. Participants gain greater awareness of their own assumptions when asked to make them explicit and can change their minds as a result of new insights which can lead to consensus (Rouwette and Vennix 2006; Biggs et al. 2011): but alignment of mental models does not mean that the resultant conceptual model is correct (Mathieu et al. 2000; Scott et al. 2013). It is tempting therefore to think about eliciting mental models for qualitative purposes only (such as conceptual models), but Lane (2008) asserts that insights into the behaviour of systems are not gained without simulation: Rouwette and Vennix

(2006) also found some evidence that the inclusion of quantitative models are more likely to lead to the above benefits. However, this does not rule out the use of process–based simulation models to test the causal inferences of conceptual models.

Finally, in case of contested land use, there could be a significant obstacle in developing shared conceptual models that is perhaps not found within organisations or groups that have similar goals and objectives. Participants may not agree on how some components of the system interact; for example, if some participants think that *A* does not cause *D*, to use the earlier example in Figure 2.8a, and cannot agree then the participatory process could falter (e.g. Dougill et al. 2006) (Section 2.5.4). Therefore the process used to elicit mental models when working with groups that hold conflicting opinions about land use needs careful consideration.

2.5.4 Mental models and blanket peatlands

There have been a number of projects designed to investigate the future of blanket peatland use that involve stakeholders (e.g. Reed et al. 2013b). A significant amount of scientific work has been carried out on blanket peatlands in the UK and, although tension between stakeholders with different land–use objectives still largely remains (e.g. Elston et al. 2014; *Peatland restoration - what's in it for me?* 2015), much progress has been made. For example, the Sustainable Uplands project (http://sustainableuplands.org) funded by the Rural Economy and Land Use programme (RELU) ran from 2005–2013, and worked with stakeholders in a number of locations including the South Pennine Moors Special Area of Conservation. The project sought to combine both natural and social sciences, using interdisciplinary approaches, to explore alternative futures for the uplands based on the trade–offs and complementarities between agricultural intensification and policies designed to improve carbon storage and water quality (e.g. Dougill et al. 2006; Prell et al. 2007; Reed et al. 2009a, 2013b; Bonn et al. 2014). This work has provided valuable lessons for the implementation of participatory approaches in situations of land–use conflict (see below), and has also resulted in suggestions and proposals for the implementation of ecosystem services approaches and the use of agri–environment scheme funding for peatland restoration (e.g. Raymond et al. 2010; Bonn et al. 2014; Reed et al. 2014a).

In particular, within the Sustainable Uplands project, one study engaged stakeholders in conceptual and biophysical modelling (Reed et al. 2013b). Beginning with a one–year scoping study in the Peak District National Park (part of the South Pennine Moors Special Area of Conservation), researchers worked with five key stakeholder groups including game–keeping and farming communities, along with conservationists, and representatives of water companies and the recreation sector, who were identified using social network analysis (Dougill et al. 2006; Prell et al. 2007). A conceptual model was built by researchers from a series of semi–structured stakeholder interviews

(Carley and Palmquist 1992), and from published literature (Prell et al. 2007). The model was later refined into sub–models which were used for discussion with stakeholders. Scenarios, constructed initially by researchers from the sub–models, were to be refined at stakeholder workshops but this was not, at the time, possible because of disagreement between participants. Later in the project, scenarios were refined by participants and two scenarios (intensification and extensification) were investigated using process–based simulation and GIS (geographic information systems) models (e.g. Chapman et al. 2009b). Finally, the results were coupled with workshop discussions and adaptation strategies developed (Reed et al. 2013b).

The difficulties identified by Dougill et al. (2006) have provided important insights for participatory modelling where land use is contested. Dougill et al. (2006) proposed that stakeholders could initially be divided into groups with similar objectives, using the outputs of their social network analysis, to refine the scenarios developed by researchers from the conceptual model. However this suggestion could also be applied to the development of a conceptual model. Individuals or small groups of stakeholders could create their own cognitive models, to be aggregated later as overlapping knowledge networks (*sensu* Kosko 1988), and avoid early conflict between groups with different objectives. Furthermore, a network of social–ecological interactions built from stakeholders' mental models could be used to evaluate the impact of land–use objectives on blanket peatland carbon storage and identify differences in the way stakeholders characterise causal links. Discussion of these differences, and the interactions of the aggregate network, may help to foster collaboration and trust that could ultimately lead to ways to address conflict (Biggs et al. 2011; Young et al. 2016a), or to collaborative approaches to land–use change.

2.6 The impact of land use and restoration on blanket peatland carbon stores

2.6.1 Background

Blanket peatlands have been degraded by the direct and indirect impacts of humans that have sometimes combined with natural processes, such as erosion, to increase degradation (Parry et al. 2014). The focus of this section is blanket peatlands in the UK, however many of the effects of land use (such as drainage, afforestation, and extraction) apply to peatlands around the world. The impact of land use on biodiversity, carbon storage, and water quality links directly back to the contestation in peatlands discussed in Section 2.3. I summarise these impacts in Figure 2.10.

Peatlands are maintained by hydrological regimes that restrict the decomposition of plant litter

(Section 2.2.3), but the accumulation and decomposition of plant litter is mediated by a complex set of interactions that relate to biological, chemical, ecological and hydrological processes (Holden 2005b). The main impacts of land uses has been to change hydrological regimes and vegetation communities, either directly or indirectly, which in turn affects peat structure, the carbon balance of the peatland, and the feedback of carbon dioxide and methane to the climate system (Figure 2.10). It can be difficult to disentangle the effects of land use and restoration because the future development of a peatland emerges from low–level interactions over the long–term: however, some impacts (such as pollution, gully erosion, and wildfire) can be catastrophic to peat building vegetation and carbon stores (e.g. Ferguson et al. 1978; Tallis 1987; Lamers et al. 2000; Albertson et al. 2009). However, the effect of some land uses on peatland carbon storage is contested.

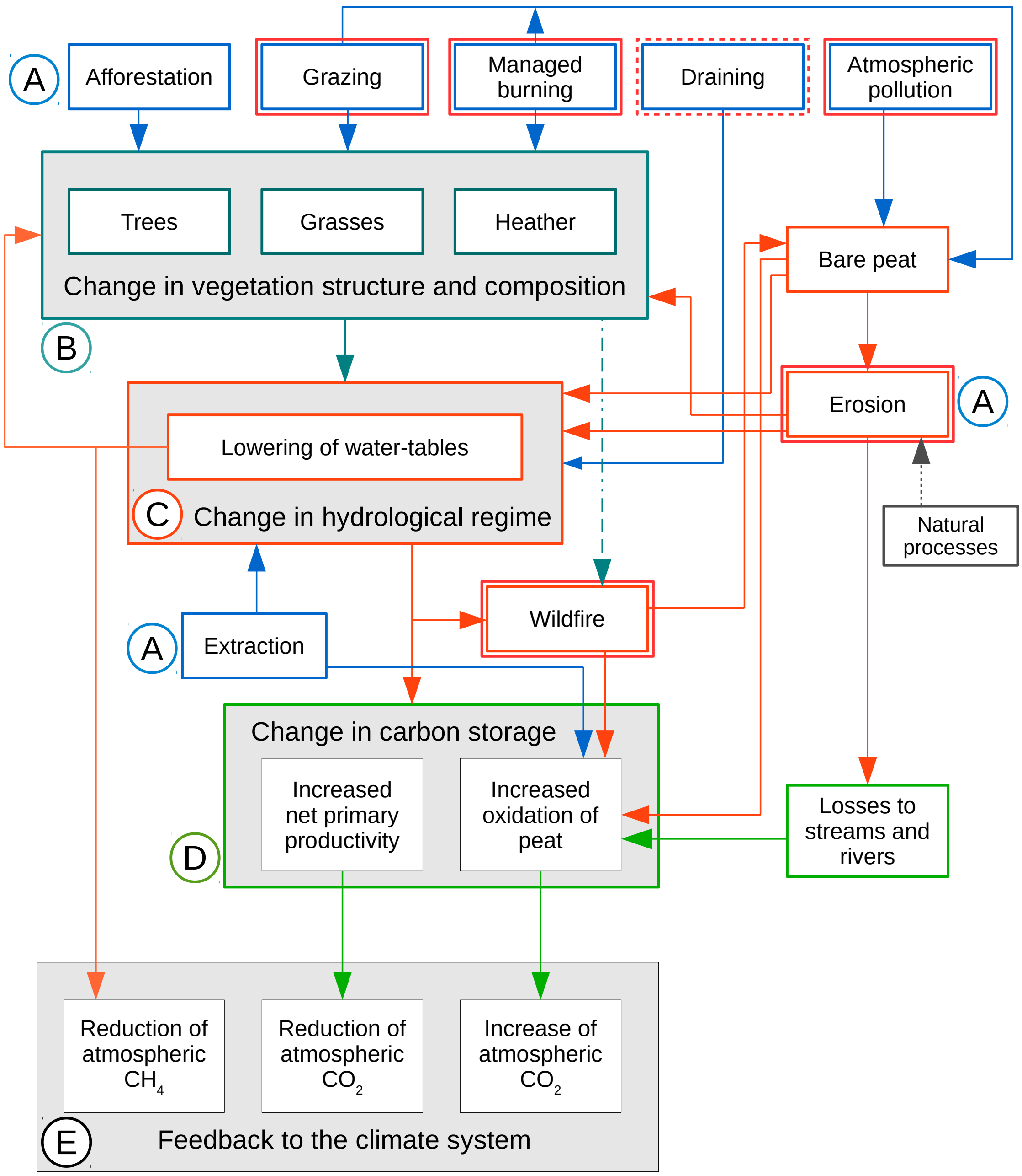


Figure 2.10. Simplified conceptual model of anthropogenic impacts on carbon accumulation in blanket peatlands. (A) Primary impact drivers. Atmospheric pollution (indirect), afforestation and draining are past drivers with impacts that persist today, whilst extraction continues in some locations. Erosion is included as a primary and secondary impact driver because its origin has sometimes occurred prior to land–use impacts, but is often exacerbated by them. **(B)** Vegetation changes driven my land use in relation to forestry, agriculture and driven grouse shooting. **(C)** Impacts of vegetation change and primary drivers (A). **(D)** Changes to the rate of peat accumulation and/or losses and potential feedback of greenhouse gasses to the climate system (**E**). Double boxes are the direct and indirect impact drivers that primarily affect blanket peatlands in the South Pennine Moors Special Area of Conservation. The dashed line around drainage indicate that it is not widespread in some South Pennine peatlands. CO_2 = carbon dioxide and CH_4 = methane. *Note:* Wildfire causes the loss of peat through combustion but is shown here as oxidation for simplification purposes. The dashed line to wildfire signifies a reduction in woody biomass through either burning or mowing.

2.6.2 Summary of the indirect and impact of humans on blanket peatlands

Indirect impacts: pollution, gully erosion, and wildfire

The effect of atmospheric pollution is to make peatlands more vulnerable to land use and erosion (e.g. Evans et al. 2005). The loss of *Sphagnum* spp. and other peat–forming vegetation from the peatland surface due to sulphur pollutants (SO_2) has produced areas of highly acidic peat without vegetation. Although sulphur pollution in the UK has fallen significantly (RoTAP 2012), areas of bare peat remain acidic and are highly susceptible to continued erosion unless vegetation is restored (Parry et al. 2014). Deposition of airborne nitrogen, which has not reduced significantly (RoTAP 2012), can also negatively impact *Sphagnum* spp. and favour vascular plants (Lamers et al. 2000).

Many blanket peatlands in the UK are subject gully erosion due to the combined impact of natural processes and land use (Tallis 1987). Gullying and subsequent erosion as a result of the mechanical failure of the peat margin (slides or bursts) pre-dates the impact of present management treatments (Tallis 1985), but fires and pollution in the past 200–300 are thought to have caused a renewed periods of erosion, which has been exacerbated by overgrazing by sheep (Tallis 1987). Gully erosion causes peat to be lost directly to rivers and streams, and changes the hydrology of gully–side and interfluve peat (the area between gullies) which increases decomposition, and changes vegetation composition (Allott et al. 2009; Evans and Lindsay 2010). Water tables in gullied peatlands have been reported to be > 50 cm deep compared to < 15 cm for intact sites (Pilkington et al. 2015).

Fires have affected peatlands for centuries either through natural sources of ignition or induced, accidentally or deliberately, by humans (Albertson et al. 2009). Whilst key bog-forming species such as *Sphagnum* spp. are, in time, able to recover from fire (Blundell and Holden 2015), uncontrolled events can lead to significant losses of above–ground vegetation and below–ground peat from combustion and subsequent erosion (Tallis 1987; Dixon et al. 2015). Frequent managed burning is often cited as necessary to reduce the risk of such events by reducing woody biomass especially on peatlands managed to encourage *C. vulgaris* (e.g. Dixon et al. 2015). However, this seems to be a vicious circle: managed burning increases the amount of woody biomass and grasses as a result of deeper water tables which in turn increases the vulnerability of peatlands to wildfire (Turetsky et al. 2011). Holden et al. (2012) argued that shallow water tables, along with the vegetation communities that tend to occur in these conditions, would reduce the risk of fire: a suggestion supported by (Turetsky et al. 2015) who demonstrated that shallow water tables increase the resistance of peatlands to fire.

Direct impacts: grazing, draining, burning, afforestation, and extraction

Grazing pressure has two main impacts on blanket bog peatlands; vegetation change and erosion as scars and paths. Low rates of sheep grazing ($\approx$0.5 ewe ha^{-1}) can eliminate dwarf shrubs leading to vegetation dominated by hare's tail cotton grass *E. vaginatum* or wet heath. Although heather is not preferred by sheep, grazing has significantly reduced areas of heather moorland in the UK (Anderson and Yalden 1981; Thompson et al. 1995). Burning can be carried out to improve grazing conditions: Clay et al. (2009) found that plots that had been grazed for 20 years or were burnt on a 20 year rotation had shallower water-tables than those burnt every 10 years. And in a study of the effects of gazing and burning, Garnett et al. (2000) found that burning had reduced carbon accumulation rates but that low–intensity grazing had no significant effect: although burnt plots had higher rates of grazing than unburnt plots. However, Ward et al. (2007) found that both grazing and burning reduced carbon storage, and strongly affected vegetation composition in favour of grasses.

Drainage of peatlands has been widespread throughout the world. Although many drains still exist in the UK, new drains are rare except where construction takes place (e.g. access roads for windfarms). Draining was initiated in order to improve sheep farming productivity but provided no obvious benefit, (Bellamy et al. 2012). Draining deepens water-tables which in turn alters peat hydrological structure, and promotes the growth of grasses and woody plants such as heather. Hydrological conditions and vegetation composition can be negatively affected at some distance from the drain (15 m in a study by Wilson et al. (2011a)). Holden (2005c) also found that increases of heather resulted in a greater density of pipes within the peatland which provide preferential flow paths for water to streams. Shrinkage and compression of peat as a result of drainage leads to subsidence and to the development of surface cracking; again providing preferential flow paths for water as well as enhanced oxidation and erosion. Studies in tropical peatlands have shown that draining can result in the decomposition of 'old' peat accumulated many years ago (e.g. Moore et al. 2013), because deeper water tables exposed previously submerged peat to oxic decay in a process known as secondary decomposition (Tipping 1995).

The repeated burning of blanket peatlands to improve red grouse habitat alters peat hydrology, accumulation rates, propensity for erosion, vegetation diversity, and vegetation composition (Ramchunder et al. 2013; Clay et al. 2015; Holden et al. 2015). Managed fires are carried out under controlled conditions that aim to minimise the impact on the peat surface. Figure 2.11 (from Holden et al. 2015) identifies the hydrological response of blanket peatlands to burning from 120 plots over five burnt and five unburnt river catchments (2, 4, 7 and >10 years after burning). Recently burnt plots were found to have deeper water tables and lower saturated hydraulic conductivity than plots burnt in previous rotations. Clay et al. (2009) also found deeper water tables in more recently burnt

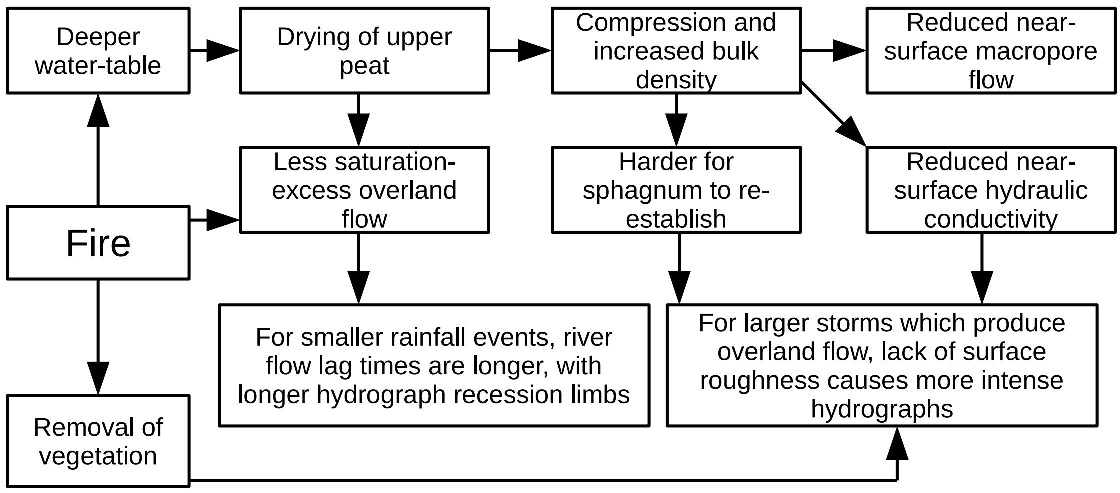


Figure 2.11. Impact of managed burning on peatland hyrdological processes. Redrawn from Holden et al. (2015).

plots (comparing 10 and 20 year burns) but higher saturated hydraulic conductivity, which they attributed to increased macropores due to the decomposition of older heather.

Some boreal and tropical peatlands are naturally forested, and can be subject to deforestation, but blanket peatlands in the UK developed without substantial tree cover (Moore 1975). Commercial afforestation of peatlands was carried out mainly in Scotland sponsored by the UK Government; but future plantation on blanket bog is unlikely to be granted the necessary permissions (Patterson and Anderson 2000). Afforestation (which includes the digging of narrow drains) increases oxidation through increased transpiration and drainage (and hence decomposition), intercepts light and rainfall, and changes the understory vegetation to the composition of a forest floor (Holden et al. 2007). Some studies have suggested that afforestation would reduce carbon dixoide and methane emissions and benefit carbon accumulation (Worrall et al. 2010; Yamulki et al. 2013). But this proposal was challenged by (Artz et al. 2013) who noted that peatlands can continue to accumulate carbon over millennia whereas forest productivity peaks after an initial period of growth and declines with stand age.

Peat extraction is carried out on industrial (for fuel and horticulture) and household (for fuel) scales (Chapman et al. 2003; Beyer and Höper 2015). Industrial harvesting of peat removes growing vegetation and the top layer of peat after drainage. Loss of accumulated carbon occurs because it is physically removed, whilst oxidation is enhanced in the peat that remains (cutover or cutaway peatlands). Extraction at this scale causes landscape–wide damage to blanket peatland ecological and hydrological functioning (Farrell and Doyle 2003; Price et al. 2003). Small scale block cut extraction has occurred in peatlands for centuries (Price et al. 2003). Trench cutting can occur within the peatland, but peat is often removed at the margin creating a cutting face; again altering the hydrological regime of the peatland through enhanced drainage.

2.6.3 The aims of peatland restoration

Many degraded peatlands around the globe have been the subject of restoration activities (e.g. González et al. 2013; Schimelpfenig et al. 2013; Parry et al. 2014). The main aim of restoration is to reestablish the hydrological conditions needed to reverse losses of gaseous and fluvial sources of carbon and provide the ecological conditions for peat accumulation by rewetting (Beetz et al. 2013). There is evidence that effects of prolonged degradation on peat structure may not be able to be reversed even after many years (Schimelpfenig et al. 2013); and in some geographic locations, there are concerns that a warming climate will add to the effects of degradation, and reduce the viable areas for peatland development, and increase carbon losses from degraded peatlands (Holden 2005b; Gallego-Sala and Prentice 2012; Li et al. 2015). Evidence of the impact of restoration is needed to inform policy and land–use decision–making (Maltby 2010). However sometimes this evidence has been equivocal or counterintuitive. For example, Bellamy et al. (2012) found some evidence that drain blocking increases typical peatland vegetation, but this effect was only apparent at one site where drains had been blocked longest. In an investigation into the effect of managed burning and carbon stocks, Clay et al. (2015) found that unburnt plots and plots that had not been burnt for several years, were greater sources of carbon than recently burnt plots. The authors attribute this to the rapid growth of vegetation immediately following burning: however, they note that carbon accumulation would only take place if peat forming vegetation was restored to the site.

The timescales and adaptive response of peatlands make it challenging to understand the effects of peatland restoration and land use on carbon storage. Revegetation of bare peat and blocking ditches or gullies are perhaps the two most common methods that underpin many restoration projects, but an understanding of their effect over centennial to millennial timescales has not yet been established. Blocking of drains and erosional gullies takes place in peatlands throughout the world (e.g. Wilson et al. 2011b; Schimelpfenig et al. 2013; Dixon et al. 2014), with the aim of reducing peat decomposition and erosion caused by deeper water tables. Although gullies and drains have a negative effect on peatland carbon storage, biodiversity, and hydrology (Wallage et al. 2006; Evans and Lindsay 2010; Carroll et al. 2011), most studies take field measurements of water tables to establish the impact of drain and gully blocking. Studies have reported water tables in restored sites as intermediate to those found in drained or gullied sites, and those found in intact peatlands (e.g. Holden et al. 2011; Pilkington et al. 2015). However, most research has taken place over a relatively short period of time (< 15 years): for example Dixon et al. (2014) and Pilkington et al. (2015) collected data over five years. As a result, a number of authors have suggested that monitoring over longer periods is needed to address this knowledge gap (e.g. Holden et al. 2011). In addition, comparisons of water tables between intact sites and restored sites, to assess the success of restoration, may be confounded by differences in peat properties (such as hydraulic conductivity) which are rarely taken into account (Baird 2014).

To add to this picture, in Section 2.4 I discussed how autogenic behaviour can, in some cases, effectively disconnect peatlands from external forcing and make it difficult to understand how peatlands respond to climate change: it is likely that these mechanisms will also affect responses to restoration or land uses. Because peatlands develop and adapt over centuries and even millennia, models that incorporate changes in peat properties could help to address questions about how restoration or land uses affect carbon storage, and of the spatial and temporal variation in water tables, in blanket peatlands over long timescales.

2.6.4 Future challenges for blanket peatland restoration in the UK

Scientific understanding of peatland feedback mechanisms is continually developing and can challenge past thinking about restoration. Management decision–making therefore needs to adapt to new research findings but there are significant gaps in the dissemination of research to the practitioners who make decisions about restoration activities (Anderson 2014). Palaeoecology can provide a useful role in helping to determine restoration objectives by informing conservationists and land–owners/–users of historical blanket peatland accumulation, and disturbance regimes (both natural and anthropogenic) (Blundell and Holden 2015). But the technique should not be used to set targets inconsistent with likely future peatland development trajectories that could exacerbate the problems of today (Jackson and Hobbs 2009).

Perhaps one of the most challenging aspects of restoring blanket peatlands, will be how to combine future land uses that provide multiple benefits and avoid win–lose or lose–lose outcomes (Reed et al. 2013b; Redpath et al. 2013). Some restoration activities already bring 'win–win' outcomes: large areas of bare peat are unlikely to recover autogenically, but restoration of vegetation cover should improve all ecosystem services. Ideally, all relevant stakeholder groups should work together to determine trade–offs or complementary objectives (e.g. Reed et al. 2013b, Appendix 1), an approach that may help to, at least, reduce some of the suspicion and tension that still exists between conservationists and land users (*Peatland restoration - what's in it for me?* 2015). However, the nature of these deliberations are likely to favour peatland restoration and the reduction of damaging land use practices: the UK Committee on Climate Change has recommended that peatland restoration is prioritised, and that there is an assessment of the degree to which land uses that degrade peatlands are funded by agri–environment schemes (Committee on Climate Change 2015, p. 16). However, conflict could be made worse if those that favour peatland restoration are seen to exploit this position (Redpath et al. 2013).

2.7 Research Gaps

This chapter has identified a number of research gaps that link together contested land use, complex systems, and stakeholder participation. These gaps are summarised below and were used to define the research aim and questions for this thesis that were presented in Chapter 1.

1. Peatland stakeholders are being challenged by the drive to restore function to degraded ecosystems, and conflict can develop about restoration and future land uses if stakeholders cannot contribute fairly to the decision–making process, or become disadvantaged because of land–use decisions. Recent studies that engage blanket peatland stakeholders have tended to focus on the provision of ecosystem services, but there is a need to incorporate a wide range of stakeholder knowledge about social and ecological interactions into peatland models of complex systems to co–develop an understanding of their impact on carbon storage.

2. Network dynamics and structure have been used in many biological, ecological, and man–made complex systems to provide insights into, for example, causal interactions, robustness, and control. But their use in peatlands has been limited to conceptual models and social network analysis to understand stakeholder relationships. Several approaches to modelling complex systems incorporate networks that can be used in participatory processes. Although these approaches have not been used in blanket peatland participatory processes, networks represent a transparent, accessible, and intuitive approach for a wide range of stakeholders to share knowledge. The quantitative and qualitative outputs of models which are based on collaborative working may lead to insights into how stakeholders perceive that land use impacts carbon storage, and into the assumptions that underpin conflict. In addition, the results of models that are co–developed may be thought of as more credible, when being used to address questions of contested land use, if stakeholder knowledge is integrated into decision–making processes.

3. There are a number of process–based models that simulate peatland processes but few simulate blanket peatland development, and none of these are based on a series of hydrologically connected peat columns that include a feedback between decomposition, hydraulic conductivity, and water–table position. Such a model would allow a spatially heterogeneous peat structure to develop, and enable the effect of both land use, and climate change to be simulated.

4. The long–term (i.e. centennial to millennial) impact of land use or restoration on carbon storage has not yet been explored using models that simulate blanket peatland development.

Field–based studies that investigate the impact of land use or restoration are of relatively short duration, especially when compared to the timescales involved in peatland development, and the long–term effects are poorly understood. Therefore, an understanding of how land use or restoration affects blanket peat carbon storage in the long term could be used to inform management decisions. The inclusion of peat properties such as hydraulic conductivity, may help to provide insights into the spatial and temporal differences between degraded, restored, and intact sites.

2.8 Study region and methods summary

This section describes the study region, the selection of two models, and the participatory fieldwork undertaken between June 2013 and February 2015. Section 2.8.1 describes the study region. Section 2.8.2 outlines the series of participatory workshops used to co–develop a network model of blanket peatland interactions, and desribes the selection of a suitable participatory modelling approach. Finally, Section 2.8.3 describes the selection of a current peatland development model than can be further developed to represent blanket peatlands. The processes used to develop models, the model equations and algorithms, and the analysis of model and workshop outputs are reported within subsequent chapters.

2.8.1 Study region

The South Pennine Moors are designated as a Special Area of Conservation (SAC) and Special Protected Area (SPA) and as such are protected according to the Habitats Directive and Natura 2000 (Natural England 2014). The South Pennines SAC covers an area of $\approx$650 km^2 including 274.4 km^2 of blanket peatlands which, when supporting peat forming vegetation, are classed as a priority feature (JNCC 2015). However, many blanket peatlands in the South Pennines are in a highly degraded state as a result of the combined effects, over many years, of; erosion, atmospheric pollution, overgrazing, wildfire, and managed burning (Tallis 1987; Anderson and Radford 1993; Bragg and Tallis 2001; Ramchunder et al. 2013; Pilkington 2015). The South Pennine SAC incorporates the Peak District National Park (Figure 2.12), of which the northern extent (known as the Dark Peak because of the underlying gritstone) was designated an environmentally sensitive area in 1988 because of degradation of heather moorland as a result of overgrazing (Condliffe 2009). There are a number of areas of blanket peat in the Dark Peak (Figure 2.12b), which although likely to be some of the most degraded anywhere in the world (Pilkington 2015), have been undergoing restoration treatments for a number of years. Restoration of these peatlands includes revegetation of bare peat, stock exclusion, and blocking of erosional gullies (Pilkington 2015).

Restoration in the South Pennines and Dark Peak has included partnership organisations such as Moors for the Future (www.moorsforthefuture.org.uk) and Pennine Prospects (www.pennine-prospects.co.uk) who work with landowning and non-land owning individuals and organisations. Funding for restoration comes from sources such as the EU LIFE programme (Parry et al. 2014), agri–environment schemes, and private funding from water companies (Severn Trent, United Utilities, and Yorkshire Water). Stakeholders who use blanket peatlands in the Dark Peak and wider South Pennines SAC are highly heterogeneous, they often have conflicting land–use objectives and include those with interests in; farming, field sports, rambling, tourism, peatland science, forestry, water quality and supply, and conservation. The recruitment of stakeholders as participants in my research is reported in Chapter 3. Stakeholder workshops, reported in Chapters 3 and 4, took place in several locations in the Dark Peak, with some participants travelling from the wider South Pennines. The research in this thesis was supported by the Moors for the Future Partnership, who are based in Edale, Derbyshire which is located in the Dark Peak.

The South Pennines can therefore be characterised according to the three themes outlined in Chapter 1: (1) the South Pennine blanket peatlands are degraded and land use is often contested; (2) there is a complex set of social and ecological interactions that make it difficult to understand and predict the impact of land use or climate change, and challenging to address issues of decline and degradation; and (3) future land–use decisions should incorporate knowledge from across the stakeholder community in participatory processes. These characteristics form a model case study region from which to address the research aim of this thesis.

2.8.2 Methods: participatory processes

Participatory modelling was used to enable stakeholders to contribute a wide range of knowledge to a co–developed model of blanket peatland social and ecological interactions. The overarching objective for this process was to chose a simple, intuitive approach that was accessible to all stakeholders regardless of background or experience. Next, the approach needed to be flexible so that individual, or groups of, stakeholders could build network diagrams which could be aggregated, and the connections between nodes validated at a later date. As a result I chose to use a mental modelling approach (Section 2.5.3) because stakeholders could be asked to describe, using natural language, the components of the network and both the causal direction and impact of interactions between components (Jetter and Schweinfort 2011; Elsawah et al. 2015): the participatory processes used are described in detail in Chapters 3 and 4.

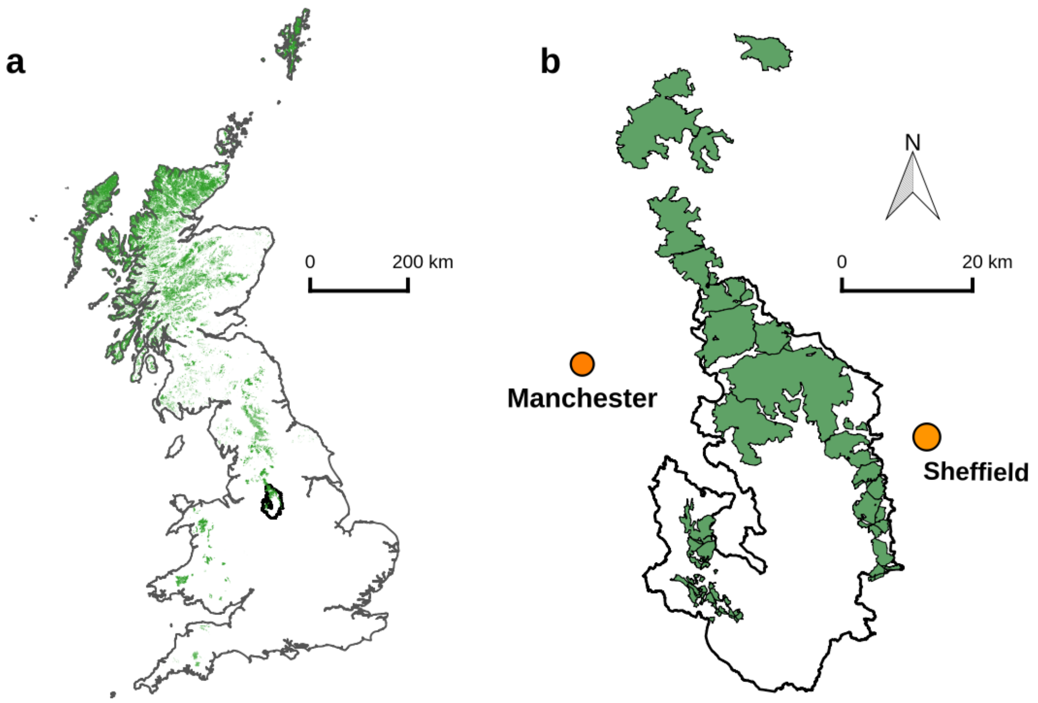


Figure 2.12. Blanket peat distribution. **a** Great Britain with location of Peak District National Park. Shaded areas are blanket peatlands. **b** South Pennine Moors Special Area of Conservation (SAC) and Peak District National Park Boundary. Shaded areas are mainly blanket peatlands. Peak District National Park outline, blanket peatland extent, and South Pennine SAC; Joint Nature Conservation Committee Support Co. ©Crown Copyright. All rights reserved 2016. Outline of UK mainland; ©Crown Copyright/database right 2016. An Ordnance Survey/EDINA supplied service.

Selection of participatory modelling approach

One of the main benefits of working with networks is that it is intuitive to think about components of a system being connected together by causal links (Gershenson and Niazi 2013). Network modelling seems to be an approach well suited to participatory modelling of complex systems where elicitation can involve building or drawing the interactions between the various elements of a system. Drawings or maps can be used as conceptual models to focus stakeholder discussions, to identify priorities for further research using simulation models, or for scenario development (e.g. Bestelmeyer et al. 2004; Etienne et al. 2011; Reed et al. 2013b).

There are a number of different approaches that enable stakeholders to directly contribute (i.e. not from interviews or documentary evidence) mental models during a participatory processes (Section 2.5.3). Several of these methods involve the development of a conceptual model: Actors, Resources, Dynamics, and Interactions (ARDI) (Etienne et al. 2011), is one such approach where the conceptual model can used to develop simulation models – a similar approach to that used by (Reed et al. 2013b). Direct approaches such as ARDI, Hexagons (Hodgson 1992), systems dynamics (causal loop, and stock/flow diagrams) (e.g. Lane 2008; Corral-Quintana et al. 2016),

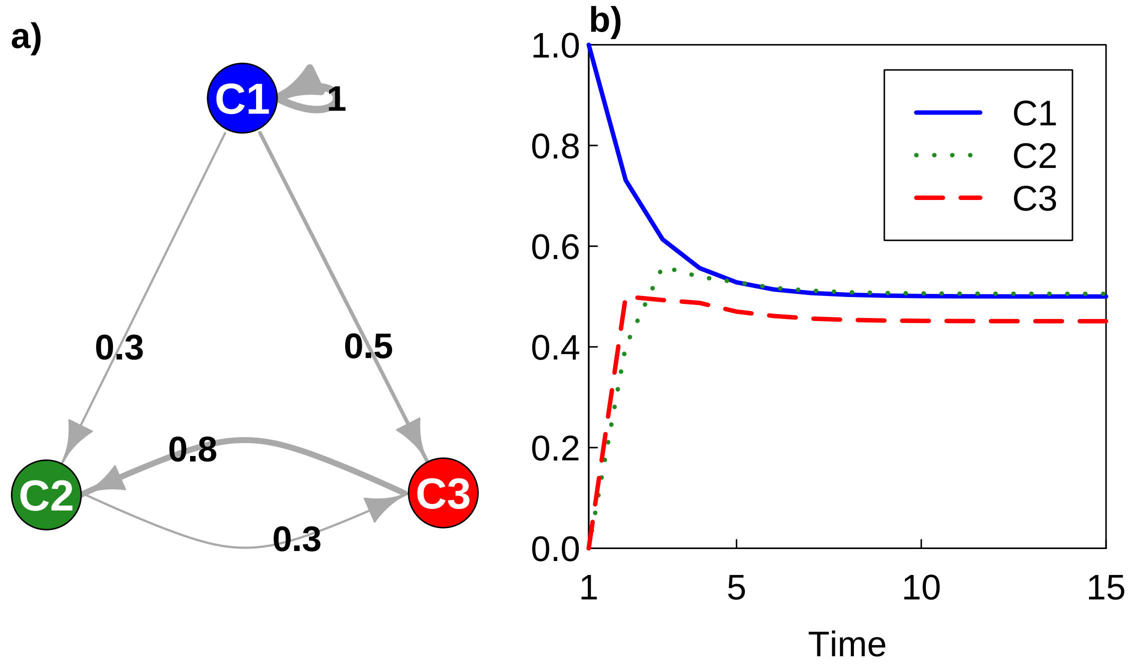


Figure 2.13. Example of fuzzy cognitive map (FCM). **a** Network of three concepts or nodes indicating the direction and strength of each connection (also known as a directed weighted graph). The thickness of each connection (known as an edge) is proportional to strength (the weight) of interaction. The loop from C1 $\Rightarrow$ C1 is known as a self-loop. The concepts can refer to social or ecological components in a coupled system. **b** FCM output over fifteen model iterations. The stable values of each concept at iteration 15 can only be interpreted as a relative ranking. Replicated from an example provided in Knight et al. (2014, p.194 and Fig.2d).

elicit mental models by using diagrams. And mental models have also been used during the collection of data to build more complex agent–based models (e.g. Elsawah et al. 2015, although the diagram was built from stakeholder interviews). Fuzzy cognitive maps (FCMs) (Kosko 1986) are causal diagrams that can be directly developed and parameterised by stakeholders using numeric or linguistic link weights to create a directed weighted network of interactions: importantly FCMs can include feedback loops. A simple mathematical model can then be used to develop an output of system dynamics that is semi-quantitative in nature (i.e. can only be interpreted in relative terms) (Penn et al. 2013) (Figure 2.13). Christen et al. (2015) used a FCM to identify the reasons that Scottish farmers did not comply with agricultural environmental legislation, which they classify as a wicked problem. By comparing networks of farmers and non–farmers (who designed and communicated of regulation), the FCM was used to highlight areas of conflict and common ground between stakeholders, which could be used in future policy development.

All of the approaches discussed here would be suitable to incorporate knowledge from different blanket peatland stakeholders. I chose to co–develop a blanket peatland fuzzy cognitive map because; (1) the process is transparent: creating a network by diagramming is intuitive, and the concept of the simple mathematical model can be explained relatively easily; (2) differences in how stakeholders perceive interactions can be used to build agreement, or identify areas of strong disagreement that need to be addressed; and (3) network structure (including driver nodes) can be used as a framework by stakeholders to discuss how to achieve land–use objectives, and the outputs can be used to model the impact on carbon storage. To the best of my knowledge, an FCM has not been used in a participatory process designed to investigate carbon storage in blanket peatlands . However, because of the semi–quantitative nature of fuzzy cognitive maps, a peatland development model was needed to explore the long–term quantitative effect of land use on carbon storage.

Participatory modelling, and land–use objectives workshops

Figure 2.14 illustrates the research process used during this thesis. The approach combines existing methods and analyses for participatory modelling, and is similar to other frameworks and processes used in previous participatory studies that incorporate models of complex systems. For example, Reynolds et al. (2009) identified five principles of the Drylands Development Paradigm (human–environment systems are coupled; slow variables determine system dynamics; there are alternative states separated by thresholds; coupled systems are networked; local management and policy knowledge should be integrated with scientific knowledge). Whitfield and Reed (2012) proposed an assessment approach for drylands that was based around three concepts and related participatory methodologies (evaluation of ecosystem services; modelling of system complexity; assessments of history and future scenarios), and suggested integrating conceptual understandings of the system and process–based models.

The study of Reed et al. (2013b), discussed in Section 2.5.4, linked a cognitive model partly based on stakeholder knowledge and process–based models in the same region as this study. However, the process reported here differs as follows. (1) a series of participatory workshops will be used to elicit individual and group cognitive models (weighted directed networks) which will be combined to create a network of blanket peatland interactions. (2) During a separate workshop, any differences in the direction of individual interactions will be identified, reviewed, and agreed by participants. (3) The structure of the network will be analysed to identify the driver nodes, and the outputs used by stakeholders to identify how three land–use objectives could be achieved; the objectives will be related to livelihoods, water quality and carbon storage. (4) The results of the workshop will be used to determine the relative impact of each objective on carbon storage. (5) Finally, these outputs will be linked to a peatland development model to explore the centennial to millennial impact of two driver nodes on carbon accumulation in blanket peatlands.

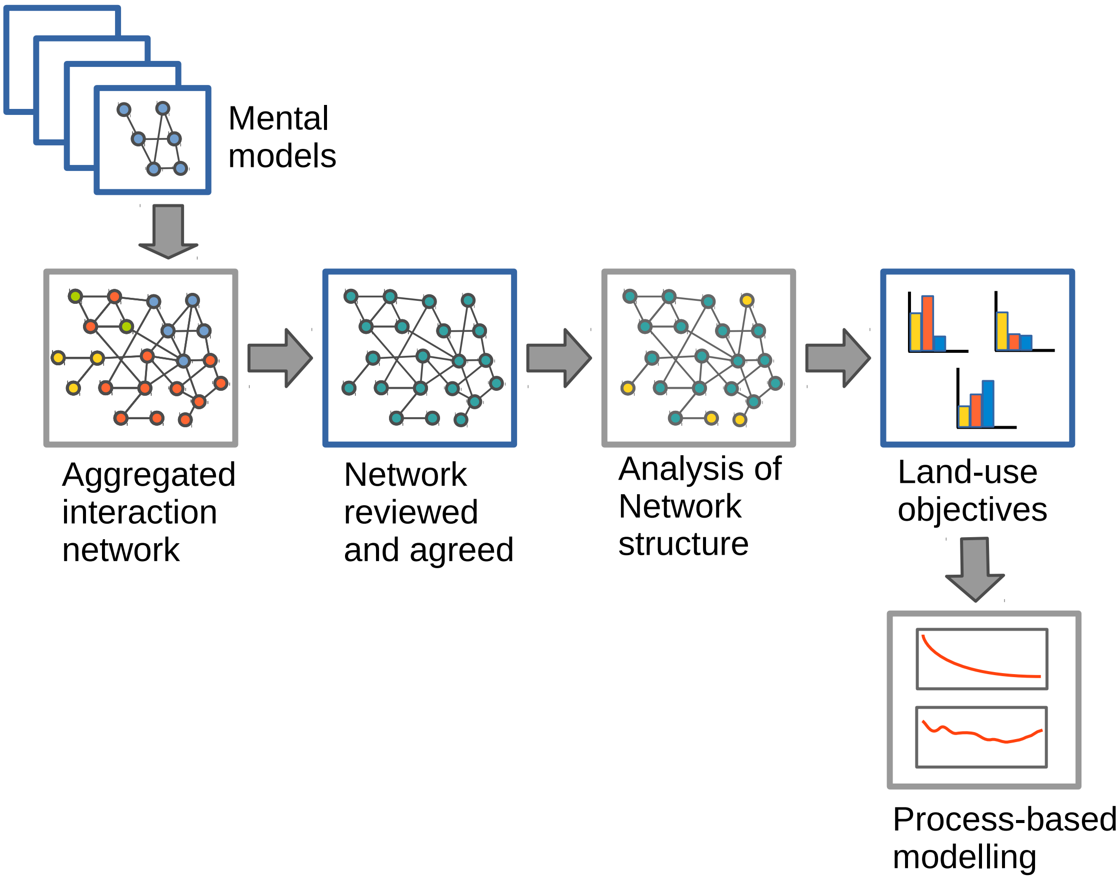


Figure 2.14. Outline of research process used in this thesis The process was designed to connect different sources of stakeholder knowledge and determine the impact on carbon storage using network and process–based models of blanket peatlands when conceptualised as a complex system . Process steps in blue boxes involved stakeholder participation.

2.8.3 Methods: selection of peatland development model

The aim of this section is to briefly review the suitability of three current peatland development models, that simulate the accumulation of peat as a result of bottom–up interactions between peat layers, and select one for use or adaptation in order to model blanket peatland development. The purpose of incorporating a process–based simulation model within this research is to understand the long–term effects on blanket peatland development of selected outputs from participatory modelling workshops. The models reviewed are Millennia, the Holocene Peat Model and **`DigiBog`**. These models build up columns of peat from individual layers (cohorts) starting with a single layer: the models can comprise a single column (one dimension, 1D) or be made up of several columns arranged to form a whole or a section of a peatland (2D / 3D). The main processes that I consider here are, external forcing by climate variables (in this case rainfall and temperature), vegetation dynamics, decomposition, and hydrology.

Millennia

Millennia (Heinemeyer et al. 2010) is the only model reviewed here that sets out to represent UK blanket peatland development. Net primary productivity (i.e. the production of plant and root inputs to the model) is designed to represent that of eight plant functional types found on UK blanket peatlands. Plant functional types are determined by water–table position and the resultant litter input (including root litter) is partitioned into soluble, holocellulose and lignin fractions (Heinemeyer et al. 2010). Decomposition takes place according to the water–table position with different rates for oxic and anoxic decay expressed as a ratio that varies with depth up to 10 cm below the water–table. Annual peat accumulation is calculated from litter addition and decomposition losses which determines long term peat development.

Millennia, has three significant drawbacks which preclude its use as a model of a complex system. (1) Horizontal water movement occurs only as runoff which is a function of water–table depth, slope and net rainfall inputs (rainfall less evapotranspiration). It appears that Millennia comprises a set of individual columns that are not hydrologically linked and therefore the model lacks cross-scale feedback related to peatland size and/or shape (Belyea and Baird 2006). (2) Hydraulic conductivity is not calculated: the available pore space is calculated according to depth–based assumptions and so the negative feedback mechanism between water–table depth, saturated hydraulic conductivity and decomposition is absent from the model (Heinemeyer et al. 2010, p. 223, section 4.7.1). Finally, (3) water–table depth is determined by the climate inputs Heinemeyer et al. (2010, p. 212, section 2.3.2) and therefore the autogenic response of the peatland to external forcing is also absent.

Holocene Peat Model (HPM)

HPM (Frolking et al. 2010) models peatland development as a single column (1D). Vegetation and subsequent net primary productivity is based on twelve plant functional types controlled by water–table and peat depth (and hence isolation from mineral inputs). Decomposition takes place according to the Peat Decomposition Model (Frolking et al. 2001) and is used to determine bulk density which in turn increases hydraulic conductivity, affects water–table position and litter production; which are linked in a set of feedbacks. Water input to the peat column is from incoming rainfall and outgoing evapotranspiration (which is depth dependent). Horizontal movement of water occurs depending on water–table depth and relative transmissivity (depth averaged saturated hydraulic conductivity multiplied by peat depth below the water table). The proportion of peat mass remaining after decomposition is used by both HPM and **`DigiBog`** to calculate saturated hydraulic conductivity (K_{sat}, Morris et al. 2015b). Morris et al. (2015b) compared the functional form of the equations used in both models with empirical data for different microform types and found that whilst **DigiBog** overestimates their data, the exponential form used is compatible (and closely represents hummock data). But the form used in HPM is inconsistent with the reviewed data in a way that lessens the negative feedback that dampens water–table movement in fresh peat Morris et al. (2015b). Finally, the parameters of the twelve plant functional types used by HPM to model vegetation dynamics are likely to require a suitability review for compatibility with UK peatlands and appear to be an overcomplication that would be difficult to parameterise.

DigiBog

Belyea and Baird (2006) called for those modelling peatland development to think about peatlands as complex systems (Section 2.4), which resulted in the **DigiBog** peatland development model (Baird et al. 2011; Morris et al. 2011a). **DigiBog** couples both ecological and hydrological processes that interact in the production of plant litter, peat decomposition and hydrology to mediate the addition of peat to grow a virtual peatland (Figure 2.15). Of the models reviewed here **DigiBog** is the only one truly capable of modelling peat over more than one dimension and therefore take account of the spatial heterogeneity that is present in many peatlands. In **DigiBog** a peatland can consist of several columns that are hydrologically linked (Figure 2.16). Within annual peatland growth cycles, peat in each layer is decomposed (losing mass and thickness) depending on its position above or below the water–table. The result is that layers within columns can have differing hydraulic properties to their neighbours, and spatial heterogeneity can develop. Water flow takes place in only two horizontal dimensions, but peat properties that affect flow also vary vertically. As a result, water movement in **DigiBog** is classed as 2.5D (Baird et al. 2011). Cross-scale feedback is provided by the links between plant litter production and decomposition on peatland size and/or shape and the reciprocal link on peatland hydrology (Figure 2.15).

DigiBog models plant litter input differently to Millennia and HPM. Instead of using PFTs to determine litter input, **DigiBog** models litter production based on a plant assemblage in relation to mean annual water–table. The empirical data for the function was originally determined by Belyea and Clymo (2001). **DigiBog** treats plant litter as homogeneous and takes no account of the effect of decomposition on different litter fractions as do Millennia and HPM. Baird et al. (2011) put forward two arguments for the use of this function instead of PFTs. The first is that water–table position does in fact have a significant effect on litter production. Secondly, this simple approach was deemed to be beneficial for analysis of the drivers of peatland development which could be complicated by autogenic behaviours. Although originally developed for raised bogs, there is little limitation its use as a model for blanket peatland development. Modifications are required to the code to combine 1D ecological and 2D hydrological models to simulate peat accumulation in 2.5D on slopes. Because of the potential drawbacks associated with Millennia and HPM, I have selected `DigiBog` for the purposes of process–based modelling of peatland development.

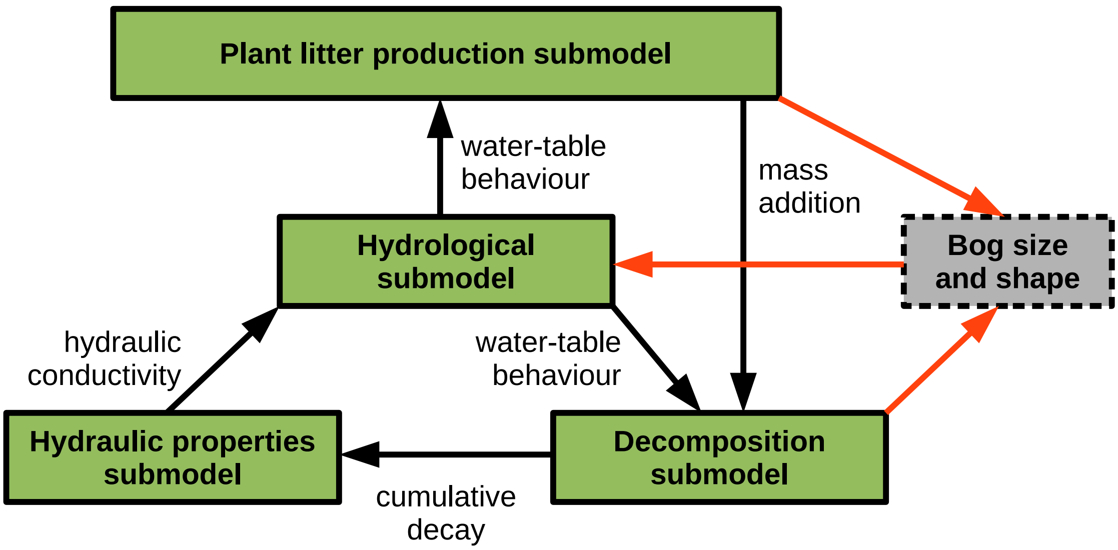


Figure 2.15. `DigiBog` conceptual model. Conceptual model showing links between sub–models (black arrows) and cross–scale feedbacks (orange arrows). Redrawn from Baird et al. (2011).

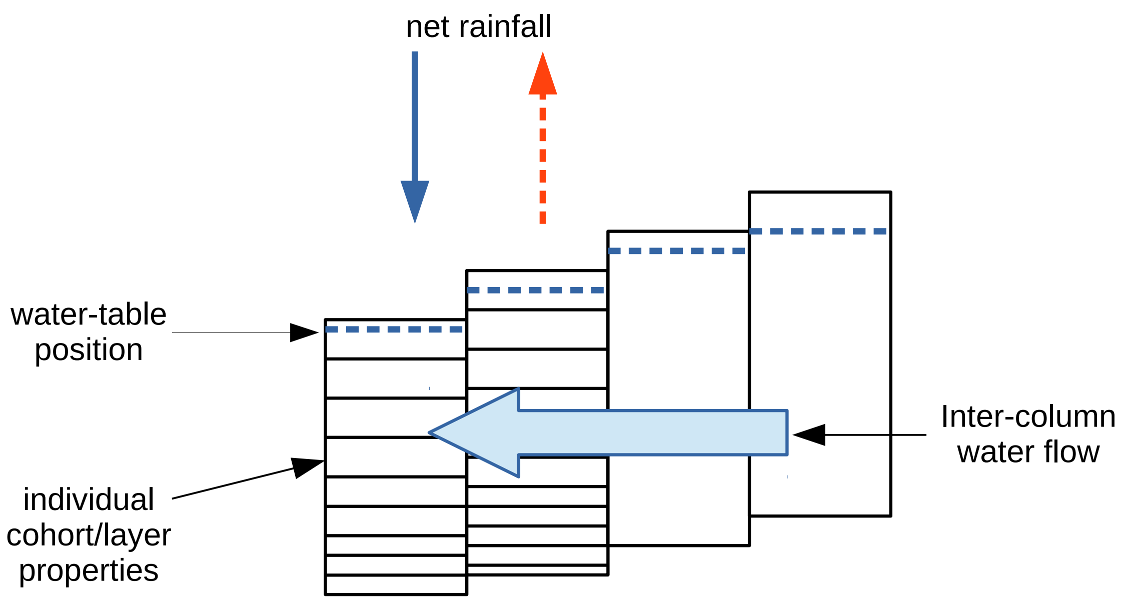


Figure 2.16. `Digbog` column and layer structure. Column and layer structure over a sloping base. Net rainfall comprises rainfall (blue arrow) and evapotranspiration (dashed orange arrow). Redrawn for a sloping base from (Baird et al. 2011).

2.9 Conclusion

This chapter provides a rationale for the studies included in this thesis by linking contested land use, complex systems, and stakeholder knowledge. The aim of this thesis is to develop new knowledge about blanket peatlands as a complex system in order to enhance current understanding of the impact of social and ecological interactions on carbon storage. Because of the contested nature of blanket peatland use I have chosen to establish the impact of social–ecological interactions on carbon stores based on a model co–developed by peatland stakeholders. I propose that stakeholders work together to agree on an aggregate model of the structure of blanket peatland interactions and use this model to determine how land–use objectives could be achieved; which may also provide further insights into the contested nature of land use and restoration (Chapters 3 and 4). To further investigate and quantify the long–term impact of factors from the co–developed model on carbon stores, I will develop a new version of the **`DigiBog`** peatland development model (Chapters 5 and 6) to explore blanket peat accumulation over centennial and millennial timescales.

Chapter 3

Co–development of a model of blanket peatland social and ecological interactions

3.1 Chapter summary

In Chapter 2, I developed a rationale for this research that included the integration of a broad range of stakeholder knowledge to develop a model that could be used in future land use discussions, communications, and decision–making processes. Here I set out the participatory processes used to elicit stakeholder mental models and construct preliminary and validated group cognitive maps (directed weighted networks). The group cognitive map forms the network model of social and ecological interactions used in subsequent chapters. The methods used to elicit stakeholder mental models, create a validated group map, and define the importance of map components are discussed. In this case I use the term validation to relate to the interactions between components which form the structure of the network rather than the validation of model outputs. I also discuss the terminology used in this and subsequent chapters related to cognitive maps, models, networks, and their components.

A series of workshops and one-on-one meetings were held to capture stakeholder mental models, which were aggregated and reviewed prior to a validation workshop where a group cognitive map was finalised and agreed. Existing methods of elicitation were used in combination with the analysis of network community structure, to facilitate the validation of the pairwise interactions in the cognitive map. The validation process was introduced to frame discussions about perceived differences between interacting pairs, and to attempt to resolve any differences. The scope of the methods used is outlined in Figure 3.1 along with the relevant chapter section numbers.

This chapter addresses research question 1;

What are the key interacting social and ecological factors that are needed to represent blanket peatlands as a complex system?

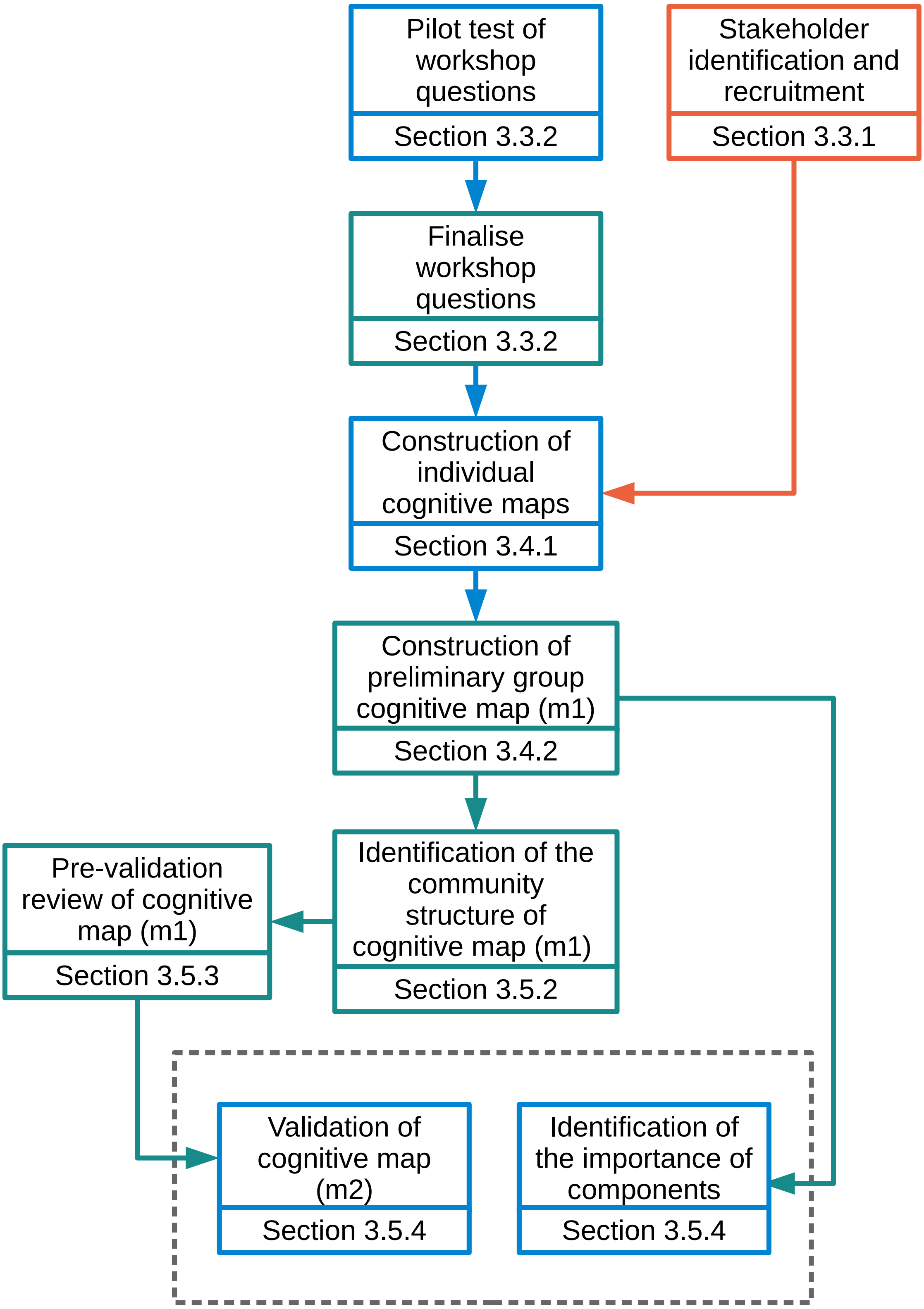


Figure 3.1. Process used to construct and validate a group cognitive map. Blue boxes represent workshop activities, green boxes represent pre– or post–processing work and the orange box represents the selection and recruitment of stakeholders. (m1) denotes the pre–validated cognitive map and (m2) the post–validated map. The dashed grey box identifies the thesis objectives addressed by the process.

3.2 Introduction

3.2.1 Conflicts in land use: addressing the challenge with mental models

Translating global targets for ecosystem restoration into actions on the ground is challenging and can be made more difficult by competing land–use objectives. One such challenge is in land-use decisions for peatlands. Peatlands are globally important for carbon storage and biodiversity, but have been subjected to degradation because of land use (e.g. Moore et al. 2013; Schimelpfenig et al. 2013; Douglas et al. 2015), and conflict at local scales is common where livelihoods are challenged (e.g. Maltby 2010; Tolvanen et al. 2013; Noordwijk et al. 2014). In the UK the shift in emphasis from blanket peatlands as agricultural landscapes to important stores of carbon, and the drive to improve water quality at source, has resulted in conflict between groups with different land–use goals. One approach that may help address conflicts over land use is for stakeholders to participate in bottom–up collaborative processes that seek to include disparate sources of knowledge, and the perspectives from those whose livelihoods are perceived to be negatively affected (Biggs et al. 2011; Redpath et al. 2013).

Studies have shown that including disparate knowledge sources from stakeholder groups can lead to new insights, the alignment of mental models and even consensus (Rouwette and Vennix 2006; Biggs et al. 2011; Scott et al. 2013). As a result, the elicitation and sharing of mental models, *"the cognitive frameworks people use to interpret and understand the world"* (Biggs et al. 2011), is often used in participatory processes to identify how a group or an individual perceives the causal dynamics of a system (e.g Doyle and Ford 1998; Elias 2008; Etienne et al. 2011). Mental models can be obtained indirectly or directly. Indirect methods use previously written documents or analysis of interviews (e.g. Axelrod 1976; Carley and Palmquist 1992), whereas direct methods require participants to define the causal relationships themselves usually as some form of diagram such as causal loops (Elias 2008), influence diagrams (Moon and Adams 2016), cognitive maps (Upham and García Pérez 2015), or fuzzy cognitive maps (Christen et al. 2015). The relationships captured with direct methods are verified by the participant in real–time or during the diagramming; a process missing with indirect methods (Jones et al. 2011). Other approaches exist that combine interviews, questionnaires and diagramming in one or more stages, for example consensus analysis (Stone-Jovicich et al. 2011) or the participatory modelling study by Prell et al. (2007) that used interviews and published literature to later create a cognitive map of social–ecological processes for blanket peatlands in the Dark Peak area of the Peak District National Park in the UK.

As an alternative to the approach used by Prell et al. (2007), I used fuzzy cognitive mapping (FCM) to directly elicit mental models from individuals or small groups to build and validate a

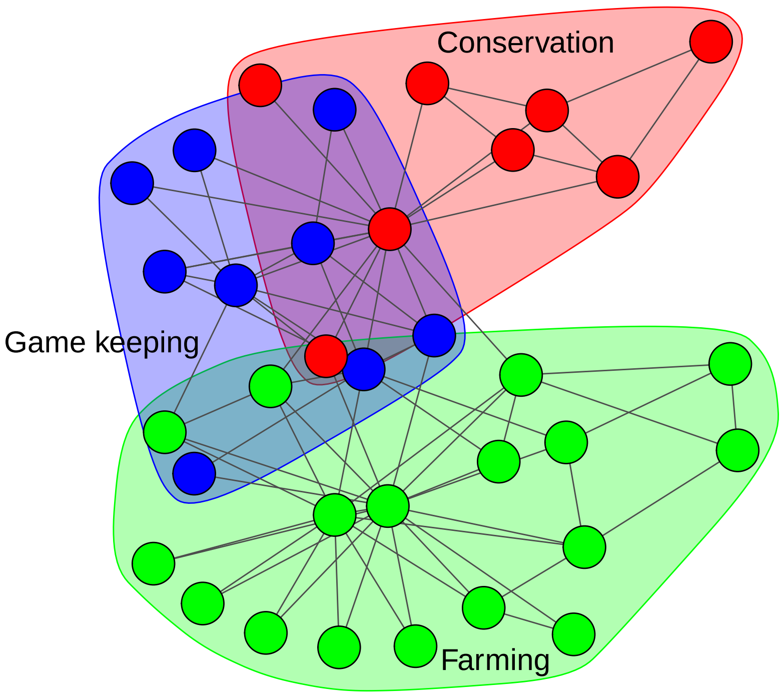


Figure 3.2. The concept of overlapping knowledge when combining cognitive maps. In this hypothetical example, three cognitive maps created by stakeholders from game–keeping, conservation and farming have been combined to create a model of blanket peatland interactions. Some areas of knowledge overlap whilst others add new connections (Gray et al. 2012). Network modified from Csardi and Nepusz (2006).

group cognitive model of the causal interactions within a UK blanket peatland.

3.2.2 Rationale for the approach taken

People's representations of the world are incomplete, inconsistent and possibly inaccurate (Jones et al. 2011), but combining representations of mental models, as shown in the example in figure 3.2, provides an opportunity to increase the reliability of a model system (Kosko 1988), incorporate multiple sources of knowledge, and provide a framework for consensus through the process of sharing and discussion (Rouwette and Vennix 2006).

Fuzzy cognitive mapping (FCM) was chosen to elicit stakeholder mental models because (1) combining multiple FCMs from a wide range of knowledge sources is a straightforward process (Kosko 1988). (2) The concept of building a causal diagram is easily understood, there is no need to think in terms of equations, and diagrams can be completed in a relatively short period of time (say around 90 minutes). (3) The construction of a map is intuitive, but not trivial because the process requires the participant to think deeply about their assumptions of the relationships between the components they include (Jones et al. 2011; Gershenson and Niazi 2013; Scott et al. 2013). Diagramming also requires participants to be explicit about relationships between components which involves examining, perhaps long held, causal beliefs. (4) Because FCMs can be easily combined, they can also be easily updated with new knowledge. (5) FCMs are also useful in situations where there are ‘wicked’ problems and finding solutions is particularly problematic (Rittel and Webber 1973; Özesmi and Özesmi 2004). (6) Finally, a combined group FCM is in fact a network of weighted interactions within the system of interest and can be analysed, not only using the typical modelling associated with FCMs (e.g. Penn et al. 2013), but also with the network analysis tools that have provided new insights into the structure of complex systems (e.g. Newman 2010; Barabási et al. 2011; Penn et al. 2014). Box 1 describes the terminology of networks and graphs used in this thesis.

BOX 1 · Chapter terminology. Cognitive and fuzzy cognitive maps are graphs. The analysis of network structures is underpinned by graph theory (Barabási and Oltvai 2004) which has been used since the 1700’s (Boccaletti et al. 2006) to solve problems about how objects relate to each other, including the analysis of social networks and FCMs (Kosko 1986; Newman 2010). The terminology of graph theory is used in this chapter and chapter 4 to describe the overall structure of relationships and components that were defined by workshop participants.

The concepts within an FCM (e.g. water–table depth, managed burning) are known in graph theory as nodes or vertices and the lines connecting components are known as edges, links or arcs. In FCMs, map components (nodes or vertices) are often called concepts or factors. When discussing map components I use the terms node or concept depending on context, and when discussing the connections between nodes I use the term links or interactions.

Several terms are used to describe the overall nature of graphs: undirected graphs are comprised of nodes connected by bidirectional links; directed graphs (digraphs) are comprised of nodes connected by links that are unidirectional; and signed digraphs are digraphs where links are have either positive or negative signs, and can be weighted either linguistically or numerically (Kosko 1986; Newman 2010) (Figure 3.3). FCMs are also known as directed cyclic graphs because of the inclusion of feedback cycles, a key advantage of this approach when modelling complex systems. Feed–forward approaches such as Bayesian networks are known as directed acyclic graphs.

One of the main measures used to describe graphs is that of the number of links that connect a node to its neighbours and vice versa and is known as the node degree, k. The average degree of a graph is shown as $\langle k \rangle$. In–degree and out–degree describe the incoming and outgoing links in a digraph. Each link between a pair of nodes includes the link out from a node (out–degree) and the connection in to a node (in–degree).

3.2.3 *Background to the development of fuzzy cognitive maps (FCM)*

FCMs were developed as a result of two ground–breaking pieces of work by Axelrod (1976) and later by Kosko (1986). FCMs were first proposed by Kosko (1986) to represent varying degrees of causality between the components (concepts) of a complex system. Kosko's aim was to model the dynamics of a system where, even if the direct relationship between concepts is certain, the propagation of effects through time is almost impossible to understand due to the interaction of feedback loops (Carvalho 2013). FCMs were a development of Axelrod's work on cognitive maps which sought to represent the qualitative dynamics of social systems. Axelrod (1976, p.56) posits that a cognitive map is *"a particular kind of mathematcial model of a person's belief system* [...] *derived from assertions of belief"*. The interaction between two concepts within a cognitive map is seen as a positive or negative causal assertion; where a positive assertion, $\odot \xrightarrow{+} \odot$, represents an increase; and a negative assertion, $\odot \xrightarrow{-} \odot$, a decrease in the connected component (Axelrod 1976, p.59). These diagrams are known as signed directed graphs or digraphs (Figure 3.3a).

Axelrod goes on to discuss the possibility of using values (weights) in conjunction with the sign of the link (signed weighted digraph; Figure 3.3b), and identified that causality could be indeterminate if, for example, there are both positive and negative connections to a concept $\odot \xrightarrow{+} \odot \xleftarrow{-} \odot$. Although the addition of weights would allow causality to be calculated, Axelrod rejects this approach mainly because of the difficulty in determining this weight from documentary evidence.

Kosko (1986) asserted that there is often a degree of uncertainty associated with knowledge representations which can be described as fuzzy; and that this fuzziness makes processing representations of knowledge more difficult. FCMs were proposed as a way to address this problem. Kosko also initially rejected the signed weighted digraph approach rejected by Axelrod (1976) because of the difficulty when determining real numbers to use as causal weights. Instead Kosko

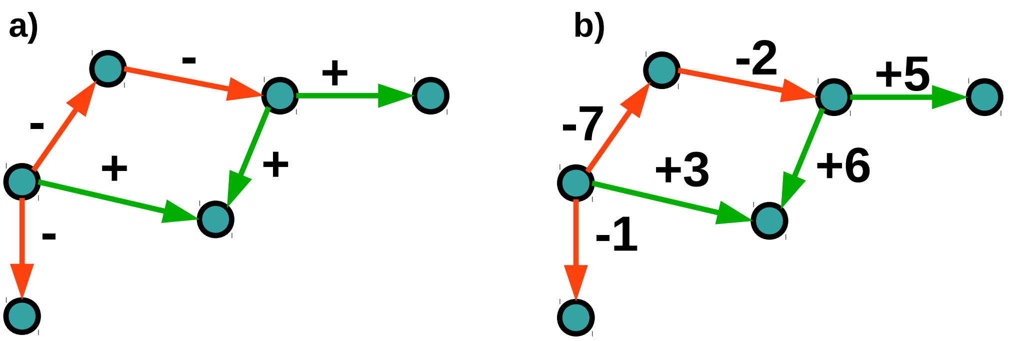


Figure 3.3. Directed graphs (digraphs). **a)** Signed digraph. **b)** Signed weighed digraph. As a cognitive map, circles represent concepts, orange arrows represent negative causality and green arrows represent positive causality. The weights associated with the arrows in **b)** represent the magnitude of causality e.g. +3 is a threefold increase.

proposed a system of linguistic fuzzy causal algebra be used to determine weights by phrases such as *a little*, *a lot*, *usually* (Kosko 1986, fig.3). In a critique of FCMs, Carvalho (2013) concluded that, in subsequent work, Kosko (e.g. 1988) goes on to redfine FCMs by combining fuzzy logic and neural networks and returns to the idea of using real numbers to define the relationship between two concepts.

FCMs are a type of artificial neural network (ANN) with the addition of feedback cycles (Carvalho 2013; Dengel et al. 2013). Using this approach, Carvalho argued that FCMs emphasise the qualitative dynamics of a complex system over causality. It this later development of FCMs (and variations thereof) that has been used in many FCMs (Papageorgiou and Salmeron 2013). Since Kosko, FCMs have been used to develop an understanding of systems in a number of applications such as; slurry grinding (Banini and Bearman 1998), ecosystem management (Hobbs et al. 2002), analysing the effect of ecosystem perturbations (Ramsey and Veltman 2005), predator management (Dexter et al. 2012), developing land cover scenarios (Soler et al. 2012), modelling of fisherman behaviour (Wise et al. 2012), and agricultural policy design (Christen et al. 2015).

3.2.4 Modelling systems with fuzzy cognitive maps

FCMs describe the causal assertions of experts which are represented as diagrams and mathematical models (Carvalho 2013). The diagramming (mapping) stage is carried out to formulate the parameters of the mathematical model, i.e. the system concepts to be modelled and their relationships to each other. As an example, in this section I use a subset of the simplified conceptual model of anthropogenic impacts on blanket peatlands from Section 2.6, shown in Figure 3.4.

Concepts are given a value [0,1] or [-1,1] to represent their activation status (the former is used here); and the link (relationship) between two concepts is given a value [-1,1] to represent the causal interaction between the two (Groumpos 2014). The FCM is elicited from stakeholders in three steps (1) identification of concepts, (2) connection of concepts and (3) specification of weights between concepts. These steps may be done in sequence or together (Jetter and Kok 2014). It is likely that there will be some iteration during the process as concepts, connections and weights are added, removed or adjusted. Some studies use a pre-determined list of concepts generated by the researcher (e.g. Fairweather 2010), which has the advantage that all participants use the same list of concepts. However, the diversity of concepts is likely to be limited, and Axelrod (1976, p.6) asserts that there should be no advance specification of concepts.

The relationship between concepts can be determined by participants directly using numerical values between, for example, [-1 ,1] or [-10,10] (Özesmi and Özesmi 2004; Christen et al. 2015), or linguistic values (e.g. weak, medium, strong; Penn et al. 2013) (Figure 3.4). Wenstøp (1980)

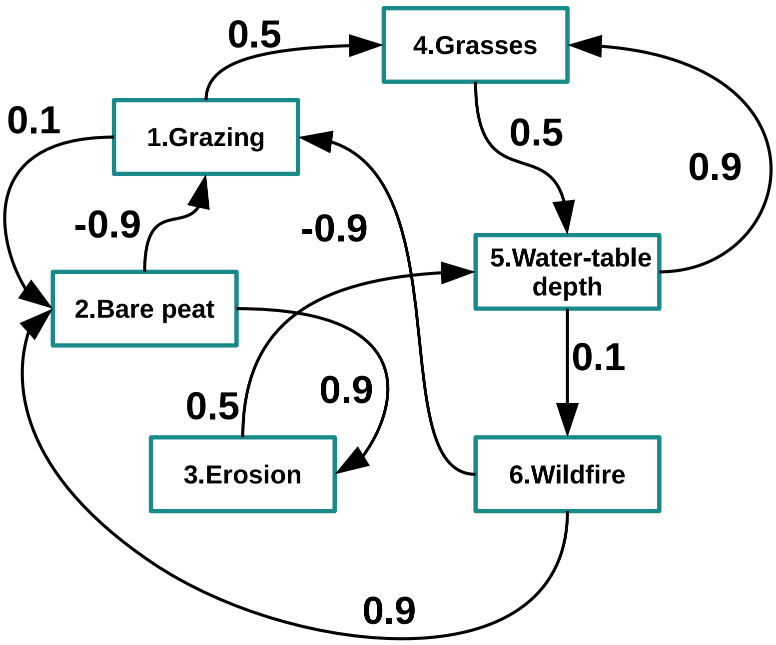
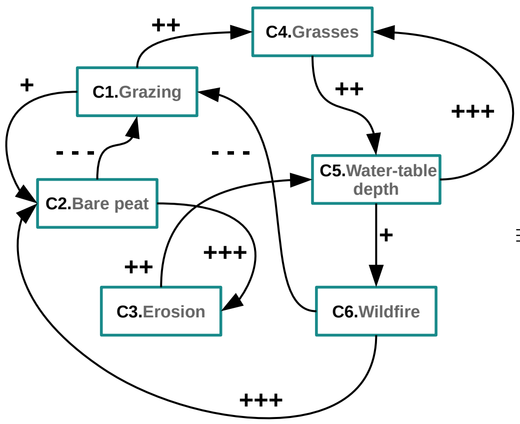


Figure 3.4. Fuzzy cognitive map. The concepts and connections were derived from a conceptual model of anthropogenic impacts on blanket peatlands Section2.6, Figure 2.10. Relationships (weights) [-1,1] have been added to connections to illustrate the magnitude of the causal effect.

proposed that linguistic values are more intuitive in applications where people are asked to make assessments because they are fuzzy. However, Wenstøp's fuzzy set approach is complex and therefore, in many FCM applications, linguistic values are translated directly to numerical values for modelling purposes. FCMs are not qualitative models (Carvalho 2013; Jetter and Kok 2014). Because the process uses qualitative information directly in a numerical context and values can only be interpreted relative to one and other, FCMs are said to be semi-quantitative (Kok 2009; Penn et al. 2013). The weights (*w*) given to relationships between FCM concepts determine the amount of change in subsequent concepts (Groumpos 2014). So, FCM inference is about the magnitude of change that occurs in the modelled system. Carvalho (2013, p.11) draws attention to the causality in FCMs: *"Causal relationships in causal maps should always involve change"*. For example, the relationship $\text{WILDFIRE}\xrightarrow{\mathbf{0.9}}\text{BARE PEAT}$ from Figure 3.4 implies that, a greater amount of wildfire causes a large (*w* = 0.9) increase in bare peat (i.e. signifying the extent of change).

For modelling purposes, the digraph created by a participant is coded into a connectivity (adjacency) matrix. The FCM from Figure 3.4 would be coded into the following connectivity matrix (*W*); where C_1 ... C_6 refers to the numbered concepts and w_{ij} the causal weight associated with each directed edge. Causality flows in the direction of the *i*th row to the *j*th column. There is a causal increase if $w_{ij} > 0$, a causal decrease if $w_{ij} < 0$ and no causality if $w_{ij} = 0$ (Groumpos 2014).



$$\equiv W = \begin{array}{cc} & \begin{matrix} C_1 & C_2 & C_3 & C_4 & C_5 & C_6 \end{matrix} \\ \begin{matrix} C_1 \\ C_2 \\ C_3 \\ C_4 \\ C_5 \\ C_6 \end{matrix} & \begin{bmatrix} 0 & 0.1 & 0 & 0.5 & 0 & 0 \\ -0.9 & 0 & 0.9 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.5 & 0 \\ 0 & 0 & 0 & 0 & 0.5 & 0 \\ 0 & 0 & 0 & 0.9 & 0 & 0.1 \\ -0.9 & 0.9 & 0 & 0 & 0 & 0 \end{bmatrix} \end{array}$$

FCMs can be developed in groups or by individuals. There are a number of advantages and disadvantages related to each approach. In groups, participants can work together which may result in shared insight, but group dynamics may discourage some people from becoming involved and result in less variety in the final map (Kosko 1988; Özesmi and Özesmi 2004; Gray et al. 2012). Individuals can define their own areas of expertise and avoid the reduction in complexity, variety and reliability, and the increase in bias that could occur during group work (Kosko 1988). However, the aggregation of a number of individual FCMs could result in the loss of some connections if causality has been specified in opposite directions (i.e there is disagreement) (Jetter and Kok 2014). For example, A $\xrightarrow{\mathbf{0.6}}$ B + A $\xrightarrow{\mathbf{-0.6}}$ B = A $\xrightarrow{\mathbf{0}}$ B.

In this study, a significant concern was that stakeholder disagreements, of the nature encountered by Dougill et al. (2006, p.270), might preclude the completion of an FCM in a group context: as a result Dougill et al. suggested working with groups of like–minded stakeholders in the first instance. In order to address this and the issue of opposing causality, it was decided to elicit FCMs from individuals or, where the stakeholders had the same land–use goals, from small groups. A validation process was implemented to gain agreement from all stakeholder groups about the components that should be included, the direction of causality, and to realise the benefits of sharing insights and understanding. In order to prevent the loss of connections between separate FCMs, links between concepts that had been defined in opposite directions were identified, discussed and agreed during model validation.

Three main steps were used to develop the final fuzzy cognitive map. (1) Preparation: recruit participants with knowledge and experience of blanket peatland management and conservation. Identify a set of questions for FCM generation (Section 3.3). (2) Model co–development: collect FCMs from participants and processes the data to create a group FCM (Section 3.4). (3) Model validation: carry out a review of the group FCM, identify pairs of concepts with opposing causality and analyse the community structure. Ask participants to validate the concepts and links (Section 3.5). The co–developed model was later used to investigate how stakeholders would deliver three different land–use objectives.

3.3 Preparation: participant recruitment and workshop questions

3.3.1 Participant recruitment

Stakeholders have been involved in modelling organisational systems since the 1960s (Voinov and Bousquet 2010), and have become increasingly involved in model building to address problems in natural resource management (Krueger et al. 2012; Bommel et al. 2014) including where stakeholders have conflicting goals (e.g. Elston et al. 2014). To ensure that the relevant people are invited to join participatory processes, Reed et al. (2009c) emphasised the need for systematic analysis of stakeholders and propose that this can be achieved by identifying, categorising and investigating stakeholders and their relationships.

The blanket peatlands of the South Pennines (including the Dark Peak area of the Peak District) have been the subject of a number of participatory projects since 2005 which were carried out as part of the Sustainable Uplands project funded by the Rural Economy and Land Use programme (RELU) (e.g. Reed et al. 2013b). As part of this project, stakeholder analysis identified eight groups; water companies, the recreation industry, the farming community, conservationists, grouse moor managers and landowners, the tourism industry, forestry and statutory bodies (Dougill et al. 2006; Prell et al. 2009). Following (Dougill et al. 2006), I reviewed these groups with employees of the Moors for the Future partnership and an updated list of stakeholders was produced: we also added a group for peatland scientists. As a result of this review, I categorised stakeholder groups according to perceived knowledge of the causal interactions within blanket peatlands in a simplified version of the interest–influence matrix (Grimble and Wellard 1997; Reed et al. 2009c) (Figure 3.5). As a result tourism and recreational stakeholders were not invited to participate in cognitive mapping. The approach used to capture cognitive maps meant that additional cognitive maps from other groups could be added prior to validation if deemed necessary.

The selected stakeholder groups are highly heterogeneous in their goals and responsibilities (Reed et al. 2013b), and landowners include organisations interested in conservation, water companies and those with farming and game–keeping interests. To further complicate matters, a single organisation may have multiple objectives; for example, the National Trust is a landowner with tenant farmers and also has a focus on blanket peatland restoration, and a number of grouse moor owners have also embarked on restoration programs (e.g. *Ancient peatlands grow again* 2015). Between and within stakeholder groups there is a wide and disparate source of technical, process and experiential knowledge about the social and ecological interactions that shape blanket peatlands.

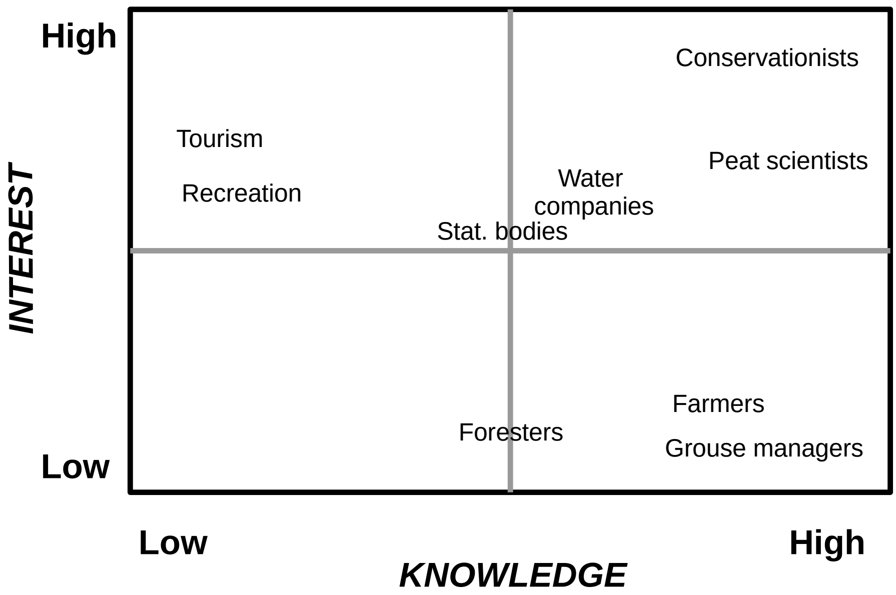


Figure 3.5. Interest–knowledge matrix. Simplified and adapted version of the interest–influence matrix (Reed et al. 2009c) for identifying stakeholder groups for cognitive mapping. Knowledge refers to the social–ecological causal relationships within blanket peatlands.

The aim of the group cognitive map was to bring together these different knowledge sources and aggregate them: some concepts within individuals' maps would be shared and others would add new concepts and/or new links, increasing the diversity of the group cognitive map (Gray et al. 2012).

The first stage of the process was to identify a group of people with relevant knowledge who would be willing to contribute to the co–development of a group cognitive map. Following identification of appropriate participants, contact was made via telephone and/or email and an invitation sent to ascertain if the person identified would be prepared to join a working group, and to identify additional individuals via snowball sampling which is commonly used to extend and refine stakeholder identification (e.g. Klerkx and Proctor 2013; Jetter and Kok 2014; Couix and Gonzalo-Turpin 2015). Potential participants were invited to a series of workshops where the end goal was to define options for the future land use of upland peatlands in order to maintain or increase carbon storage. The aim of the first workshop was stated as 'determining the positive and negative effects of the interactions of management and ecology'.

A total of 38 stakeholders were contacted with the aim of recruiting at least two people to represent each of the main groups identified above. A number of individuals requested payment in order to attend the workshops. This request was rejected by Moors for the Future because they were concerned that it might set a precedent for future participatory events. However, the impact of this decision was to reduce the number of people from land manager communities who were willing to attend daytime workshop events. But because of the flexibility of the mapping process, additional

Table 3.1. Stakeholder background for cognitive mapping

Group	WS1	WS2	EVE	IND	Total
Farming	1		6		7
Game–keeping	1			1	2
Water industry	2	1			3
Government*	1				1
Statutory agency†		4			4
PDNPA	1	1			2
Conservation	3				3
Restoration	2				2
Scientist	2			1	3

IND = one-to-one mapping session, WS = workshop, EVE = evening workshop.
* Defra
† Includes Natural England, Environment Agency & Forest Enterprise

maps were captured from individuals who did not attend the workshops at a time and place of their choice, and also during an evening workshop. The final number of stakeholders that attended cognitive mapping events was 27: a breakdown of stakeholder groups is shown in Table 3.1.

3.3.2 Development a set of questions and 'start concepts' for mapping workshops

FCMs are generated in response to questions related to the causality of the system and interactions being investigated (e.g. Özesmi and Özesmi 2004; Wise et al. 2012). In order to test the suitability of the question to be used in mapping workshops, and to review the concepts generated by the group in response to the question, a pilot workshop, that did not include anyone from the stakeholder group identified in Section 3.3.1, was run. I also selected five concepts as 'start concepts' for the stakeholder mapping workshops: start concepts are given to participants at the starting of the mapping process and can be used to activate the generation of knowledge for the particular topic area (Jetter and Kok 2014).

Seven people with a background in peatland science attended the workshop. In this case a two–part question was tested to set the scope of the topic under consideration, but was also broad enough so that participants were not restricted in the subject matter that could form a concept: *"What are the variables that can be used to describe the ecology and hydrology of upland peat and associated land management? How do these variables interact with each other?"* A mean of 16.9 (from 11 to 24) concepts per person were generated. However, a significant amount of guidance was needed and most people in the group felt that this was because the question was too general. Based on this feedback, a set of five questions was developed to encompass water movement, vegetation abundance, domestic animals and wildlife and the accumulation of peat.

Final mapping workshop questions

Consider these factors within the Peak District:

- "What are the factors that influence the way water; 1) is stored in peatlands or; 2) moves from upland peatlands into adjacent rivers?"
- "What are the factors that influence the way different types of vegetation increase or decrease in abundance on upland peatlands?"
- "What are the factors that influence the survival and abundance of the domestic animals and wildlife that live on upland peatlands'
- "What are the factors that influence the accumulation of upland peat?"

What are the interactions?

- "How do the factors you have identified above interact with each other?"

Negative influence			Positive influence		
- - - Strongly negative	- - Negative	- Weakly negative	+ Weakly positive	+ + Positive	+ + + Strongly positive

Figure 3.6. Rating scale for strength and direction of causality in stakeholder fuzzy cognitive maps. An alternative scale of 'a little', 'some', 'a lot', was considered but it was decided that the combination of sign and strength would reinforce the direction as well as magnitude of causality.

Selected start concepts

1. Increased rainfall
2. Intensity of grazing by sheep
3. Dominance of grasses
4. Rate of peat decomposition
5. Water–table depth

3.4 Model development: data collection and processing

3.4.1 Fuzzy cognitive mapping workshops

The process used during mapping workshops was developed from the published scientific literature (including Özesmi and Özesmi 2004; Gray et al. 2012; and an early view of Jetter and Kok 2014). Three workshops were held to obtain FCMs from the stakeholders identified in Table 3.1. Two comprised stakeholders from a cross section of the identified groups, and one was an evening workshop attended by six farmers from the Dark Peak area of the Peak District. Two additional meetings were held for individuals who were not able to attend workshops (Table 3.1).

At the beginning of the mapping process, a presentation was given to outline the aims and objectives of the complete stakeholder engagement process and for the workshop. An example of a FCM from an unrelated topic was drawn to demonstrate how a map should be developed (Gray et al. 2012). A set of instructions was provided to each participant which included the five questions identified in Section 3.3.2, along with a supply of pens, pencils, Post–It® Notes and paper. An explanation of the five start concepts (Section 3.3.2) was also given, and a set provided to each participant. The instructions for the workshop were reviewed with the group or individual.

Participants were asked to (1) consider the five questions and develop a set of concepts which could include the start concepts (inclusion was not mandatory); (2) review the list of concepts and combine or discard where appropriate. Participants were asked to clarify any questions they had about concepts at any time; (3) define the connections between concepts (including feedback cycles); (4) define the causality of the relationship. A six–point scale was used from strongly negative to strongly positive (Figure 3.6) (Kok 2009). I asked participants to consider how *an increase(decrease) in 'A' causes an increase(decrease) in 'B'*; and (5) review the map and make any amendments (add, remove or modify concepts and causal connections). At the end of the mapping process participants were encouraged to review the maps completed by others (Figure 3.7).

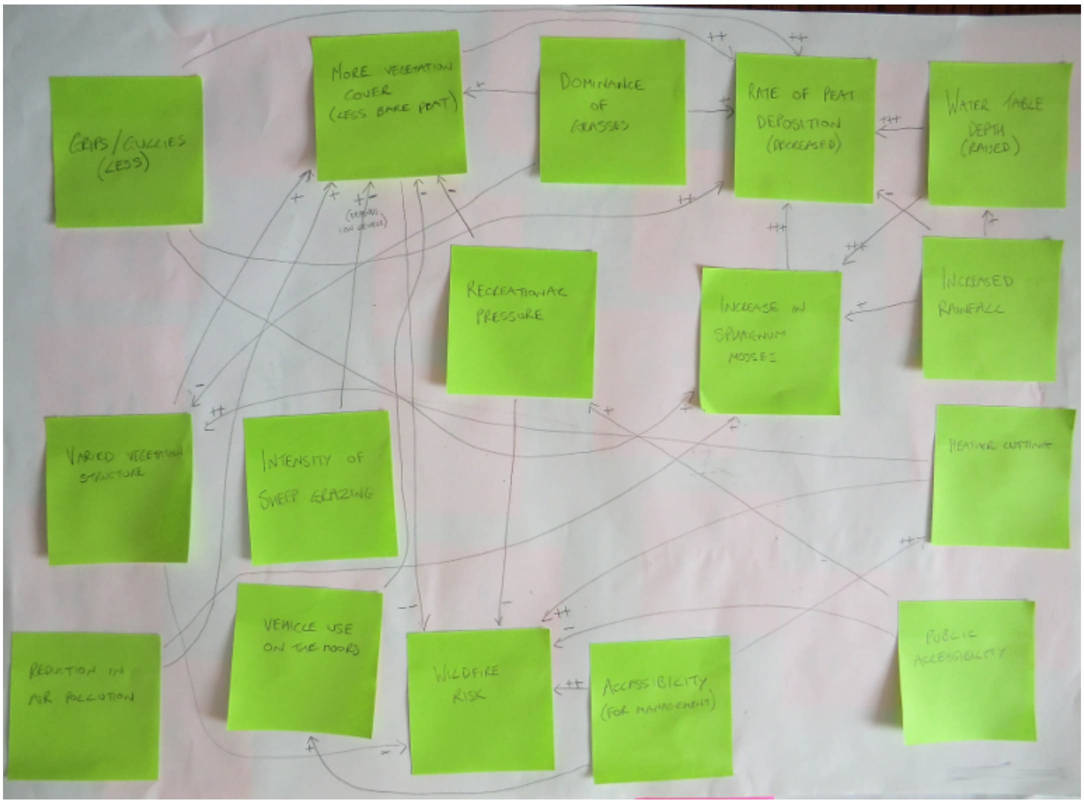


Figure 3.7. Two fuzzy cognitive maps developed during participatory workshops.

3.4.2 Coding Workshop outputs

Twenty–one completed maps with a total of 390 concepts were obtained from workshops and individual meetings. The number of concepts (C) per map varied from 15 to 50 and the number of connections from 19 to 122. Each map was reviewed and coded into a square connectivity (adjacency) matrix (W) using the scale from Figure 3.6, similar to Penn et al. (2013), to convert causal linguistic values into numerical values (w_{ij});

$$w_{ij} = \begin{cases} +0.9 & \text{if the connection} = +++, \\ +0.5 & \text{if the connection} = ++, \\ +0.1 & \text{if the connection} = +, \\ -0.1 & \text{if the connection} = -, \\ -0.5 & \text{if the connection} = --, \\ -0.9 & \text{if the connection} = ---, \\ 0 & \text{otherwise} \end{cases}$$

The connections between each pair of concepts in the connectivity matrices and workshop maps were checked by two people. Individual connectivity matrices were aggregated into a group matrix of $n \times n$ dimensions where n was equal to the total number of aggregated concepts from all individual maps.

Prior to aggregation, the contribution of each participant can be weighted [0, 1] according to expertise or uncertainty, or based on the assessment of the participants themselves (Özesmi and Özesmi 2004; Papageorgiou and Salmeron 2012). Algorithms have also been developed that can adjust the causal weights assigned by participants using learning rules (e.g. Groumpos 2014). Both of these approaches were rejected because one of the aims of this research was that participants co–develop and validate a blanket peatland model, and the use of a 'black box' process that adjusted their contributions would have negated the work done by participants to validate concepts and connections. Hence, all FCMs were assumed to to have the same contribution to the aggregated FCM (i.e. a weight of one) (Özesmi and Özesmi 2004).

Aggregation involved two processes. The first was qualitative, where similar concepts were identified and combined into a single concept (e.g. Gray et al. 2012). Additionally, concepts that were expressed slightly differently but had the same meaning, in the context of the original FCMs, or would have been the same if causality was reversed, were combined (Nakamura et al. 1982) (Figure 3.8a, b). Where I had any doubt, I contacted the originator of the map and asked for clarification. In the second process, FCMs were combined by adding connectivity matrices

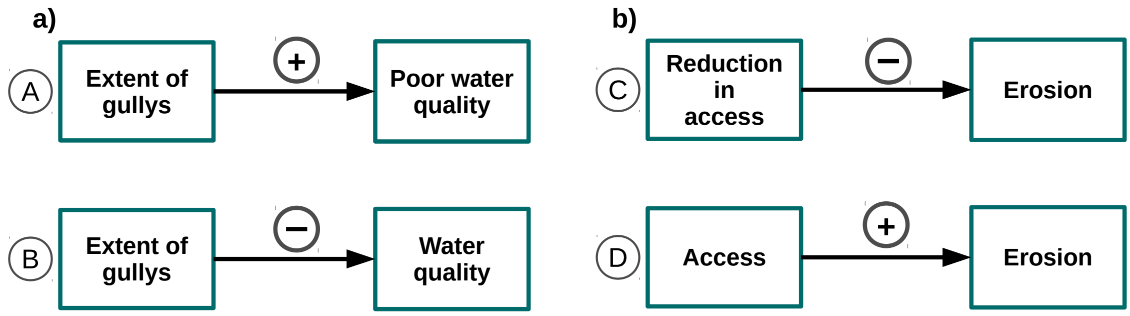


Figure 3.8. Coding fuzzy cognitive maps. Examples of map simplification through combination and examination of causality. **a)** (A) An increase in the extent of gullys causes an increase in poor water quality. (B) An increase in the extent of gullys causes a decrease in water quality. Implementing the approach in (B) facilitated the removal of poor water quality as a concept for a number of other connections. **b)** (C) An increase in the reduction in access causes a reduction in erosion. The approach adopted in (D); an increase in access causes an increase in erosion, is more intuitive and enabled reduction in access and access to be combined in a number of cases.

(Kosko 1988) and the link weights normalised back to [-0.9, 0.9] . All concepts had at least one weighted link; the minimum required to infer causality. Following aggregation, the connectivity matrix comprised 135 concepts connected by 581 links (Figure 3.9). The mean number of links per concept (degree) $\langle k \rangle$ was 8.6: the extent of*Sphagnum* cover was the most connected concept ($k = 52$).

3.5 Model validation: group FCM analysis and validation workshop

3.5.1 Validation objectives

When used in modelling, the term validation usually refers to a process of checking outputs to determine if the model is a plausible representation of the modelled system. Here I use the term to refer to the process of checking, by stakeholders, the concepts, links and weights of the group FCM (e.g. Penn et al. 2013). The overarching aim of validation was to produce a shared understanding of the group FCM, build ownership of the model and achieve consensus about the causal relationships. The validation process was completed in a workshop setting. A mixed group of 12 participants, mainly from the original mapping group (seven of the 12) took part in FCM validation. The main objectives were to (1) review and agree the concepts and link weights of the group FCM, including those identified to have opposing causality; and (2) define a number of FCM subsets for later use and analysis. The subsetting criteria was based on five classes of the importance of each concept in relation to the maintenance or increase of blanket peat in the Peak District. The importance ranking was defined as; highly important (5), very important (4), important (3), some importance (2) and unimportant (1) for maintaining or increasing stores of peat.

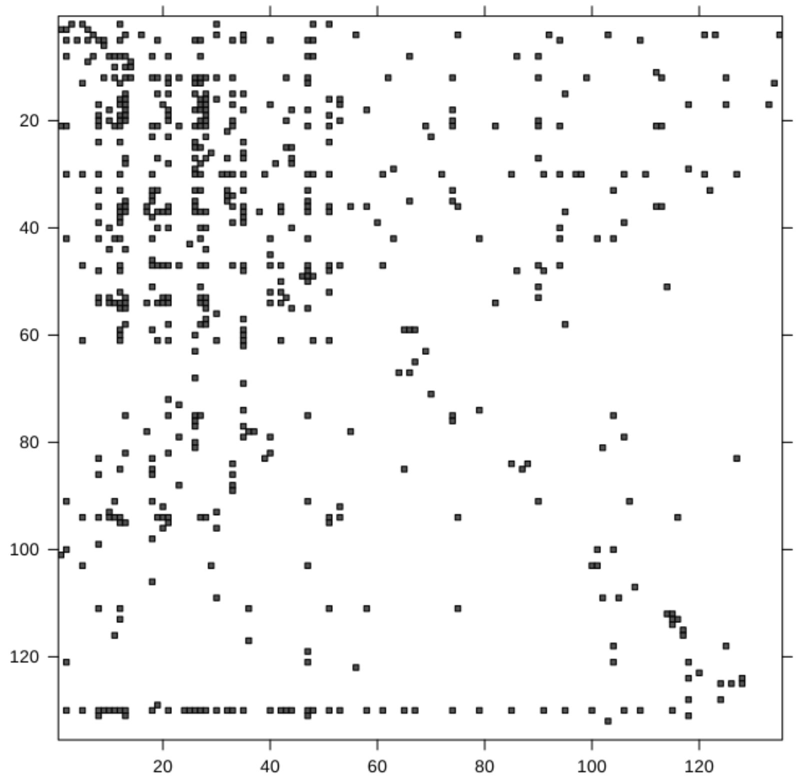


Figure 3.9. Group connectivity matrix. Aggregated workshop FCMs combined into a group FCM of 135 concepts and 581 links. Concepts were added starting in the upper left, $C_1 \rightarrow C_1$, location of the matrix. New concepts became less frequent with each additional FCM which accounts for the increasing sparseness of the matrix. Each point in the matrix represents one of the links between concepts and causality flows from row to column.

I assumed that the validation workshop would be challenging to complete within the limited time available, and so three tasks were carried out in preparation for the workshop in order to achieve validation objectives: (1) the group FCM was split into clusters of concepts and relationships so that different 'chunks' of the FCM could be worked on by participants at the same time; (2) with a peatland expert, I reviewed all concepts and connections, and (3) pairs of concepts with opposing causality were identified.

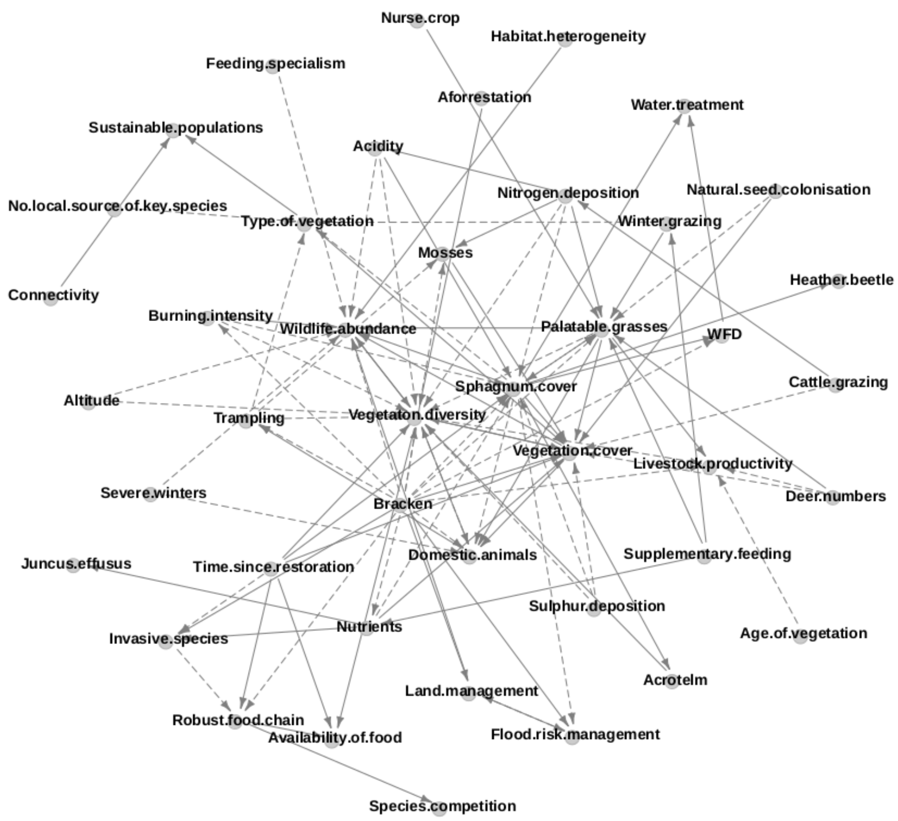
3.5.2 Clusters: identification of community structure

When transformed back into a FCM, the aggregated connectivity matrix represented a complex structure that would have been difficult to visualise and interpret in a workshop setting. Given this complexity, my aim was to divide the FCM into manageable sub-graphs that were convenient to use within the validation workshop. To achieve this aim, the FCM was analyzed for community structure (Girvan and Newman 2002). The principle underlying community detection is to produce clusters of densely connected nodes with few connections between the clusters (Newman 2010, p.354). A number of approaches have been developed to detect community structure in graphs with and without weighted links (e.g. Girvan and Newman 2002; Newman 2006; Reichardt and

Bornholdt 2006; Gulbahce and Lehmann 2008). Because the links of the aggregated FCM are both positively and negatively weighted, a detection algorithm that incorporated both signs was initially preferred. The approach developed by Reichardt and Bornholdt (2006) and Traag and Bruggeman (2009) and implemented as the *spinglass.community* algorithm (Csardi and Nepusz 2006) for R (R Core Team 2015) was used. The algorithm creates clusters by maximising the number of positively weighted links within communities and the number of negatively weighted links between communities.

Six communities were detected comprising 47, 40, 21, 20, 4 and 3 concepts from the group FCM (representing 34.8%, 29.6%, 15.5%, 14.8%, 3% and 2.2% of the group FCM respectively). Modularity (Q) is a measure of the fraction of FCM concepts connected to concepts of the same community (Newman 2010, p.222, section 7.13.1): modularity for the group FCM was $Q = 0.37$. A review of each cluster revealed that several were not fully connected. Given that the objective of the validation workshop was to review the strength and direction of links as well as concepts, any disconnected concepts within a cluster could not have been fully validated. To ensure each cluster was connected, the group FCM was reprocessed using only the direction of causality between concepts (i.e. positive or negative) and Q recalculated. Six clusters were again detected and $Q = 0.33$. Although the composition of clusters had changed somewhat (43, 39, 36, 8, 7, 2 concepts, that represented 31.8%, 28.9%, 26.7%, 5.9%, 5.2% and 1.5% of the FCM), Clauset et al. (2004) proposed that a value of Q greater than 0.3 is an indication of good community structure, and so I accepted these clusters for the purposes of the FCM validation workshop. Figure 3.10 shows an example of two of the resulting clusters.

a)



b)

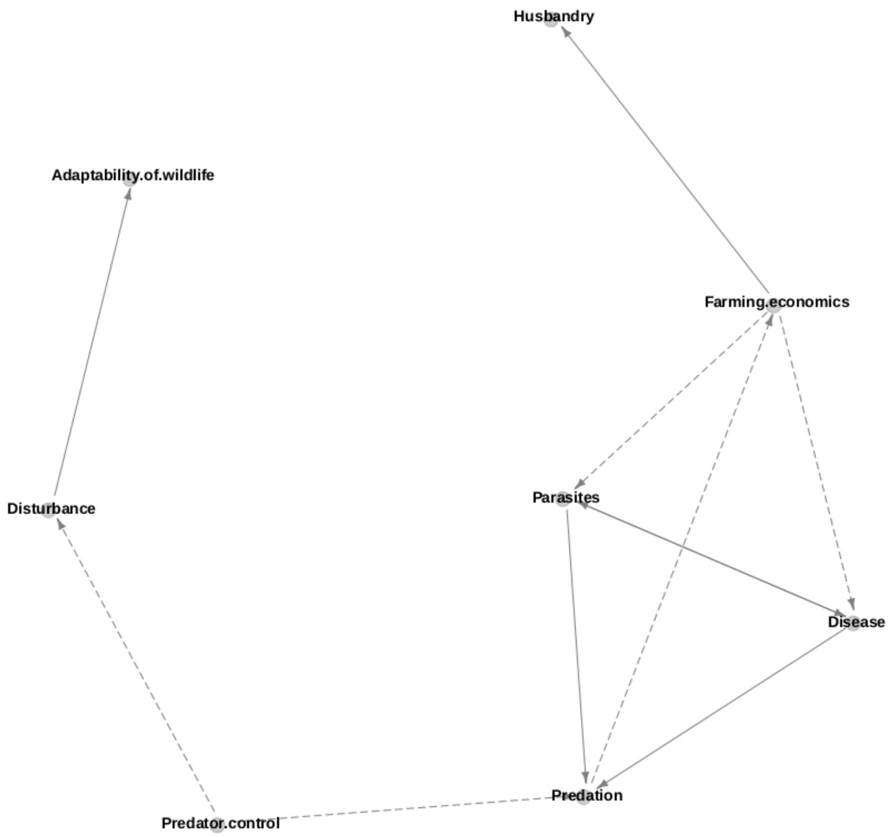


Figure 3.10. Group FCM clusters for validation. Fully connected clusters were used for the purpose of FCM validation. **a)** The largest cluster comprised 43 concepts (31.8% of the group FCM) **b)** Cluster of eight concepts (5.9% of the group FCM)

3.5.3 FCM review

All concepts, links and clusters were reviewed by two people to highlight potential concerns. These concerns were used to prioritise the review of concepts and causal relationships during the validation process. Concerns primarily included the direction of causality or concepts that were thought to be hard to work with such as geomorphology, feeding specialism, and type of vegetation. Worksheets were produced for each cluster, and the concerns highlighted for the set of positive and negative links within each cluster. The worksheets detailed each pair of connected concepts and associated link weights within each cluster.

To reveal if, during the creation of individual FCMs, any paired concepts had been specified with opposing causality (i.e. there was disagreement about causality), each participant's FCM was compared to all other participants' FCMs. The maximum and minimum weights for each causal relationship across all FCMs were stored in two matrices, one for positive causality and the second for negative causality. The two resulting matrices were compared to each other and the location of interactions with opposing causality recorded in a third matrix. This analysis identified 16 paired concepts for discussion during the validation workshop (Table 3.2).

Table 3.2. Paired concepts with opposing causality

Direction of causal relationship $\xrightarrow{\pm}$	
Stocking density	Heather cover
Stocking density	Palatable grasses
Stocking density	Vegetation diversity
Stocking density	Wildfire
Agri–environment schemes	Stocking density
Wildfire	Heather cover
Heather cutting	Heather cover
Total rainfall	Vegetation diversity
Temperature	Total rainfall
Rainfall intensity	Water storage
Vegetation cover	Water storage
Gullys	Water storage
Pipe networks	Water storage
Type of vegetation	Water–table
Water–table	Water movement
Gully blocking	Access

3.5.4 Validation

At beginning of the workshop, the processes that had been used to collect and produce the aggregated FCM was reviewed, and the scope of objectives for validation, that could be achieved within the allocated time, were discussed and agreed with participants. Diagrams of the aggregated group FCM and each of the six clusters were shown and discussed, along with the concepts that had been originally defined with opposing causality. To ensure all participants reviewed all clusters, participants were split into four mixed–background groups. The workshop took the following format; (1) Each group classified each concept for importance, (2) each group reviewed in turn the six FCM clusters and the 16 concepts with opposing causality, (3) together, all participants reviewed and agreed the importance classification from (1), and (4) together all participants reviewed and agreed any final differences in concepts, link weights or causality. The above order was adopted in order to enable two non-participants to check for any differences between groups from the review of clusters, and was done whilst (3) took place.

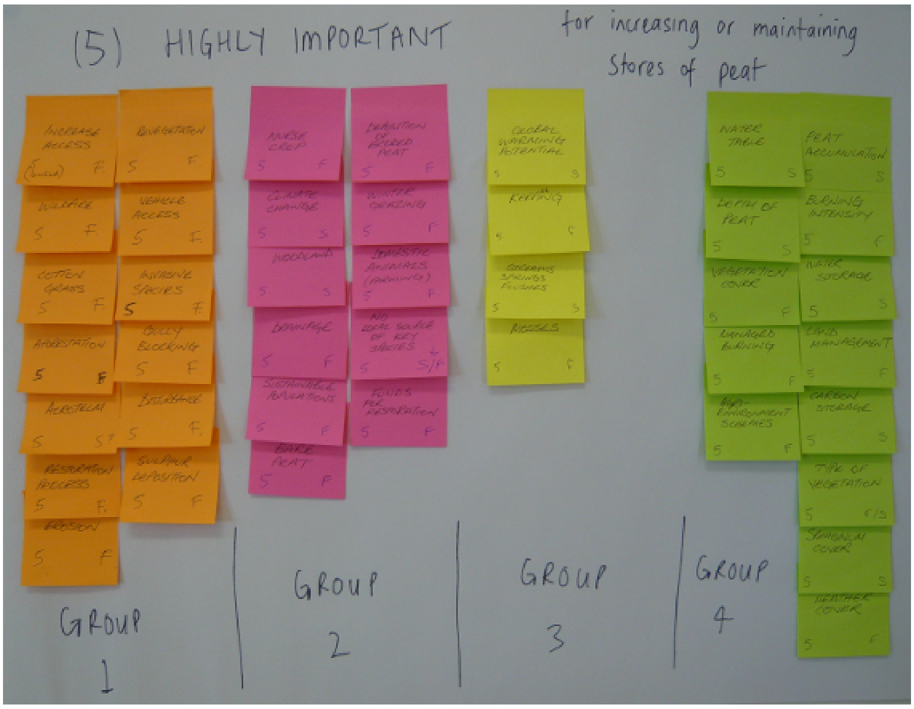


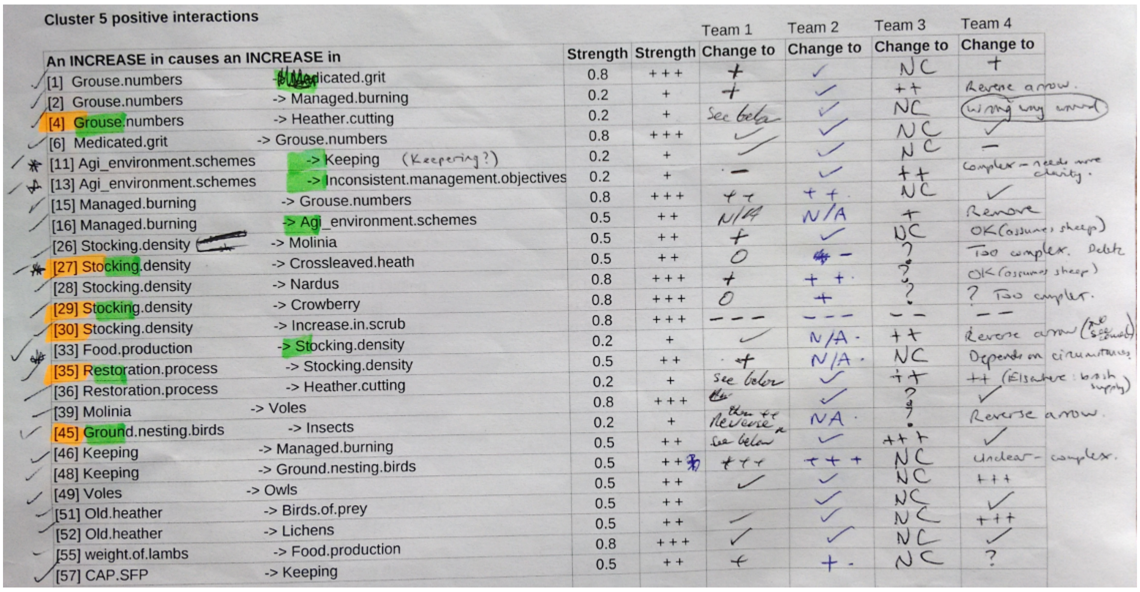
Figure 3.11. Example of the importance classification of FCM concepts. All concepts were classified according to importance for maintaining or increasing stores of peat. The example here shows the initial classification for the highly important subset prior to participant review and agreement.

Classification of importance

To rank importance (1–5), participant groups were given a full set of 135 FCM concepts (each on a separate card) and asked to review and assign an importance class to each concept. Classifications were recorded on the card and rated concepts were later organised by participant group within each class of importance (Figure 3.11). Differences between ratings for a concept were identified by removing a concept from its importance class and placing it underneath the remaining concepts in that class. Collectively, all participants were asked to review the classifications and remove concepts from their grouped class if they disagreed with the ranking given. Where differences were identified, the groups involved were asked to discuss their decision with the other groups and a final classification was agreed by all participants.

Validation of FCM concepts, weights and causality

To facilitate validation, a pack was produced for each cluster that comprised a diagram (see Figures 3.10a and b) and two worksheets detailing paired concepts (including those with opposing causality) and link weights (including both numeric and linguistic values; Figures 3.6 and 3.12). In order to avoid confusion between positive and negative causal interactions, a separate worksheet was produced for positive and negative causes so as to avoid any potential errors as a result of mixing causality on one sheet. Each group was asked to review each cluster in turn beginning



Cluster 5 positive interactions

An INCREASE in causes an INCREASE in		Strength	Strength	Team 1 Change to	Team 2 Change to	Team 3 Change to	Team 4 Change to
[1] Grouse.numbers	-> Medicated.grit	0.8	+ + +	+	✓	NC	+
[2] Grouse.numbers	-> Managed.burning	0.2	+	+	✓	++	Reverse arrow.
[4] Grouse.numbers	-> Heather.cutting	0.2	+	See below	✓	NC	wrong way round
[6] Medicated.grit	-> Grouse.numbers	0.8	+ + +	✓	✓	NC	✓
[11] Agi_environment.schemes	-> Keeping (Keepering?)	0.2	+	✓	✓	NC	−
[13] Agi_environment.schemes	-> Inconsistent.management.objectives	0.2	+	−	✓	++	Complex – needs more clarity.
[15] Managed.burning	-> Grouse.numbers	0.8	+ + +	++	++	NC	✓
[16] Managed.burning	-> Agi_environment.schemes	0.5	+ +	N/A	N/A	+	Remove
[26] Stocking.density	-> Molinia	0.5	+ +	+	✓	NC	OK (assumes sheep)
[27] Stocking.density	-> Crossleaved.heath	0.5	+ +	0	−	?	Too complex. Delete
[28] Stocking.density	-> Nardus	0.8	+ + +	+	+ +	?	OK (assumes sheep)
[29] Stocking.density	-> Crowberry	0.8	+ + +	0	+	?	? Too complex.
[30] Stocking.density	-> Increase.in.scrub	0.8	+ + +	− − −	− − −	− −	− −
[33] Food.production	-> Stocking.density	0.2	+	✓	N/A	++	Reverse arrow (see comment)
[35] Restoration.process	-> Stocking.density	0.5	+ +	+	N/A	NC	Depends on circumstances
[36] Restoration.process	-> Heather.cutting	0.2	+	See below	✓	++	++ (Elsewhere: brash supply)
[39] Molinia	-> Voles	0.8	+ + +		✓	?	✓
[45] Ground.nesting.birds	-> Insects	0.2	+	Reverse then ++	NA	?	Reverse arrow.
[46] Keeping	-> Managed.burning	0.5	+ +	See below	✓	+++	✓
[48] Keeping	-> Ground.nesting.birds	0.5	+ +	+++	+++	NC	Unclear – complex.
[49] Voles	-> Owls	0.5	+ +	✓	✓	NC	+++
[51] Old.heather	-> Birds.of.prey	0.5	+ +		✓	NC	✓
[52] Old.heather	-> Lichens	0.5	+ +	✓	✓	NC	+++
[55] weight.of.lambs	-> Food.production	0.8	+ + +	✓	✓	NC	✓
[57] CAP.SFP	-> Keeping	0.5	+ +	+	+	NC	?

Figure 3.12. Validation worksheet. Each group recorded the weight or direction of causality between paired concepts along with any comments. Pairs highlighted in orange were flagged during the pre–validation review; those highlighted green were flagged during validation for collective review during the workshop.

with the concepts highlighted as part of the pre–validation review. Groups were asked to record agreement, changes in weight or causality and add comments. All groups recorded their decisions on the same worksheet (Figure 3.12).

The outcomes of cluster validation were reviewed collectively. This became an iterative process because in some instances, one group proposed to alter the direction of causality between two concepts (from say, causing an increase to causing a decrease) whilst the other groups had not. Each of these new instances of opposing causality were reviewed. At this stage the group either agreed the nature of the relationship or chose to defer the decision to an external expert because of perceived lack of knowledge or available evidence (which occurred twice); MEDICATED GRIT $\xrightarrow{-}$ INSECTS (i.e. an increase in medicated grit causes a small decrease in insects) and PEAT–ACCUMULATION $\xrightarrow{+}$ PERCOLATION RATES (i.e. an increase in peat–accumulation causes a small increase in percolation rates).

Groups were able to change the original weight (w_{ij}^{orig}) between concept pairs (including reversing causality), add new weighted connections between existing concepts or remove connections or concepts. Except in the case of new instances of opposing causality, and discussion of specific concept pairs raised by the groups, there was insufficient time within the workshop to review all changes made to all concepts. For example, in the case of FOOD PRODUCTION $\xrightarrow{+}$ STOCKING DENSITY, two groups agreed with this rating, one proposed to increase the strength of the relationship ($++$) and the fourth suggested the relationship should be removed completely ($w_{ij}^{orig} = 0$). It was therefore necessary to develop a set of rules to produce the final output from the workshop which was used to create a modified connectivity matrix. New weights (w_{ij}^{new}) were determined:

For groups G_1 ... G_n,

1. If the majority of groups proposed to remove a concept C_i, $C_i w_{ji}^{new} = 0$
   ELSE
2. If the majority of groups proposed to remove a weight, $w_{ij}^{new} = 0$,
   ELSE
3. If the majority of groups proposed to change a weight, $w_{ij}^{new} = \frac{1}{n}\sum_{k=G1}^{Gn} w_{ij}^{k}$,
   ELSE
4. If there was no change to the original FCM value, $w_{ij}^{new} = w_{ij}^{orig}$,
   ELSE
5. If there was no agreement between groups, $w_{ij}^{new} = w_{ij}^{orig}$,

The final list of concepts classed by importance for maintaining or increasing blanket peat is shown in Table 3.3.

Table 3.3. Validated cognitive map concepts listed by importance ranking

Unimportant	Some importance	Important	Very important	Highly important
Robust food chain	Predation	Wfd	Domestic animals	Sphagnum cover
Medicated grit	Cattle grazing	Deer numbers	Supplementary feeding	Bare peat
Ground–nesting birds	Habitat heterogeneity	Nitrogen deposition	Nutrients	Vegetation cover
Shooting & trapping	Severe winters	Time since restoration	Age of vegetation	Burning intensity
Clough woodland	Predator control	Natural seed colonisation	Flood risk management	Land management
Foxes, crows, mustelids	Wildlife abundance	Altitude	Invasive species	Agri–env. schemes
Water quality	Grouse numbers	Disease	Trampling	Afforestation
Heather beetle	Voles	Parasites	Farming economics	Nurse crop
Nardus	Insects	Methane consumption	Methane production	Streams, springs, flushes
Availability of food for wildlife	Slope	Methane transport	Methane emissions	Mosses
Species competition	Adaptability of wildlife	Molinia	Stocking density	Sulphur deposition
Juncus effusus	Lichens	Weight of lambs	Food production	Disturbance
Cross–leaved heath	water treatment	Heather cutting	Old heather	Carbon storage
	Owls	Increase in scrub	CAP SFP	Carbon dixoide
	Birds of prey	Temperature	SSSI agreement	Acrotelm
	Stream ecology	Microtopography	Inconsistent management objectives	Managed burning
	Hare numbers	Firebreak	Extent of grips	Restoration process
	Waders	Livestock productivity	Rainfall intensity	Keepering
		Crowberry	Gullys	Heather cover
		Pools	Water movement	Water storage
		Management access	Access	Water–table depth
		Vehicle access	Percolation rates	Depth of peat
			Public accesibility	Peat accumulation
			Pipe networks	Total rainfall
			Drought	Wildfire
			Woodland	Cotton grass
			Solar radiation	Revegetation
			Humidity	Gully blocking
			Wind	Climate change
			Evapotranspiration	Drainage
			Husbandry	Erosion
			Runoff flashiness	Funds for restoration
			Productive farming	Global warming
			Palatable grasses	Deposition of eroded peat
			Farming knowledge	
			Livestock condition	
			Acidity	
			Vegetation diversity	
			Winter grazing	

Agri–env. schemes = Agri–environment schemes. Keepering = the activities of game keepers.

3.6 Results: a co–developed model of blanket peatland causal interactions

3.6.1 Model summary

The finalised FCM developed from the validation workshop, classified according to perceived importance (Table 3.3), comprised 126 concepts and 524 links, compared to 135 concepts and 581 links for the pre–validated FCM. A visualisation of the validated FCM is shown in Figure 3.13. The mean number of connections for a concept (mean degree, $\langle k \rangle$) was 8.3 and the maximum degree (k^{max}) was 51, compared to 8.6 for and 52 respectively for the pre–validated FCM. Three additional calculations were carried out to explore the structure of the validated FCM. These were, density, interconnectedness, and assortivity, chosen because each relates to a characteristic of the FCM that may be important when determining land uses. (1) Density has been shown to be related to network controllability (Liu et al. 2011) which could help to identify the concepts to manipulate to determine land use objectives; (2) interconnectedness may give insights into network (and hence system) structure and robustness (Jeong et al. 2000), and (3) assortativity can show if a concept is likely to be connected to other concepts of the same importance class (Newman 2010).

(1) Graph density defines the number of actual connections as a fraction of the number of possible connections: $E/(C \times (C - 1))$, where E is the number of links and C is the number of concepts. The density of the validated FCM was 0.03 similar to the values for some food webs reported by Liu et al. (2011) who found that dense networks are more controllable. They estimated that ≈25–55% of nodes would need to be controlled to 'steer' networks of comparable density to a desired state. It therefore seems reasonable to expect a similar percentage would be required for the FCM. (2) Interconnectedness, defined as the mean shortest path length between all pairs (Jeong et al. 2000) was 3.35: values of between three and five have been reported for some biological networks (Jeong et al. 2000). Smaller values indicate shorter average path lengths between node pairs, which are found in many networks, and infers a speedier transfer of information, or contagion between nodes (Newman 2010, page 241). (3) Graph assortativity is the likelihood that a concept will be connected to a concept of a similar type; where a value of 1 represents complete assortativity and -1 is fully disassortative (i.e. likely to connect to dissimilar concepts) (Newman and Girvan 2003; Boccaletti et al. 2006). I investigated the likelihood that concepts were connected to concepts with a similar number of connections (k), and to concepts of the same importance class. The results were -0.014 and 0.216 respectively indicating that there was no assortativity or disassortativity in relation to the number of connections (i.e. neutral), but that nodes with the same importance class were more likely to be connected.

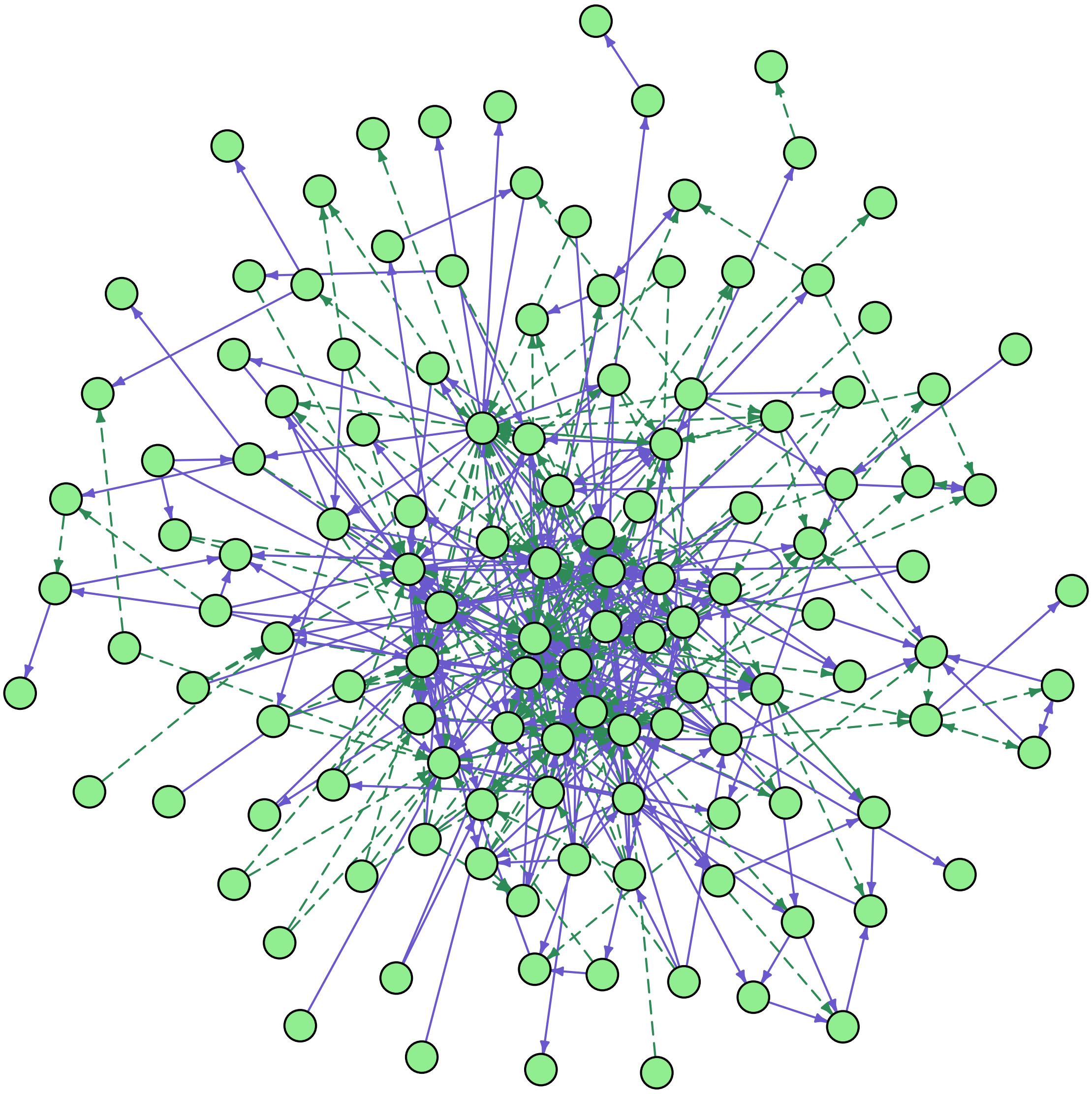


Figure 3.13. Co–developed model of blanket peatland social and ecological interactions. Final validated model of the interactions between blanket peatland ecological and hydrological processes and land–use. There are 126 nodes and 524 paired connections (links) which form a directed weighted graph. Solid lines indicate a causal increase between paired concepts; dashed lines indicate a causal decrease. Arrow heads represent the direction of causality.

3.6.2 The importance of concepts for blanket peatland persistence

To investigate differences between importance classes given to concepts, a subset of the validated model was created from scratch for each importance class (Table 3.4 and Figure 3.14a–e). All importance subsets included a number of concepts where degree (k) = 0 (i.e. disconnected from the rest of the network), found to be highest in the network of concepts ranked important (11) and lowest in the highly important network (3) (Table 3.4). The Kruskall–Wallis test was used to analyse the differences between the subset networks (R Core Team 2015). This analysis found a significant difference between at least two of the subsets and degree (H = 40.67, $p < 0.000$). To further investigate these differences, the Wilcoxon rank sum test was used and the resulting p values were corrected for familywise error using the Hochberg method. There was no significant difference between degree and subsets; unimportant (a), some importance (b) and important (c) (a and b, $p = 0.91$; a and c, $p = 0.91$; b and c, $p = 0.91$). Subsets based on classes very important (d) and highly important (e) were significantly different to other subsets classes. The highly important subset was also significantly different to the very important subset ($p < 0.05$ in all cases; Table 3.5) (Figure 3.14 f; letters a–e refer also to the panels in Figure 3.14).

The network subsets in Figure 3.14 clearly show that the number of links increases with increasing importance class (Table 3.4). The subset based on the highly important class is more connected than the other importance classes because five concepts included in this class (Sphagnum cover, water–table depth, wildfire, water storage and peat accumulation) connect 61.8% of the network. This value is similar to that reported by Jeong et al. (2000) for networks with a few highly connected 'hubs'.

Table 3.4. Summary of networks subset by importance classification

Importance class	$\langle k \rangle$	Concepts	links	k^{min}	k^{max}	$k = 0$
Unimportant	0.62	13	4	0	2	6
Some importance	1	18	9	0	5	8
Important	0.73	22	8	0	3	11
Very important	2.46	39	48	0	8	6
Highly important	7.06	34	120	0	22	3

k = concept degree, $\langle k \rangle$ = mean concept degree

links = connections

Table 3.5. Results of test for differences between importance classification*

	Very important		Highly important	
Importance class	W	p^+	W	p^+
Unimportant	108.5	< 0.010	58.5	< 0.000
Some importance	189.5	< 0.018	102	< 0.000
Important	206.5	0.004	107	< 0.000
Very important			39.1	0.02

* = Wilcoxon rank sum test used to assess degree (k) and importance correlation

p^+ = Hochberg correction applied for familywise error

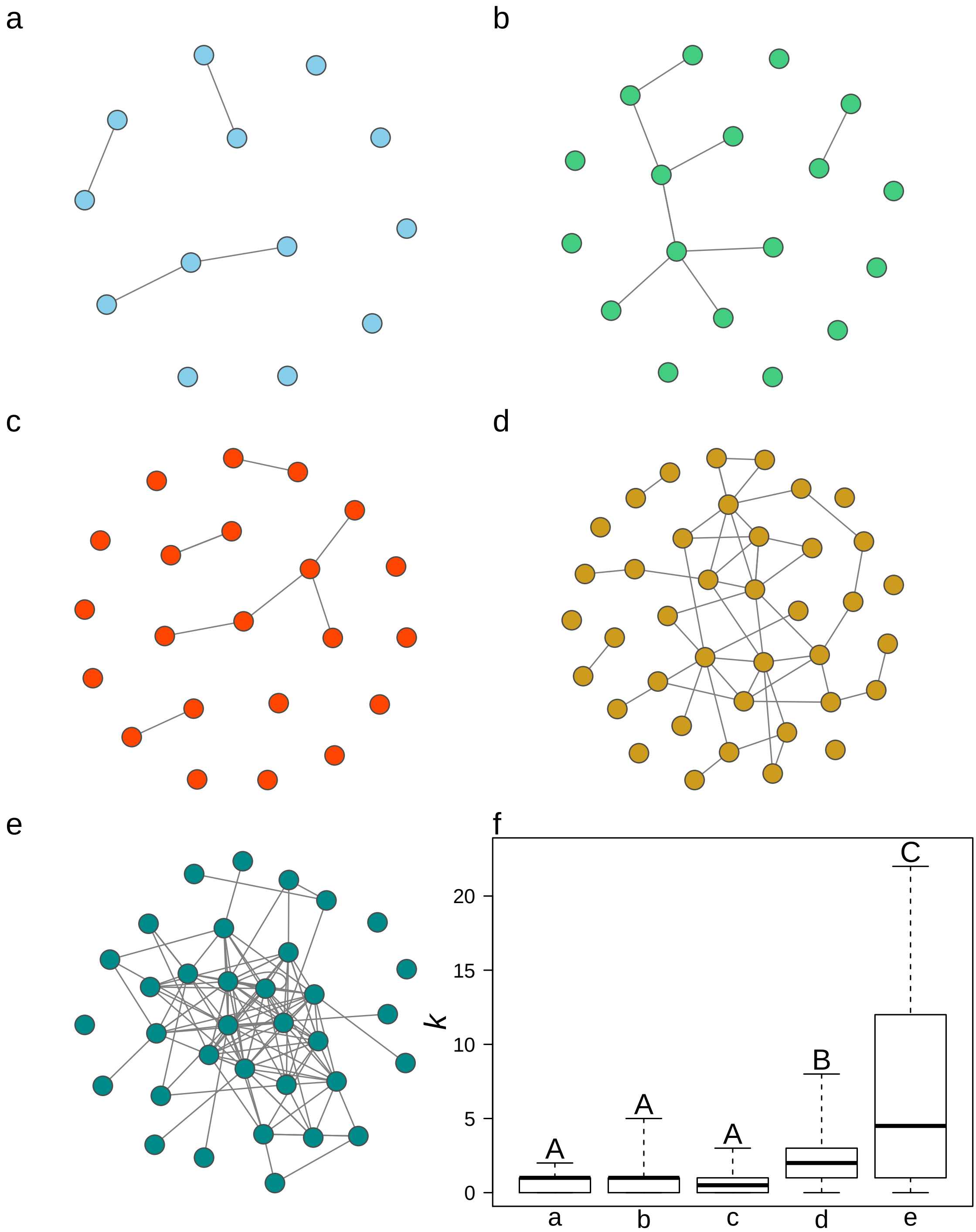


Figure 3.14. Networks based on importance classification. Each network represents the concepts classifed by workshop participants as; **a** unimportant , **b** some importance, **c** important, **d** very important and **e** highly important. Networks based on importance rating become increasingly connected with increasing importance because a few highly connected concepts are rated very important **d** or highly important **e**. **f** Concept degree (k) according to importance class **a–e**. The ends of whiskers represent k^{min} and k^{max}, ends of boxes are first and third quartiles and the black bar is the median value for each classification. Significant differences were calculated using the Wilcoxon rank sum test corrected for familywise error using the Hochberg method and are indicated by different capital letters (Wilcoxon rank sum test).

Notably, concepts related to farming such as stock density, farming knowledge, farming economics and productive farming were classed as very important. Whereas concepts related to (1) grouse moor management; managed burning, burning intensity, heather cover and game keeping, and (2) restoration; gully blocking, funds for restoration, revegetation, nurse crop and restoration process were all rated as highly important (Table 3.3).

3.6.3 Observations made during model development and validation

Concept definition

Perhaps the most challenging issue for participants during the validation process was the definition of concepts. However, because concepts were not pre–defined, definitions could not be created prior to the mapping process. For example 'Farming economics' was highlighted by one of the groups. In this case the definition could be easily resolved as the originator of the concept was present. Where the originator was not present, the issue was resolved by group discussion. Although concepts could not have been defined before the mapping process, participants were asked to add additional information to the concept description during mapping, but this was not always done.

One solution would be to ensure that each concept is clearly defined immediately after each map had been created (e.g. Christen et al. 2015). In the case of mapping workshops the process could have been done by the group after concepts had been listed and before they were connected, with the caveat that additional time would be required to identify concepts where the definition was unclear. That said, even if definitions are clear, some participants may still not understand the concept being discussed if it falls outside of their area of expertise. However, based on the discussions that took place during the validation workshop, the review of the precise definition of a concept by participants seems to be a valuable interaction between stakeholders with different backgrounds that could promote a greater understanding between groups.

Pre–defined concepts are not participants' mental models

During a later (non–mapping) workshop, a small group of participants requested a set of concepts in order to complete the connections to a single new concept (bracken) that they wished to add to the group FCM. A list of 63 concepts were provided because of time limitations (all concepts except those with less than five links). I observed the group discuss how bracken is connected to the other concepts in the list and eventually 39 new connections were added. However, the process revealed that in some cases, the group tried to determine connections between bracken and other concepts when no one in the group appeared to have the relevant expertise. This observation suggests that there is a risk to the credibility of the aggregated map if participants feel that they must find connections between concepts. When using pre–defined lists participants are not being challenged to develop maps that reflect on their own mental models as is the case when starting with a blank sheet of paper. This observation agrees with the original assertion by Axelrod (1976) that concepts should not be predefined. In fact, although I added these newly defined connections to the group FCM, they were challenged and removed during the validation process, highlighting the value of this second participatory stage in the FCM development process.

Linear causal relationships

Causal relationships in FCMs are monotonic and linear (Hobbs et al. 2002). During the validation workshop, a number of participants commented that this simple representation (an increase(decrease) of a concept causes an increase(decrease) in a connected concept) did not enable the specification of their interpretation of causality, which in some cases was non–linear. Comments such as "depends on the starting point", were added to the validation worksheets to note where this was the case.

Penn et al. (2013) also discuss this issue and propose that a set of non–linear functions could be chosen by participants to describe the causal relationship between two concepts. The authors suggest that choosing functional relationships could be carried out during a second workshop but recognise that it would be a challenge to make the process intuitive. Some studies have developed approaches to address this issue (e.g. Mendonça et al. 2013), but model development is made more complicated. Although an attractive option to modellers and some participants, it seems likely that mapping of causal relationships in this way would reduce the intuitiveness of the development process when used for a range of stakeholders and could negate the simplicity of the process used here. Depending on the aim of participation, other participatory approaches suitable for modelling complex systems, with feedback cycles, that can already incorporate non–linear relationships, such as agent–based modelling (e.g. Millington et al. 2011; Bommel et al. 2014; Wood et al. 2015) would seem to be more appropriate.

3.7 Discussion

The cognitive model co–developed by peatland stakeholders is a complex network of 126 concepts interconnected by 524 directed links, some of which form feedback loops. More than that though, the model is a representation of peatland knowledge from a wide variety of sources across the peatland community of the South Pennines. Knowledge networks (Axelrod 1976; Kosko 1988) in the form of cognitive and fuzzy cognitive models, provide a method to combine knowledge sources using an intuitive process that participants can grasp quickly.

3.7.1 Model co–development and validation

Attempts to develop an aggregate FCM in a single workshop with a group that contests land use has a number of drawbacks. Firstly, the final model that is likely to be less diverse (Kosko 1988), but the main risk with such an approach is that the group may make it difficult to develop a model at all if there are disagreements over causality.

This study addressed these limitations (refer also to Section 3.2.3) which resulted in a highly heterogeneous aggregate model of blanket peatlands that included 126 concepts and 524 pairs of causal interactions: the model was unusual in its complexity (see Hobbs et al. 2002, for a FCM with greater than 160 concepts). Many FCMs include less than 50 concepts (e.g. Dexter et al. 2012; Kontogianni et al. 2012; Penn et al. 2013; Christen et al. 2015); with some stating that FCMs with greater than 20 to 30 concepts are difficult to work with (Özesmi and Özesmi 2004). However, the approach used here captured a wide variety of concepts and relationships which enabled the FCM to be subset depending on the priorities of participants. In future, it would be possible to use different classifications to subset the model in order to address alternative questions. The development of such a heterogeneous model also provides an opportunity to use social network analysis tools (Newman 2010) to explore network structure and to investigate robustness and controllability (e.g. Albert et al. 2000; Liu et al. 2007; Morone and Makse 2015); both of which could be of interest when considering the impact of blanket peatland land use.

Eliciting individual mental models also enabled participants to define their own areas of expertise by answering different combinations of the set of questions asked at the mapping stage. Each participant's model overlapped with the expertise of others and created new combinations of concepts and interactions that produced an aggregate model that addressed all five questions asked. Although others have previously proposed that the reliability of an FCM improves with an increase in the number of participants (experts) (e.g. Kosko 1988), and that activities in the context of contested land use should not begin with mixed group consensus–seeking workshops (Dougill et al. 2006), this study has provided an example of how these these suggestions can be achieved in practice. Furthermore, evidence provided here has emphasised that individuals (or small groups) should not be provided with a full set of concepts that have been pre–defined (Axelrod 1976). This is important in the case of contested land use because mental models provide an insight into how system dynamics are perceived by participants and these perceptions influence land use decisions (Biggs et al. 2011). Restricting cognitive maps to pre–defined concepts may distort mental models and limit the value of the output.

3.7.2 Validation: opportunities to build consensus and identify issues that underpin conflict

A number of participants stated that the process of generating their own cognitive map was challenging and thought provoking because they were being asked to make their beliefs and assumptions explicit. This process can lead to new insights (Rouwette and Vennix 2006; Scott et al. 2013), but the opportunity to align mental models and gain consensus or identify the issues that underpin conflict through group work is lost when individual maps are acquired. However, the process described here overcomes this issue through group validation of concepts, links and weights. Ideally, validation would be achieved by a group comprised of all those who developed the individual models: although this was not possible in this case study, seven of the twelve validation participants did meet this criterion.

Importantly, instead of simply aggregating the weights of paired concepts with opposing causality, which could have resulted in the loss of the connection, these concepts were identified prior to validation, and were later discussed and agreed by the stakeholder group during the validation process. These differences in mental models were conflicts in how stakeholders perceived interactions to take place, and therefore the FCM can provide a useful mechanism to highlight important points for discussion that may underpin aspects of conflict (Christen et al. 2015). Such disagreements could be resolved (as was the case here) or be used for collaborative research projects. Whilst individuals' FCMs were checked and validated before aggregation, the group process showed the potential for using FCMs in cases of contested land use. The complexity of the aggregated FCM would have precluded validation of the model as a whole and so a clustering algorithm was used (Reichardt and Bornholdt 2006; Traag and Bruggeman 2009) to simplify the aggregate model. Arguably, the validation process 'weights' the opinions of those who provided individual FCMs by reviewing concepts and links strengths: in some FCMs this process is performed by algorithms (Groumpos 2014). Validating the FCM inside the participatory process also ensures the process remains transparent. Asking participants to validate the structure of the FCM by working together in mixed groups may also lead to greater ownership of the finalised model. Lack of transparency and ownership can become barriers to acceptance of model results and to stakeholder collaboration (Voinov and Bousquet 2010; Biggs et al. 2011; Elston et al. 2014).

3.7.3 Focus on contested land use

The selection of concepts according to their importance produced what was initially a surprising result. None of the concepts directly related to farming was seen as highly important for increasing or maintaining stores of peat. However, the choice to class activities related to restoration and grouse moor management as highly important reflects the current conflict between grouse moor managers and conservationists about the negative impact of managed burning on blanket peat stocks, and hydrology (Douglas et al. 2015; Holden et al. 2015). The exclusion of farming from the highly important classification was reviewed in a later workshop: the group discussed the fact that there has been a significant reduction in the high levels of sheep that resulted in vegetation change and erosion of blanket peatlands(Anderson and Radford 1993) and, as a result, farming was not now perceived to have the same negative impact on peat stocks. The inclusion of subsetting criteria shows how the approach used here has brought a focus on the key issues of land use that are relevant to participants.

3.7.4 Challenges and limitations

Participant engagement

On a number of occasions, stakeholders who had agreed to participate in the process did not attend workshops. This is understandable because workshops were organised some weeks in advance and circumstances change. However, from the initial recruitment process through all workshops it was challenging to maintain commitment to attend workshops from stakeholders, especially within land manager groups (farming and grouse moor management). For example, three people from the land manager community agreed to take part in the validation workshop but one failed to attend on the day, meaning that two of the twelve participants were from farming and grouse moor management communities: fortunately, both had contributed cognitive maps. In some cases a flexible approach could be adopted to address this issue: for example, with the help of the Derbyshire County advisor for the National Farmers Union (NFU), I was able to organise an evening mapping workshop and discussion with a group of farmers.

Some stakeholders may not wish to get involved in workshops because they lack interest (Prell et al. 2007) or simply do not perceive the process to be important (e.g Hossard et al. 2013). Others may not want to attend events where they may be required to work in groups or in a classroom style environment. Added to this, peatland stakeholders in the South Pennines have been involved in a number of participatory projects and some stakeholders may have become fatigued (Capistrano and Samper 2005; Reed 2008). For the cognitive mapping process, most of these issues can can be addressed with some thought about how the process could be adapted, such as evening or weekend workshops at a local meeting point (e.g. Knapp et al. 2011) or by carrying out discussions at participants' workplaces which may be a farm or a hilltop (e.g. Abel et al. 1998). Although the flexible approach used to obtain individual cognitive maps meant that alternative arrangements could be made, the validation process (which is where sharing, consensus, and alignment of mental models can take place) must be completed as a group.

Whilst it is clearly highly important to identify relevant participants, stakeholders will make a judgement about the value to them of engaging in projects. Some stakeholders declined to attend because their time in workshops would not be compensated. It is, therefore, tempting to think that providing financial compensation for people who attended workshops would have been the answer, but some people who initially asked to be paid still participated after payment was refused and, in general, participants did not claim travel and subsistence costs even though they were able to do so. This observation supports Reed et al. (2013c) who reported that reward can be non–financial and includes access to decision makers and concern for local livelihoods.

In hindsight, one way land manager attendance might have been improved further could be related to the design of the project which was developed and agreed with the Moors for the Future Partnership, but other stakeholder organisations were not involved in shaping the research process. Discussions with the NFU, Moorland Association and Heather Trust (for example), who represent farming and grouse moor communities, may have been able to help design a project of increased relevance and a more effective means of engagement. Building a working relationship with key members of the land management community can improve participation, as evidenced by my collaboration with the NFU County Advisor to gain the participation of a group of farmers.

There needs to be some assessment of the impact on this process of stakeholder attendance. My original aim was to obtain the cognitive maps from at least two people from the groups identified in Section 3.3 which was achieved (with the exception of one map from forestry), and two people from the land manager community attended all subsequent workshops. The workshop outputs are clearly dependant on the composition of the participant group, but it is, of course, impossible to say what the difference to the final model would have been if there been a different combination of attendees. Nevertheless, participants co–developed a complex network of social and ecological interactions that formed the basis for subsequent analysis and discussion about the impact of carbon storage, and discussions within workshops provided useful contextual information about the inclusion of stakeholder knowledge in decision–making processes.

There were disagreements during the validation process but they were resolved by reasoned discussion. Perhaps because participants were asked to describe how pairs of concepts interacted, and not to say what it meant for the dynamics of the whole system, discussions did not develop into arguments about the implications of land–use objectives, which is a model output. The composition of the group will have also affected how importance classifications were allocated to concepts, but this is to be expected and enables a group to determine the concepts (but not the interactions) that are important to them, and could be redefined in response to other questions. Interestingly, although importance classification was done without knowledge of how concepts were linked, five of the most connected concepts were included in the highly important subset with the remaining in the very important subset. These two classifications probably represent the core concepts and interactions of blanket peatlands as a social–ecological system from the stakeholders' perspective.

Model development and processing

Cognitive maps are the causal beliefs and assumptions of each participant and therefore different combinations of participants will be likely to result in different final models (Özesmi and Özesmi 2004; Penn et al. 2013). If the group FCM had been developed and validated by a another group of stakeholders, the final model structure could have been different to some degree. I attempted to improve FCM reliability by including a variety knowledge that overlaps (Kosko 1988), and therefore differences between different stakeholder groups would probably be seen in the non–overlapping parts of the combined model (refer to Figure 3.2).

Studies have called for frameworks that promote ownership of land use problems in social–ecological systems (Kates 2011; Lang et al. 2012) and some have proposed that including mental models from stakeholders may be one way to achieve this objective (Rouwette and Vennix 2006; Biggs et al. 2011; Penn et al. 2014). That one cognitive model may be different from a model developed by another group seems less important. Ultimately those who contribute to cognitive models may not be responsible for making land use decisions, but it seems reasonable to suggest that decision–making processes that incorporate multiple knowledge sources in bottom–up approaches, are likely to be more accepted and therefore could reduce contestation. Young et al. (2016a) found that stakeholders were more likely to accept the outputs from participatory processes if they felt their knowledge was accepted and recognised, and suggested that this may include when decisions are implemented. However, several participants said to me that they felt that their knowledge was not considered when land use decisions were made and, therefore, it was not valued; which suggests that they have a negative mindset to the decision–making process from the outset. However, whilst this is only one example, it represents an opportunity to incorporate additional knowledge into blanket peatland decision–making processes.

As with Kosko's original FCM, and many developed more recently (e.g. Penn et al. 2013; Reckien 2014; Christen et al. 2015), the model developed here did not take time into account. As a result, it is implied that each concept changes at the same rate with each model iteration (Carvalho 2013) which clearly is not the case with the interaction of peatland development and land use. Although a number of authors have developed methods to address the issue of time in FCMs (e.g. Hagiwara 1992; Neocleoua et al. 2011; Wise et al. 2012; Vogt et al. 2015), obtaining the data that enables time to be represented needs careful consideration. The rate of change could be represented by a delay in the activation of a relationship between a pair of concepts (e.g. Carvalho et al. 2008) or by reducing the value of the link weight (as suggested by Hobbs et al. 2002). I reasoned that asking participants to identify the rate of change at the mapping stage would overcomplicate the process. The mapping stage is already challenging and would be made more so if participants were also asked to think about the rate of change between two concepts. Validation is also an opportunity to add time–based data, but limitations on what could be achieved in the validation workshop precluded the collection of useful data at this stage too.

3.7.5 Summary

This chapter showed how a model of a complex system can be co–developed with stakeholders who have different backgrounds and land–use objectives. The use of mental models provided an opportunity for participants to reach consensus about how peatland concepts interact and may improve ownership of the modelled system. Only by working together can peatland stakeholders address issues of land use conflict: as part of a wider process the model presented here created a framework where stakeholders could work together to share knowledge and experience, and gain new insights about the network of blanket peatland interactions. In Chapter 4, I report on the process used to analyse the structure of the FCM with tools from network science. The outputs from the analysis were integrated into a workshop process, and a group of peatland stakeholders (the majority of whom had taken part in workshops described in this chapter) proposed how three land–use objectives could be achieved.

Chapter 4

The impact of land–use objectives on blanket peatland carbon storage

4.1 Chapter summary

The cognitive model developed in Chapter 3 represents blanket peatlands as a network of interactions: there are 126 components (concepts) that interact through 524 directed links, some of which form feedback loops. More than that though, the model is a representation of peatland knowledge from a wide variety of sources across the peatland community of the South Pennines. The aim of this chapter was to use this network of interactions to identify the impact of land–use objectives on carbon storage in blanket peatlands. In a workshop setting, I used the structure of the network as a guiding framework to ask a group of peatland stakeholders to propose how they would deliver three land–use objectives; (a) maintaining or increasing carbon storage, (b) improving the quality of water supplied, and (c) supporting local livelihoods. These land–use objectives can be considered to impact across global, national and local scales (although not exclusively).

The research questions addressed by this chapter are,

2. How can these factors [from the cognitive model] be used to evaluate the impact of land–use objectives on blanket peatland carbon storage?

3. How should blanket peatlands be managed to achieve the objectives of (a) maintaining or increasing carbon storage, (b) improving the quality of water supplied, and (c) supporting local livelihoods: what are the implications for current land uses?

This chapter includes four sections that describe the methods and results employed to answer the above research questions; (Section 4.3) an investigation of the structure of the blanket peatland cognitive model; (Section 4.4) the identification of driver concepts that can be used 'steer' the system to deliver land–use objectives; (Section 4.5) a peatland stakeholder workshop that identified how driver concepts should be managed to deliver the three land–use objectives described above; and (Section 4.6) a comparison of the impact on blanket peatland carbon storage of each of the three land–use objectives. Each section concludes with a summary of key findings and the chapter ends with a discussion of the implications for blanket peatland use. The scope of this chapter is outlined in Figure 4.1.

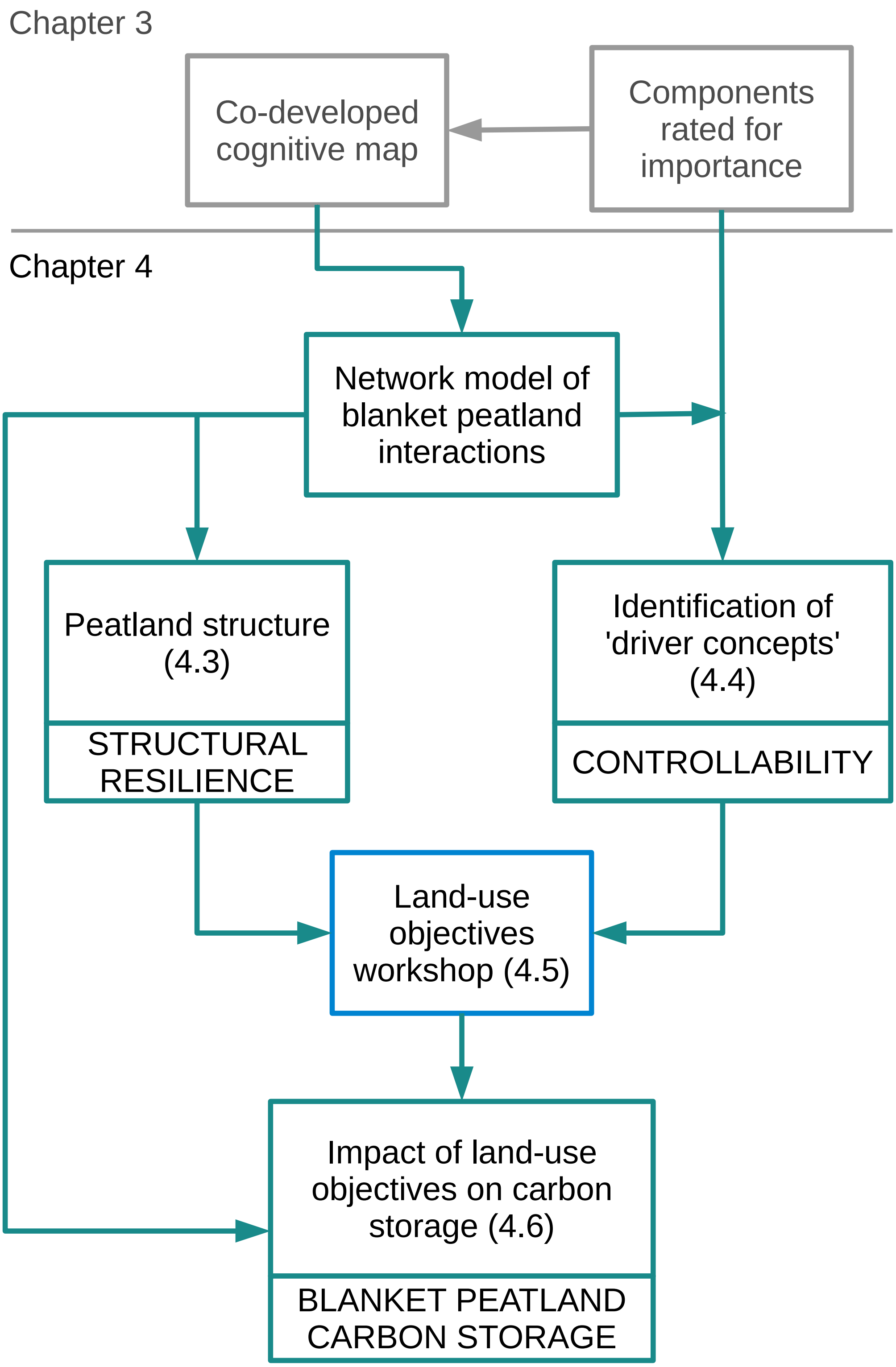


Figure 4.1. Chapter 4 analyses and outputs. Grey boxes represent the outputs from the co–developed cognitive model (Chapter 3), green boxes represent analyses and modelling and the blue box represents workshop activities. The numbers in parentheses refer to section numbers within this chapter.

4.2 Introduction

4.2.1 Background

The response of coupled social and ecological systems to land use is difficult to understand and predict, but land-use decisions in these complex systems often need to take into account multiple objectives, that range over local to global scales. Furthermore, there may be conflict between groups of stakeholders who support different objectives, making these decisions particularly challenging. Blanket peatlands in the UK are one such social–ecological system. Participatory modelling may be one way to address this challenge (Voinov and Bousquet 2010), by engaging stakeholders in order to share and advance knowledge about how the components of a system interact, and to determine how land–use decisions could affect key properties of the ecosystem (such as carbon storage) (Reynolds et al. 2009; Whitfield and Reed 2012). Several different approaches that involve stakeholders in the modelling process have been developed ranging from parameterising model functions (e.g. Chapman et al. 2009a; Elston et al. 2014), to participation in model building and scenario development (e.g. An 2012; Reed et al. 2013b; Bommel et al. 2014). Models of coupled systems should conceptualise ecosystems as complex systems that include feedback loops, and integrate local, scientific, and practitioner knowledge, of social and ecological processes and their interactions (Kates et al. 2001; Reynolds et al. 2009; Whitfield and Reed 2012; Liu et al. 2015a).

Suitable modelling approaches include agent based models (e.g. Wood et al. 2015), system dynamics (e.g. Corral-Quintana et al. 2016), and fuzzy cognitive mapping (e.g. Christen et al. 2015). Although fuzzy cognitive maps have been widely used to represent social–ecological systems (e.g. Hobbs et al. 2002; Özesmi and Özesmi 2004; Gray et al. 2015) they have been criticised because time is not represented in model iterations, the meaning of modelled relationships is often not clear (Carvalho 2013), and the function used to determine range of model outputs may not represent how each pair of concepts interacts in the real world (Knight et al. 2014; Vogt et al. 2015) (although parameterising each pairwise relationship may be unfeasible in large networks). However, fuzzy cognitive maps offer three important benefits that are useful when building models with stakeholders where land use is contested. (1) Stakeholders encode their knowledge in a model, using a simple and intuitive process, by creating diagrams of the interactions between ecosystem components (Chapter 3). (2) Importantly, the ability to obtain and later combine maps from individuals who disagree about land uses, enables a first order knowledge network (*sensu* Kosko 1988) to be developed, that can prevent stalemate in the model building process (c.f. Dougill et al. 2006). (3) Differences between interaction strength and direction of the same ecosystem components can be addressed as part of a validation process (c.f. Penn et al. 2013), that can also provide the basis for consensus building or to identify obstacles to collaboration (Chapter 3).

Fuzzy cognitive maps are directed weighted graphs, and although studies often describe a narrow set of network measures, such as degree centrality, the use of network structure as an integrated part of the participatory process has been limited to one other study (Penn et al. 2014). The authors analysed the structure of a fuzzy cognitive map to identify control nodes (the set of concepts that can be used to 'steer' the network to a desired state), and the controllability of these concepts was discussed by stakeholders. But these data could be used by stakeholders, in an additional step, to identify how multiple land–use objectives could be achieved. Furthermore, other structural characteristics of the fuzzy cognitive map may allow the impact of land–use objectives, on key ecosystem properties of interest, to be investigated. To my knowledge, studies of participatory modelling have not used the structural characteristics of a fuzzy cognitive map in this way.

Here, I investigate how stakeholders propose blanket peatlands should be managed to achieve the land–use objectives of (a) maintaining or increasing carbon storage, (b) improving the quality of water supplied, and (c) supporting local livelihoods. Firstly I tested the structural resilience of the network, and identified the concepts that could be used to steer the system (driver concepts) to achieve land–use objectives. I inferred that management of these driver concepts could be used to determine how to achieve land–use objectives. Secondly, I asked a group of blanket peatland stakeholders to propose how driver concepts should be changed to achieve the three land–use objectives. Finally, the causal relationships defined in the co–developed model of blanket peatlands were used to create a simple model of the dynamics of the system, which is often used as part of the fuzzy cognitive mapping process, to assess the relative impact on carbon storage of the changes proposed to achieve land–use objectives. I also discuss the likely impact of the changes to network resilience, and the implications of my results for current land–uses.

4.2.2 The structure of blanket peatland social–ecological interactions.

Complex systems comprise many interconnected components that interact across different scales, and one approach to represent these systems is in the form of networks (Barabási 2009). Social network analysis (Newman 2010) has provided many methods with which to analyse the structure of networks; from different classifications of the importance of individual components known as centrality (Freeman 1979), to algorithms that test the resilience of real–world biological and communication networks to the loss of certain components (e.g. Jeong et al. 2000; Morone and Makse 2015). Beyond the analysis of network structure, dynamical processes that act within or across these structures such as the collapse of ecological communities have also been investigated (e.g. Dakos and Bascompte 2014).

The configuration of connections in networks has significant implications for the for the growth, maintenance and persistence of the systems they represent (Boccaletti et al. 2006; Barabási 2012). The discovery that the components that make up some complex systems were not randomly connected, but were part of a structure formed by preferential attachment (Barabási and Albert 1999) has implications for our understanding of the spread of disease, management of traffic, and the ability of systems to withstand perturbation. Interestingly, networks across different domains and scales, such as microscopic organisms and the internet, can have similar underlying principles that govern their structural development (Jeong et al. 2000; Cohen et al. 2000). These networks are known as scale–free; structures that are formed when new concepts are preferentially attached to existing concepts rather than being randomly added (Barabási and Albert 1999). The defining feature of these systems is that the distribution of the number of connections per concept (or node) p_k follows a power law ($P_k \approx k^{-\alpha}$) (Jeong et al. 2000). In practice this means that most of the concepts within the network will have very few connections, and a few will have very many connections: these concepts are known as 'hubs' (Albert et al. 2000).

Not all real–world networks are scale–free. For example, Dunne et al. (2002) found that most food webs did not exhibit a scale–free structure (i.e. power law distributions), but concluded that their topologies sometimes displayed similar characteristics (i.e. non-random degree distributions, and short path lengths). One explanation provided for the difference between food webs and scale–free networks was the difference in the number of nodes (Dunne et al. 2002). Food webs were found to be typically composed of 10^1–10^2 nodes, as opposed to 10^2–10^7 nodes for scale–free systems. Clauset et al. (2009) tested the connection distributions of 24 real–world networks and found that 17 were consistent with power law distributions, but also that in many cases a log–normal distribution was plausible too. In addition, many distributions only follow a power law for a range of values above a certain value; known as small degree (k) saturation (Albert and Barabási 2002; Clauset et al. 2009). Finally, another defining feature of scale–free networks may be in the difference between the number of connections between the two concepts with the highest number of connections (i.e. k_{max} and k_{max-1}), which appears to be much larger in scale–free networks than in those with an exponential distribution (Ghoshal and Barabási 2011). Ghoshal and Barabási (2011) showed that in scale–free distributions, these differences could be of several orders of magnitude ($\Delta k/k \approx 5 \rightarrow 12$, where $\Delta k = k_{max} - k_{max-1} / k_{max-1}$): but in networks with an exponential degree distribution, $\Delta k/k \approx 0.01 \rightarrow 0.13$. As a comparison, $\Delta k/k = 0.24$ for the blanket peatland network.

To investigate the structure of the blanket peatland network, the cumulative distribution of the number of connections was calculated (Figure 4.2) using the igraph package (Csardi and Nepusz 2006) for R (R Core Team 2015). This relationship is known as the cumulative degree distribution

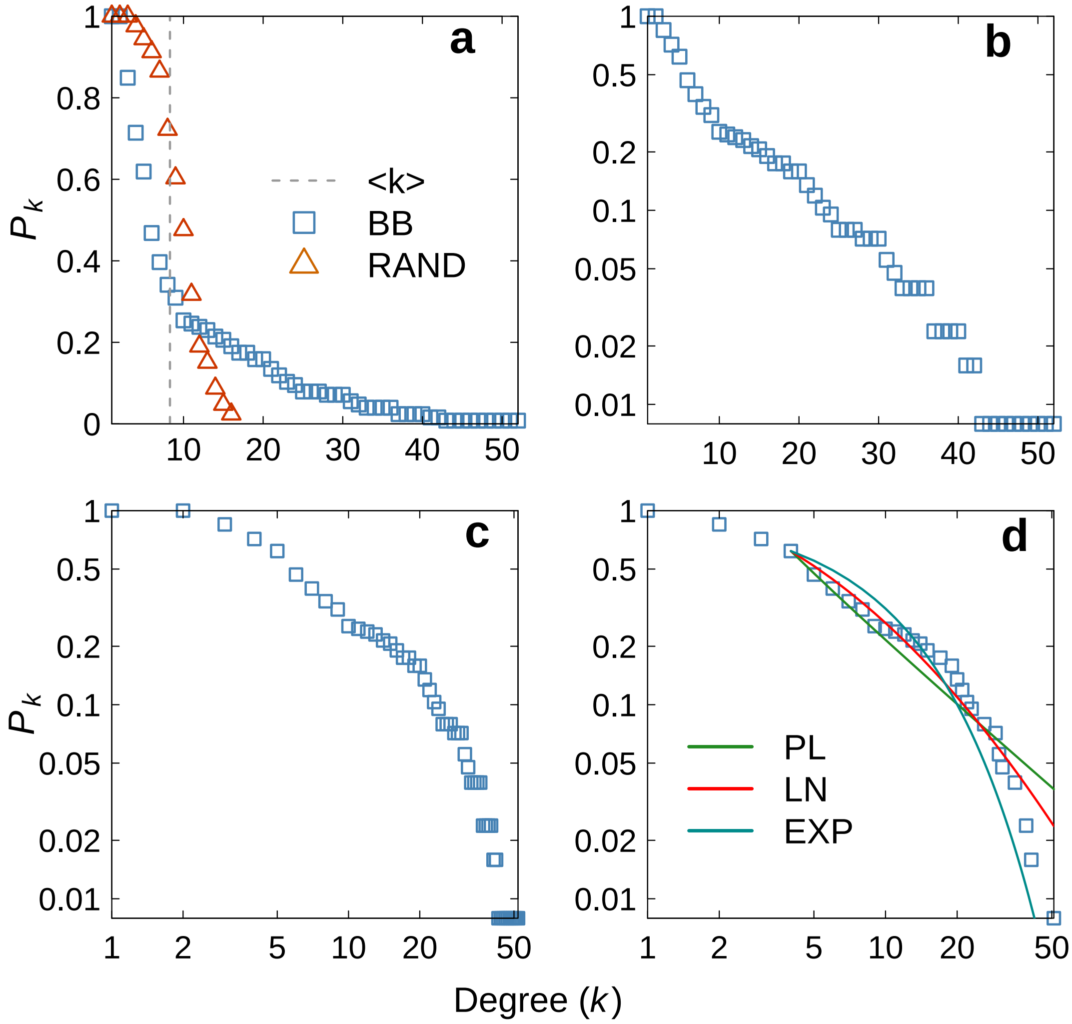


Figure 4.2. Distribution of the number of connections found in the Peak District blanket peatland network. The cumulative distribution of connections P_k is shown as a function of the number of connections k. **a** Blue squares denote the blanket peatland network by stakeholders (BB) and red triangles are the distribution of a network created randomly using the same number of concepts and connections as the BB network. The BB network has a few highly connected concepts ($k_{max} = 51$) and many concepts with a few connections: whereas $k_{max} = 15$ for the RAND model. Notably, in both networks, concepts have the same average number of connections $\langle k \rangle = 8.3$. **b** and **c** Log–normal and log–log plots of the BB network. **d** Comparison of fitted distributions; PL = power law, LN = log normal and EXP = exponential.

(P_k) and is the fraction of concepts in a network to have k connections (or the probability that a concept will have k connections, Newman 2010). In the case of the blanket peatland network, the probability that a concept has 51 connections (k_{max} in this case) was $\ll 1\%$. When plotted, the shape of the distribution is often used to visualise the structure of connections within complex networks and to infer whether the distribution is, in fact, scale–free: exponential and scale–free connection distributions exhibit a straight line on a log–normal and log–log plot respectively (Figure 4.2b and c). However, Newman (2010) warns that such plots should not be used to evaluate qualitatively the type of distribution.

Scale–free connection distributions are characterised by a degree exponent $k^{-\alpha}$ where $2 \leq \alpha \leq 3$. To calculate the degree exponent for the blanket peatland network, α was estimated ($\hat{\alpha}$, the $\hat{}$ symbol denotes α is an estimate) according to

$$\hat{\alpha} = 1 + N \left[\sum_{i=1}^{n} \ln \frac{k_i}{\hat{k}_{min} - \frac{1}{2}} \right]^{-1}, \tag{4.1}$$

where $\hat{k}_{min}$ is the estimated upper bound for values of k that do not follow a power law and the error on $\hat{\alpha}$ is given by

$$\sigma = \frac{1 - \hat{\alpha}}{\sqrt{N}}. \tag{4.2}$$

In this instance, $\hat{k}_{min} = 4$ calculated according to the method of (Clauset et al. 2009) using the R package developed by Gillespie (2015). N is the number of concepts with $k \geq k_{min}$: setting $\hat{k}_{min} = 4$ produced a sample size of $N = 78$ and applying Equations 4.1 and 4.2 (Newman 2010) to the blanket peatland data gave $\hat{\alpha} = 2.05 \pm 0.12$. To test if the distribution followed a power law, the Kolmogorov–Smirnov test statistic was calculated for 2,500 bootstrapped samples and a *p* value computed (Gillespie 2015): $p = 0.676$ and therefore, according to Clauset et al. (2009, page 18), a power law distribution should be ruled out ($p > 0.1$). As suggested by Clauset et al. (2009), direct comparisons were then made to discrete log–normal and exponential distributions to ascertain if either were a better fit (Figure 4.2 d). Again this approach was implemented by Gillespie (2015), where the sign of Vuong's test statistic *R* (used for model selection), indicates which distribution is favoured. A value of $p < 0.1$ indicates that one of the distributions is more representative of the true distribution Clauset et al. (2009). In all cases it was not possible to determine the distribution that best fitted the blanket peatland data; power law and log–normal $p = 0.168, R = -1.378$, power law and exponential $p = 0.711, R = 0.371$, log–normal and exponential $p = 0.214, R = 1.244$.

The difficulties of identifying the likely distribution are magnified with small networks (Dunne et al. 2002) such as the blanket peatland model, and as a result none of the tested distributions could be confirmed. Consistent with the topologies of some food webs, the blanket peatland network was found to have a non-random degree distribution. It is apparent from this initial analysis that the network structure features a few highly connected 'hubs', one of which is *Sphagnum* cover, connected to $\approx 36\%$ of the network. These hubs may have important implications for blanket peatland ecosystem function, and hence the decisions made to achieve land–use objectives. Although the distribution of the peatland model could not be identified, the possible impact of concepts on the structure of the network was be simulated by removing them and calculating the effect on network structure; a process carried out in Section 4.3.

4.3 Blanket peatland network: structural resilience

4.3.1 Background

Peatlands are important stores of carbon and interact with the global climate. They provide freshwater for drinking, mitigation of flooding, and livelihoods in rural communities (Holden 2005b; Maltby 2010; Charman et al. 2013). The future resilience of peatlands is key to the continued provision of these ecosystem services, but decisions made by land users can negatively impact peatland hydrological and ecological processes, and result in degradation of both ecosystem structure and function (e.g. Moore et al. 2013; Holden et al. 2015; Turetsky et al. 2015). Because the resilience of a network to perturbation has been shown to be related to its underlying structure (Albert et al. 2000; Cohen et al. 2000, 2001), an understanding of how blanket peatland resilience is affected by the loss of hydrological and ecological concepts could be taken into account by stakeholders when making assessments of the effect of land–use decisions on carbon storage.

The structure of the co–developed blanket peatland network was maintained by a framework of interactions between concepts, and therefore this framework was used to test the resilience of blanket peatlands to the loss of concepts (e.g. Jeong et al. 2000; Cohen et al. 2001; Holme et al. 2002; Morone and Makse 2015). Maintenance of the framework of interactions, is dependant on set of concepts that are structurally important (Morone and Makse 2015). These concepts influence the transmission of data, spread of disease, or the feedback mechanisms that mediate a system's response to perturbation. One approach to determine the influence, or importance of a concept in a network is by its centrality. Freeman (1979) formalised three approaches to centrality based on, (1) the number of connections (degree centrality), (2) the frequency a concept is found between any other pair of concepts (betweenness centrality), and (3) the distance of a concept from all others in the network (closeness centrality). Degree centrality is simply the sum of the number of connections in to and out from a concept; in–degree is the sum of connections in to, and out–degree is the number of connections out from, a concept. Betweenness centrality measures the influence a concept has on other concepts as a result of the control of flow within the network and defines *"the extent to which a vertex* [concept] *lies on paths between other vertices* [concepts]*"* (Newman 2010). Closeness centrality is a measure of the mean distance between a concept and all other concepts (Freeman 1979).

Since Freeman (1979), alternative approaches that define concept importance have been developed. Eigenvector centrality determines concept rank by taking into account the centrality of the concepts to which it is connected: higher scores are awarded to concepts that are connected to important concepts (Newman 2010). However, complications can arise when using eigenvector centrality with directed graphs (such as the blanket bog model) because the matrix of a directed graph is asymmetric (Newman 2010). For this reason eigenvector centrality was not used in this study. PageRank centrality (Brin and Page 1998) was developed in order to rank websites for Google's search engine. PageRank further refines the allocation of the benefit of being connected to important concepts: an important concept that is connected to many other concepts shares its centrality measure amongst them all. For example, '*Sphagnum* cover' has 20 outgoing connections and therefore according to PageRank, each connected concept receives $1/20$ of its centrality score. Measures of centrality have been used to develop an understanding of the set of concepts that influence network resilience (e.g. Cohen et al. 2000, 2001; Morone and Makse 2015). And to test how well the network is able to maintain its framework of interconnections under random failure and 'attack', concepts and their connections can be removed from the structure in a process known as percolation (Newman 2010).

Network percolation is the process of removing concepts or connections from a network to understand the effect on system structure (Figure 4.3). There are two types of percolation; site percolation is the removal of concepts (nodes) and bond percolation the removal of connections. A network is said to percolate when a giant component, a cluster that grows in proportion to the number of concepts within the network, emerges (Newman 2010). Giant components are said to scale with an increase in network size but as real networks are of fixed size, the largest cluster is used to fulfil this role (Newman 2010, page 606). During percolation, the size of the largest cluster is used to understand how a system responds to both random and targeted perturbation of concepts or connections (e.g. Albert et al. 2000).

Examples of percolation problems in real–world systems include the internet (Cohen et al. 2000), traffic systems (Li et al. 2014), metabolic networks (Jeong et al. 2000) and the spread of epidemics (Newman and Ferrario 2013; Liu et al. 2015b). Albert et al. (2000) demonstrated that communication networks such as the internet were highly robust to the random removal of concepts but were extremely vulnerable to the targeted removal of concepts with the highest number of connections (degree centrality). As Newman (2010) points out, the response of networks with highly connected hubs to perturbation is quite intuitive: removing a concept at random is likely to result in the loss of one with few connections, because they occur frequently. Removing that concept will not greatly affect the integrity of the framework of connections of the whole network. This effect can be demonstrated by comparing the resilience of the blanket peatland network to that of a network generated with randomly connected concepts. The critical fraction of concepts (f_c) that would need to be randomly lost from the framework of blanket peatland interconnections to cause a complete collapse can be estimated from Cohen et al. (2000):

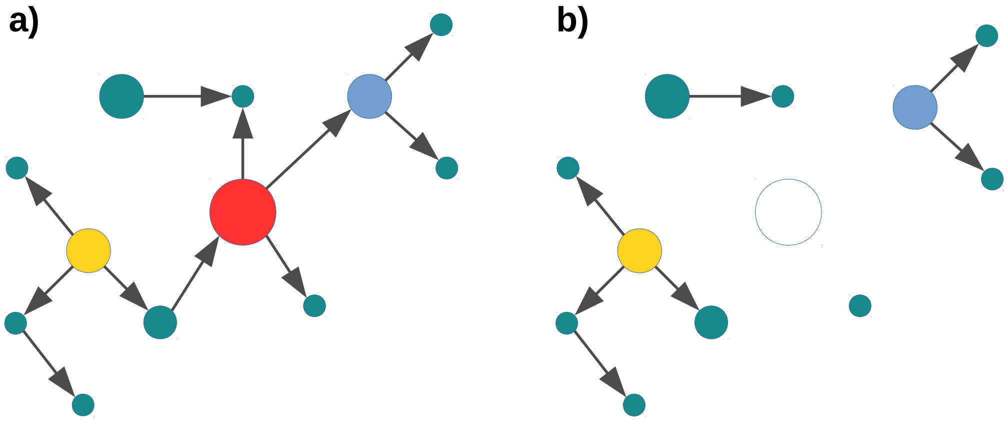


Figure 4.3. Network fragmentation by removing the concept with the most connections. **a)** In an implementation of a high degree (HD) strategy, the most connected concept (orange) in this network $k_{max} = 5$ is targeted for removal. The size of the largest cluster $S = 12$. **b)** Following removal of the orange concept $S = 5$ (centred on the yellow concept) and the network has been broken into four fragments, one of which is a disconnected concept. In network **a** both the yellow and blue concepts have $k = 3$: using a HD strategy one of these concepts would be randomly chosen for removal. However, with a high degree adaptive (HDA) strategy the number of connections for all remaining concepts is recalculated (yellow $k = 3$, blue $k = 2$) and the yellow node targeted for removal. Diagram adapted from Kovács and Barabási (2015).

$$f_c^{BB} = 1 - \frac{1}{\boldsymbol{K}_0 - 1}, \tag{4.3}$$

where $\boldsymbol{K}_0 = \frac{\langle k^2 \rangle}{\langle k \rangle}$ is from the original distribution of connections, which is compared to a random network with the same average number of connections (Barabási 2015, page 11):

$$f_c^{RAND} = 1 - \frac{1}{\langle k \rangle}. \tag{4.4}$$

The critical fraction that would need to be lost from the blanket peatland network $f_c^{BB} = 0.946$, and for the random network $f_c^{RAND} = 0.880$. Therefore, the blanket peatland network is more resilient to the random loss of components than a random network based on the same average number of connections, which is consistent with the findings of Jeong et al. (2000) and Newman (2010) in studies of complex networks. In fact, f_c^{BB} indicates that the blanket peatland network would only completely break apart after the random loss of $\approx 95\%$ of concepts. However, it is the impact of the loss of key hydrological and ecological concepts that are important for the future resilience of blanket peatlands that is of interest. During a targeted attack, the loss of a highly connected concept (hub) is likely to have a significant effect on the structure of the network because such concepts dominate the framework of interactions (Figure 4.3).

4.3.2 *Methods*

Strategy for determining the loss of concepts

To determine the effect of loss concepts on the fragmentation of the blanket peatland network, six strategies were implemented; (S1) random loss, (S2) decreasing stakeholder importance rating, (S3) decreasing number of connections (degree centrality), (S4) decreasing betweenness, (S5) decreasing closeness, and (S6) decreasing PageRank. Strategies (S3)–(S6) prioritised the loss of concepts from the network based on their calculated rank of influence, and are known as 'attack strategies' (Morone and Makse 2015). Network centrality measures were calculated using igraph (Csardi and Nepusz 2006). Each attack strategy adopted here was used to target network concepts according to decreasing influence and was later compared to stakeholder classification (S2).

Attack strategies that are based on iteratively removing concepts in order of the number of connections (i.e. highest degree centrality first, HD) (e.g. Newman 2010), use a fixed ranking of the number of connections based on the complete network state. However, the distribution of connections changes with the loss of each concept (Figure 4.3), and it is this new structure that is the basis of response to perturbation. An alternative to this approach (described by Morone and Makse 2015, supplementary information page 3) is to recalculate the ranking based on the new network structure that is created after the removal of a concept: this is known as high degree adaptive (HDA). A comparison was made between the HD and HDA approaches which demonstrated that the blanket peatland network did, in fact, respond differently when the concepts' degree was recalculated (Figure 4.4), and therefore HDA was chosen for the degree–based strategy (S3).

Percolation algorithm

Two approaches to site percolation were used. For strategies S1, S2 and S4–S6, a general algorithm was developed that followed the process proposed by Newman (2010, pages 617–620). Concepts were either randomly added (S1) or added in increasing order of influence (S2, S4–S6) to an empty network and the size of the largest cluster (S) calculated with the addition of each concept (r) until the complete network was recreated and a value for S obtained for the addition of each concept. For HDA, in order to be able to dynamically recalculate the concepts' degree, the percolation process was started with a complete network. The degree of each concept was recalculated after each removal, and the concept with the highest degree that remained in the network was identified for the next attack iteration. The value of S was calculated for each r after the removal of each concept. Both random and targeted removal of concepts requires that any percolation process is repeated many times to account for the probability of selecting a particular concept at random. Although

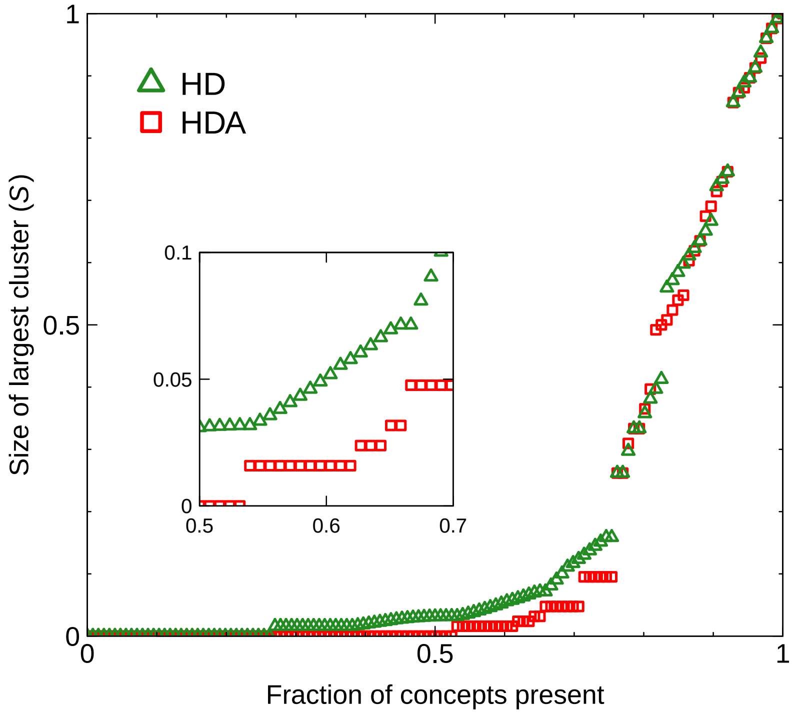


Figure 4.4. Comparison between highest degree (HD) and highest degree adaptive (HDA) strategies for network percolation. Concepts were removed from the network using both HD and HDA strategies. HDA results in a more rapid fragmentation of the network.

targeted removal of nodes is not completely random, because it is based on a particular scheme (e.g. removal by decreasing order of degree k), many concepts may possess the same number of connections and so there needs to be random selection between these concepts. The algorithm for each percolation strategy was repeated 1,000 times, and the mean size of the largest cluster calculated.

Percolation describes the probability that a concept is present in the network (i.e. it is occupied), which is known as occupation probability (ϕ): $\phi = 1$ denotes all concepts are present whereas $\phi = 0$ denotes that all concepts are absent (Newman 2010). A mean value for Sr was calculated over all iterations. Next, the mean size of the largest cluster S was obtained for the range of occupation probabilities $[0, 1]$ to give $S(\phi)$ from

$$S(\phi) = \sum_{r=0}^{n} \binom{n}{r} \phi^r (1-\phi)^{n-r} Sr, \tag{4.5}$$

where the probability that each concept is present (p_r) is given by $p_r = \binom{n}{r} \phi^r (1-\phi)^{n-r}$ (Newman 2010, page 617).

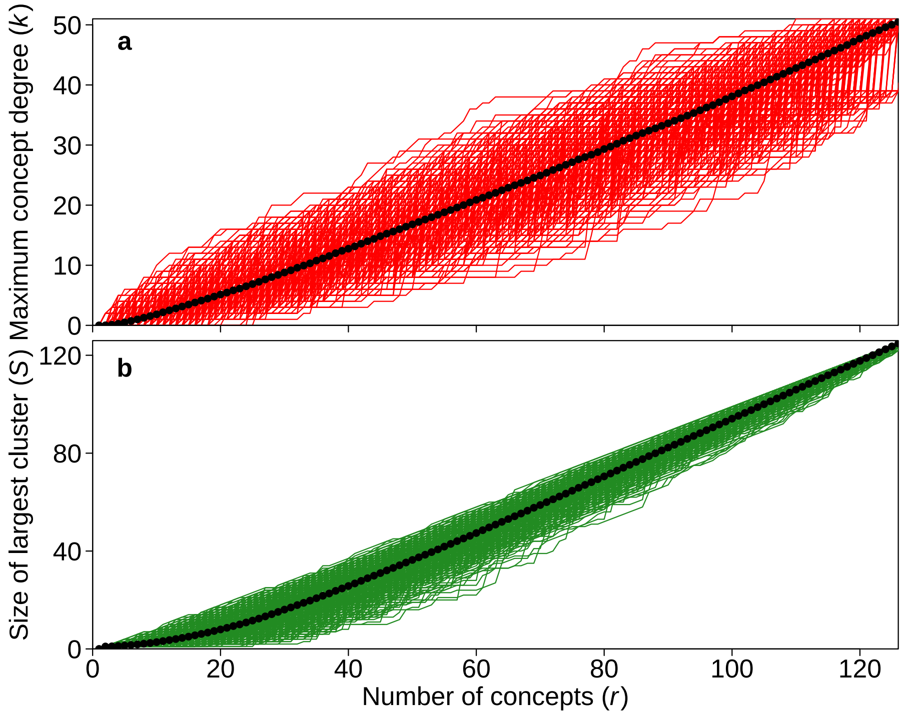


Figure 4.5. Random generation of largest cluster size with the addition of each concept to the network. Strategies S1, S2, S4–S6. a) The maximum number of connections for a single concept (r) for as each new concept is randomly added to the network. The black dots represent the concept with the highest number of connections to be randomly removed from the network. **b)** Size of the largest cluster (S) as each new concept is added to the network. The black dots are the mean values of Sr for all r (Newman 2010).

Further network configuration

The blanket peatland model represents the interface of interactions between and within two coupled systems (i.e. social and ecological). Some of the interactions are mutualistic (i.e. beneficial to both the interacting concepts) and some are antagonistic (i.e. beneficial to one concept) (Montoya et al. 2006). These are the interactions that cause an increase in the attached concept (mutualistic), or a decrease in the attached concept (antagonistic). To investigate the effects of percolation, the network was reconfigured by removing those concepts (including bare peat) that had been included by stakeholders and had been previously identified as drivers of antagonistic interactions in the conceptual model of anthropogenic impacts on carbon accumulation (Chapter 2, Figure 2.10) and Table 4.1. In addition, concepts that were directly associated with the primary antagonistic drivers were also removed: for example, bare peat, cattle grazing, winter grazing and supplementary feeding. A total of 21 concepts were removed. This was done because the loss of concepts through percolation is 'blind' to the type of concept, and it was counterintuitive to think that the removal of, for example, wildfire from the network would result in a reduction of peatland resilience; quite the reverse in fact.

Table 4.1. Drivers of antagonistic interactions

Primary impact drivers
Afforestation
Grazing
Burning
Draining
Pollution
Bare peat
Erosion
Wildfire

4.3.3 Results and discussion

The random loss of concepts fragmented the network gradually as predicted, with high levels of connectivity remaining even when most of the concepts had been removed (Figure 4.6a, RND). Although it is unlikely that these high levels of degradation would leave ecosystem function undamaged (*sensu* Newman 2010, page 608), the blanket peatland network was highly resilient to random losses (Figure 4.6a, RND), which agrees with the findings of studies that used connectivity data from other real–world complex networks (e.g. Cohen et al. 2000). The results show that when concepts were lost in descending order of influence, according to their ranking by one of the centrality measures, the network fragmented more quickly (Figure 4.6a, HDA, CC and b, BC, PR). Again this effect has been demonstrated in a number of other real–world networks (e.g. Albert et al. 2000; Cohen et al. 2001; Morone and Makse 2015), and the degree distribution data from other studies suggests that they too will behave in a similar way (e.g. Clauset et al. 2009).

The most rapid fragmentation occurred when concepts were removed according to the number of connections (S3). The first four concepts to be removed were *Sphagnum* spp. cover, water–table depth, water–storage and heather cover. These concepts clearly play an important role in maintaining the integrity of the framework of interconnections. The network had almost completely collapsed after the loss of $\approx 30\%$ of the concepts where the largest cluster (S) is around 10% of the size of the original network (i.e. highly disconnected and, very likely, not functional). The network was completely disconnected following the loss of $\approx 50\%$ of concepts.

Alternative measures of influence (closeness, betweenness and PageRank centralities) all demonstrated relatively quick initial collapses in connectivity. Further investigation, using Spearman's *rho* (R Core Team 2015), showed these alternative influencers were, to some extent, correlated with the number of connections of a concept: the higher the number of connections, the higher concept rank according to the alternative centrality measure. The correlations were; closeness centrality *rho* = 0.833, $p < 0.001$; betweenness centrality *rho* = 0.746, $p < 0.001$; and PageRank centrality *rho* = 0.588, $p < 0.001$. However, in all cases small clusters remained fully connected until almost all concepts had been removed. This was because some concepts had different numbers of interconnections but the same, or similar, values for the measure of centrality being used to fragment the network. The effect was particularly noticeable in the case of closeness centrality (Figure 4.6a, CC) where the rate of collapse slowed when between $\approx 70\% - 40\%$ of concepts were present. Studies have shown that the collapse of large scale–free networks can occur when $< 10\%$ of concepts have been removed (e.g. Holme et al. 2002; Newman 2010, p. 624). The difference between the rate of collapse in the blanket peatland network and the networks of other complex systems could be interpreted to mean that blanket peatlands show a higher degree or resilience, but the result may also be a function of network size (Dunne et al. 2002).

A number of alternative percolation attack strategies could have been selected (a review of the effectiveness of alternatives is given in the supplementary information of Morone and Makse 2015). HDA, betweenness, closeness and PageRank are all common choices, but Morone and Makse (2015) have proposed a new algorithm for attacking large networks (2×10^5 concepts with $k_{max} = 10^3$) based on the collective influence effects (CI) of concepts. The authors identified the 'optimal influencers' that maintain network connectivity and then removed them to test the effect on

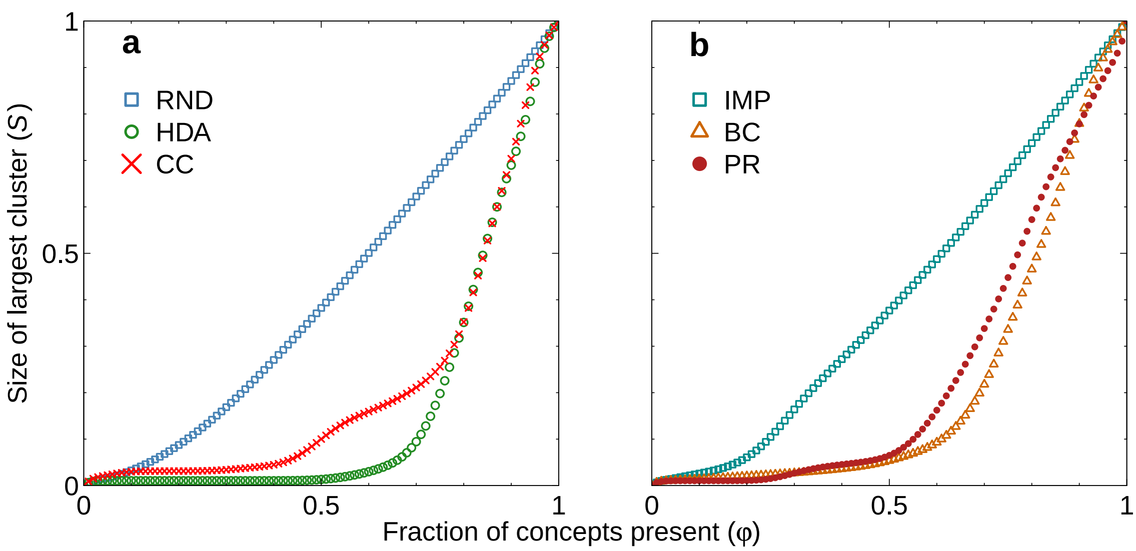


Figure 4.6. Fragmentation of the blanket bog network. a The network is highly resilient to random removal of concepts because a few highly connected hubs prevent fragmentation. RND = random. The loss of highly connected concepts (HDA) results rapid fragmentation of the network. CC = closeness centrality. **b** Comparison of alternative classifiers. IMP = stakeholder importance rating, BC = betweenness centrality and PR = PageRank centrality. The size of the largest cluster (S) is shown as a function of the probability that a concept is present (ϕ). Each simulation was repeated 1,000 times.

the largest cluster. They found CI to be more effective than existing approaches for large networks. However, of the reviewed alternatives HDA performed reasonably well, was more effective than the alternatives (consistent with the results shown here) and, in comparison to CI, is relatively simple to implement.

Lastly, when concepts were lost in the order of importance, as classified by stakeholders , the network resisted fragmentation almost to the same degree as when concepts were removed randomly (Figure 4.6b, IMP). This was because stakeholder classification was based on the perceived importance of a concept, and not the number of connections. A number of concepts that were rated as 'highly important' had <5 connections (e.g. climate change and trapping eroded peat), whilst other concepts with higher numbers of connections (>15), were perceived by stakeholders as less important (e.g. temperature and water quality).

4.3.4 Summary of key findings from this section

There were two key findings from this section which both revolve around the importance of a few highly connected concepts. Firstly, an analysis of the structure of the blanket peatland cognitive model found that although the specific distribution of connections could not be confirmed, there are a few highly connected concepts (hubs) that maintain the framework of interconnections within the blanket peatland network. This means that blanket peatlands are resilient to the loss of concepts that are not highly connected (i.e. most concepts) but are vulnerable to the loss or damage to hubs (such as *Sphagnum* spp., the most highly connected concept) which can result in the collapse of the network (Albert et al. 2000; Jeong et al. 2000) and, by inference, ecosystem function.

Secondly, a comparison of stakeholder importance classification, and network–based measures of importance (centrality) highlighted the benefit of including additional knowledge–based information about concepts. The mechanistic analysis of the network structure identified that a few concepts were key to maintaining the framework of interactions in blanket peatlands (comparable to other networks of complex systems). However, stakeholders highlighted that there were other 'highly important' concepts in the network with relatively few connections. One possible explanation for this result is that stakeholders have also identified highly important small degree concepts that may play a different role in blanket peatland carbon storage, that was not designed to be taken into account by the percolation algorithms, but are related to the management or control of the network (e.g. Liu et al. 2011), which is explored in Section 4.4.

4.4 Blanket peatland network: controllability

4.4.1 Background

The role of 'hubs', such as *Sphagnum* cover, water–table depth and heather cover, was found to be central to maintaining the integrity of the blanket peatland network (Section 4.3). My results imply that land users should attempt to protect the function of mutualistic hubs to promote future blanket peatland resilience and carbon storage. However, there are two challenges to this goal: (1) the concepts within the network (both social and ecological) that need to be managed or controlled, if this goal is to be achieved, have not yet been identified. It is not known if it is the hubs themselves that should be managed directly or if some other combination of concepts could provide the conditions that promote hub integrity. (2) Furthermore, identification is only a first step: even if these concepts are known, land users would subsequently need to propose, and agree, the actions that should be taken to influence the direction of the network in order to achieve a particular set of land–use objectives. The aim of this section is to address the first part of this challenge and identify the concepts that should be managed to steer future peatland development. The second part the challenge is explored in Section 4.5.

Although control theory, especially in relation to engineered systems, has been in existence for some time (e.g. Kelly et al. 1998), a number of recent studies have focussed on developing tools that can identify a subset of concepts (nodes) to provide control of a complex system (Liu et al. 2011; Jia et al. 2013; Gao et al. 2014). These concepts are known as driver or control nodes (hereafter driver concepts). Using an approach that can identify the subset of concepts, that provide control, is attractive because it would be impractical (if not impossible), and prohibitively costly to try to control all of the concepts within the blanket peatland network. Once the relevant concepts have been identified, the process also provides an opportunity for the stakeholder group to continue to work together, share mental models, experiences, and knowledge, and use the set of driver concepts to decide the actions that should be taken to achieve land–use objectives.

Methods have been developed to identify the number of driver concepts required to control a complex system (e.g. Liu et al. 2011) if it is 'structurally controllable' (Lin 1974). The approach proposed by Liu et al. (2011) uses the maximum matching of the links (interactions) in a network to determine the number of driver nodes. The maximum matching determines the set of interactions that do not share the same start or end concepts (these are the matched interactions). Concepts are matched when they are connected to the network by a matched interaction (Liu et al. 2011). Driver concepts are those that are connected by unmatched interactions. Network control is based on control of these unmatched (driver) concepts (Figure 4.7) Liu et al. (2011). A matching is classed

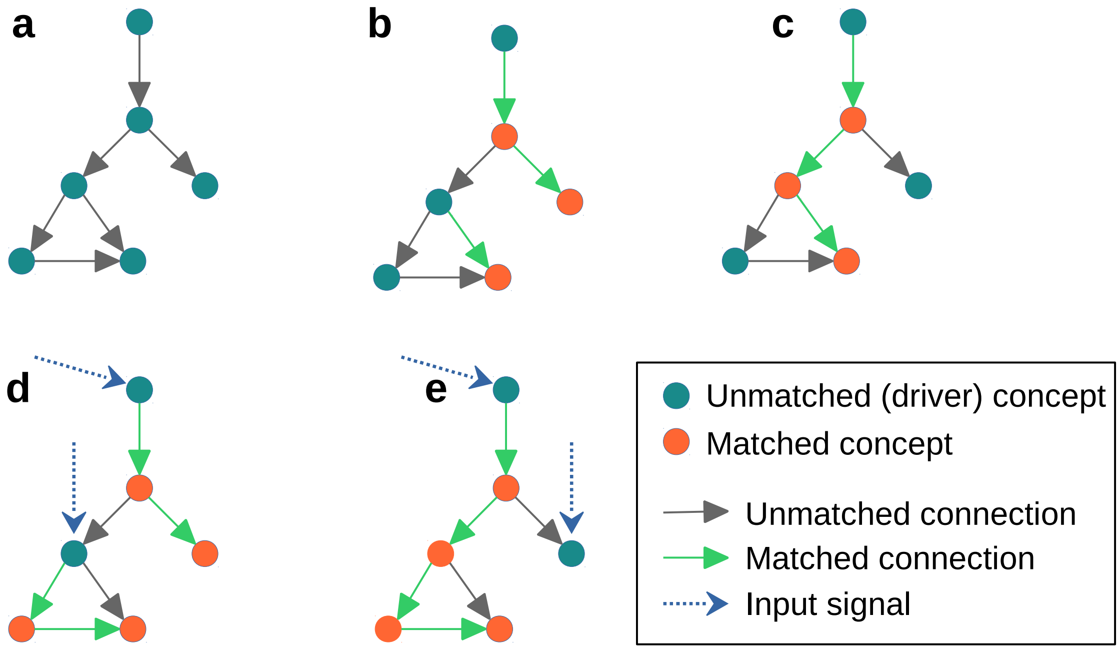


Figure 4.7. Schematic of the process used to identify driver concepts. Matched connections do no share the same start *or* the same end concepts and matched concepts are pointed at by matched connections (Liu et al. 2011). Driver concepts are pointed at by unmatched connections: they are unmatched concepts. **a** Six concept directed network. Four matchings are shown (**b–e**). **b** and **c** have three matched connections and three unmatched concepts. **d** and **e** are two maximum matchings with four matched connections and two unmatched (driver) concepts ($N_D = 2$). The dashed blue arrows represent the inputs needed (management actions) to steer the network. Diagram based on Liu et al. (2011).

as the maximum matching when there are no other matchings of greater size (cardinality) (Tassa 2012); although there may be several maximum matchings of the same size. The method developed by Liu et al. (2011) offers several advantages over other approaches that can be used, to identify the set of concepts required, for system control. (1) It is computationally more efficient when used with large networks (Lombardi and Hörnquist 2007; Pu et al. 2012); (2) there is no requirement for link weights to be known (although they are in the case of the blanket peatland network); and (3) knowledge of the non–linear dynamics of each interaction are not needed. In complex networks such as the one of blanket peatlands, the non–linear dynamics of each interaction are probably not known and, if they were, including them all in an algorithm would be unfeasible, (Liu et al. 2011). Fortunately, for the identification of the number of driver concepts (N_D), Liu et al. (2011) found that in most systems linear dynamics are likely to be sufficient (Liu et al. 2011, pages 6–7 supplementary information).

Liu et al. (2011) also proposed that as driver concept density n_D increases (i.e. $n_D \rightarrow 1$), a network becomes more difficult to control, until all concepts need to be controlled (an impossible task in large heterogeneous networks). A wide range of values of n_D, has been found for a number of networks, from 0.965 ($N = 4,441$, $\langle k \rangle = 5.80$) for a yeast network, to 0.013 for some organisational networks. $n_D = 0.511$ for the Ythan Estuary food web ($N = 135$ and $\langle k \rangle = 8.90$, Dunne et al. 2002) which was a similar size to the blanket peatland network ($N = 126$, $\langle k \rangle = 8.32$). Although the method of Liu et al. (2011) identifies the number and density of the minimum set of concepts required for network control, it does not identify the specific concepts that comprise this set. Clearly it is important to identify the driver concepts in order to determine if or how they could be managed. To address this problem, Penn et al. (2014) developed a method to discover the identity of the set of driver concepts (configuration) for inclusion in participatory modelling.

4.4.2 Methods

Driver concepts were identified for the complete blanket peatland network, and the highly important subset using the algorithm[1] reported by Penn et al. (2014). Because the number of driver concepts for the complete blanket peatland network was expected to be in excess of 30 (based on the values of n_D found by Liu et al. 2011), it would have been impractical to ask a stakeholder group to work with a large number of concepts in a workshop setting. Additionally, depending on the number of concepts identified it may be impractical to actually implement control for a large number of concepts (Gao et al. 2014). The highly important subset was selected because these concepts had been shown to form a smaller highly connected group, of 34 concepts, within the overall network of 126 concepts, which it may be both feasible and adequate to control (Gao et al. 2014). Therefore, a two step process was used: firstly, the complete network was analysed to establish the overall number of driver concepts, and to determine if the number and type of concept was in agreement with the findings of Liu et al. (2011). Secondly, the number of driver concepts was determined for the highly important subset in order to identify a set of driver concepts, that could be practically used by stakeholders, to propose; (1) which driver concepts were controllable (Penn et al. 2014), and (2) how controllable driver concepts should be managed to achieve blanket peatland land–use objectives.

The highly important subset formed a network in which three concepts (nurse crop, disturbance, and streams, springs and flushes) became disconnected (i.e. $k = 0$). As a result these concepts could not be part of any configuration of driver concepts used for network control, and so were removed from the network (therefore $N = 31$). The maximum matching process resulted in the classification of concepts into two groups; those that could be used to 'steer' the network (drivers), and those that were not classified as drivers which were labelled 'ordinary' in this case (Figure 4.8). The significance of the relationship between the number of connections within each group (driver and ordinary), was determined by use of the two–sample Wilcoxon rank sum test (R Core Team 2015); chosen because the number of connections per concept was not normally distributed (Figure 4.2). The calculation of driver concepts produced a number of possible configurations that were compared

[1] Algorithm developed and provided by Dr. David J.B. Lloyd, Dept. of Maths, the University of Surrey

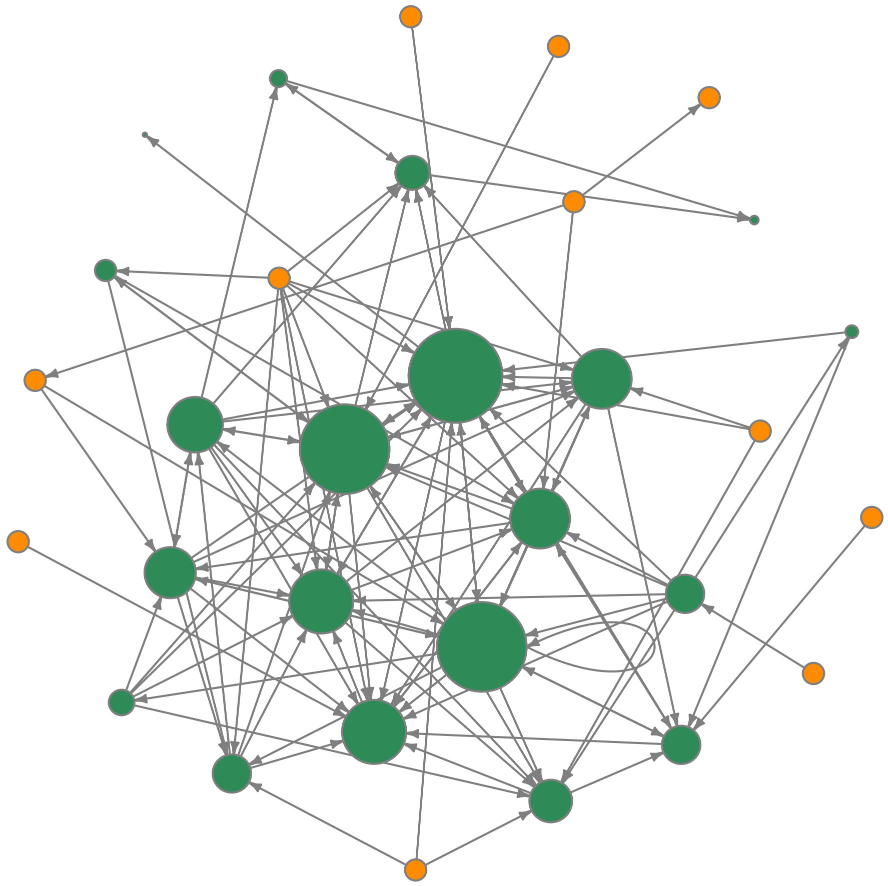


Figure 4.8. Driver concepts for the blanket peat network (highly important subset) co–developed in the Peak District. Orange nodes represent driver concepts, green nodes represent ordinary concepts. To improve visualisation, the size of driver concepts is twice their mean number of connections $\langle k_D \rangle^{hi} = 2.46$. The size of the ordinary concepts is in proportion to the actual number of connections of each concept. The mean number of connections for ordinary (green) concepts is $\langle k_O \rangle^{hi} = 10.65$. hi indicates the values are in relation to the highly important subset.

and aggregated into a single set of concepts. Although this step was not strictly necessary, because all configurations should produce full control, the participatory process was simplified by the inclusion of a single aggregate matching for use by stakeholders.

4.4.3 Results

There were 2,328 configurations of driver concepts for the complete network comprising (N_D) 40 concepts ($n_D = 0.317$). Further analysis revealed that the majority of concepts were found in all configurations. Aggregating all configurations into a single set produced 49 unique driver concepts. The mean number of connections of the driver concepts within this aggregated configuration was $\langle k_D \rangle = 4.020$, whereas the mean number of connections within the non–driver (ordinary) concepts was $\langle k_O \rangle = 11.052$, and the mean number of connections for all concepts was $\langle k \rangle = 8.317$. There was a significant difference in the number of connections between driver and ordinary concepts (Wilcoxon rank sum test statistic W = 2734, $p < 0.001$; Figure 4.9a). This confirms the findings of

Liu et al. (2011) who identified that drivers and hubs were different types of concepts to hubs, and the number of driver concepts (N_D) is a feature of the underlying distribution of interactions within the network.

It was unclear if a similar density of driver nodes would be found in the highly important subset network because it was relatively small (31 concepts), with few interactions (120). However, two configurations with $N_D = 10$ were identified (Table 4.2, Figure 4.8). Each configuration could be used as the minimum set of concepts needed to control the subset network (hi = highly important subset). Again there was a significant difference between the number of connections related to driver and ordinary concepts (Wilcoxon rank sum test statistic W = 198.5, $p < 0.001$; Figure 4.9b). The mean number of connections of the driver concepts within the aggregated configuration was $\langle k_D \rangle^{hi} = 2.455$, the mean number of connections within the non-driver (ordinary) concepts was $\langle k_O \rangle^{hi} = 10.650$, and the mean number of connections for all concepts was $\langle k \rangle^{hi} = 7.742$.

The density of driver concepts $n_D^{hi} = 0.323$, was largely unchanged from the density required to control the whole network. Although there were fewer concepts and interactions in the highly important network, this result suggests that hubs remain an important part of the framework of interactions because the value of n_D^{hi} would have been smaller than n_D if the network had become less heterogeneous. However, two driver concepts (mosses and game–keeping), which were present in the configurations for the hi subset, did not appear in the configuration required for full control of the complete network. This is likely to be because of the removal of interactions, to mosses and game–keeping, from concepts that were not rated as highly important. I chose not to reject these two concepts from the aggregate configuration because both are clearly perceived to be important in the structure of the highly important subset identified by stakeholders. Finally, rather than ask stakeholders to select one of the configurations, both were aggregated. The 11 driver concepts (Table 4.2) were later used in a workshop to provide the basis of discussion about how blanket peatlands should be managed to achieve three land–use objectives (Section 4.5).

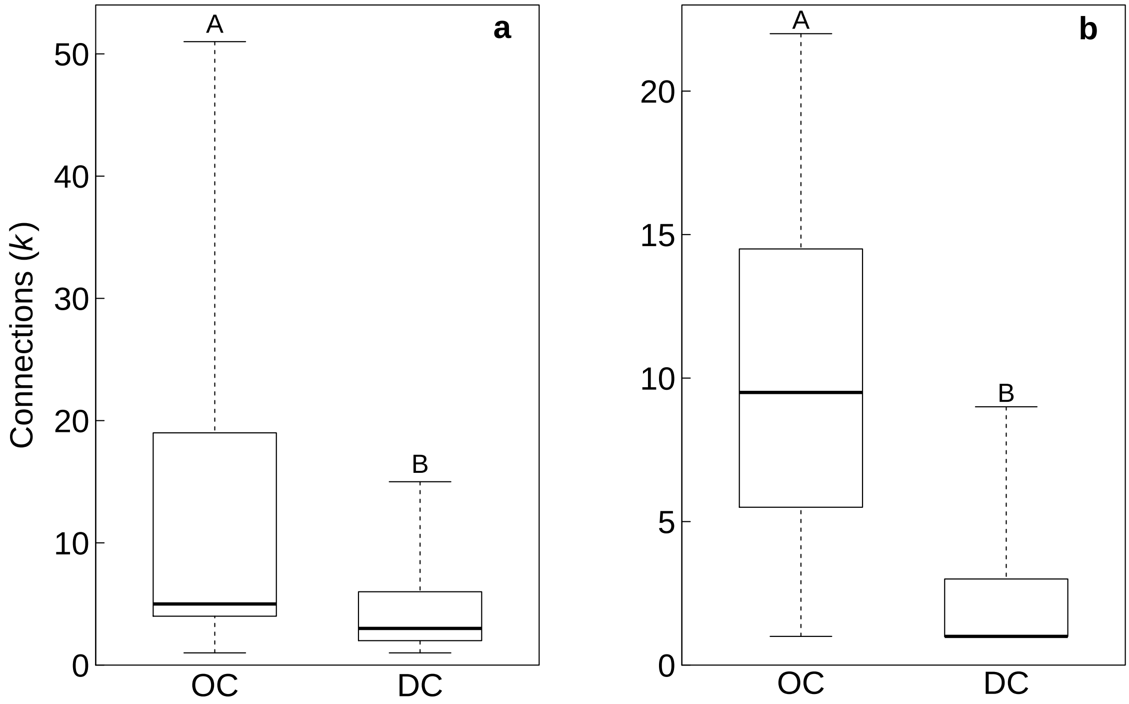


Figure 4.9. Blanket peatland network: the number of connections of driver and ordinary concepts. **a)** Complete network (126 concepts, 524 connections). **b)** Highly important subset (31 concepts 120 connections). OC = ordinary concepts, DC = driver concepts. Each box represents the interquartile range, the thick line is the median number of connections and whiskers the range. Capital letters indicate significant differences between groups of concepts ($p < 0.001$).

Table 4.2. Blanket peatland network driver concepts. Two configurations were calculated that represent an alternative set of concepts that could be used to 'steer' the blanket peatland network.

	Configuration 1	Configuration 2
1	Burning intensity	Burning intensity
2	Land management	Land management
3	Afforestation	Afforestation
4	Mosses*	Mosses
5	Agri–env. Schemes	Agri. Env. Schemes
6		Game–keeping*
7	Revegetation	Revegetation
8	Gully blocking	Gully blocking
9	Climate change	Climate change
10	Funds for restoration	
11	Deposition of eroded peat	Deposition of eroded peat

Agri. env. = agri–environment

* = Not identified as drivers in the configuration for the complete network.

4.4.4 Summary of key findings from this section

There were 49 driver concepts for the complete blanket peatland network and 11 for the highly important subset. I found that driver concepts of both blanket peatland networks (complete and highly important subset) had relatively few connections and were distinct from hubs (in agreement with the findings of Liu et al. 2011). When coupled with the findings of Section 4.3, these results suggest there are two classes of concept that should be considered when exploring land–use: hubs are key to the structural integrity of the blanket peatland network, whilst drivers provide a route to promote or maintain mutualistic hubs, or reduce the negative effect of antagonistic hubs such as wildfire. These results have addressed the first part of the challenge proposed at the beginning of this section which was to identify the concepts that provide control of the network. Section 4.5 addresses the second part of the challenge which is to determine how, and if, driver concepts should be managed to achieve land–use objectives.

4.5 Blanket peatland land–use objectives

4.5.1 Background

The complex structure of the blanket peatland network was shown to include two types of concept; hubs and drivers (Sections 4.3 and 4.4). The results from these sections identified (1) the importance of highly connected, mutualistic, hubs in maintaining the framework of interactions that underpins blanket peatland resilience, and (2) for a subset network based on the highly important classification, a set of 11 driver concepts that were distinct from hubs because they had few connections. The aim of this section is to further develop the second of these results (identification of driver concepts), and ask how driver concepts should be managed to achieve land–use objectives. Because land–use decisions in peatlands often revolve around multiple objectives, that can be contested (e.g. Douglas et al. 2015), a group of blanket peatland stakeholders were asked to; (1) classify the controllability of driver concepts, and (2) determine how controllable concepts should be managed to achieve three land–use objectives. The objectives were congruent with the principles of the Convention on Biological Diversity (Secretariat of the CBD 2010), encompassed by the aims suggested by Maltby (2010) to address peatland land use conflict (Section 2.3);

1. Enhance the vitality of the rural economy

2. Maintain or enhance the contribution to environmental security (the mitigation of climate change and supply of high-quality water)

3. Support the well–being of the wider local and national communities

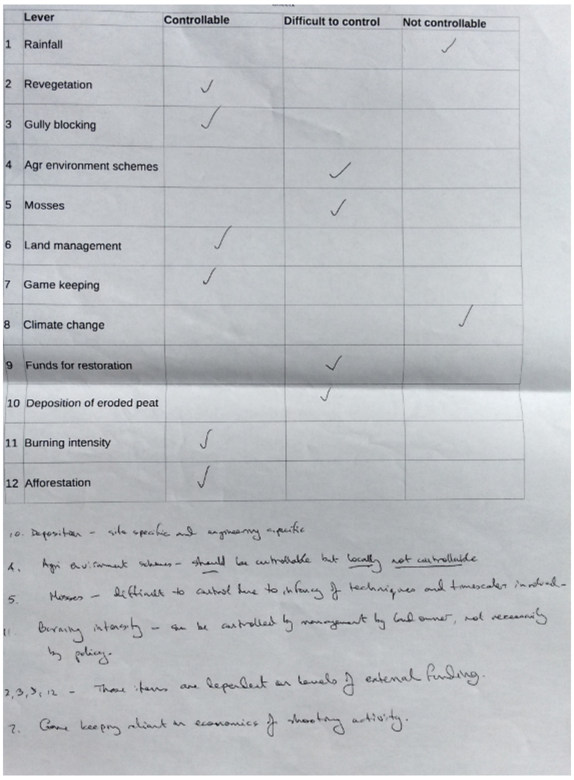
4. Safeguard important biodiversity and cultural heritage

The three land–use objectives selected by me were, (a) maintain or increase carbon storage, (b) improve the quality of water supplied, and (c) support local livelihoods.

4.5.2 Methods

The aggregate set of 11 driver concepts, calculated from the highly important blanket peatland subset, was used to determine how land–use objectives could be achieved. In order to simplify the process, the individual interactions between driver and ordinary concepts were not included. This meant that discussions could revolve around the driver concept itself, and not how it interacted with each individual ordinary concept connected to it. I used an approach that partly followed that of Penn et al. (2014) in that participants were asked, during a workshop, to state how 'controllable' were driver concepts in order to identify those that be could be influenced by participants. As a development of the process of Penn et al. (2014), I then asked participants to determine how the 'controllable' driver concepts could be managed to deliver the three land–use objectives for blanket peatlands (a–c) stated above. Eleven stakeholders participated in the workshop, nine of whom attended the original mapping workshops, and five of whom attended the validation workshop. Three other participants either cancelled their attendance the day prior to the workshop, or did not attend on the day. All three had attended mapping workshops and two had attended the validation workshop.

Participants worked in three mixed groups so that there was an opportunity to share experiences and rationales when making choices and decisions. There were two sessions and participants worked in groups: at the end of each session, the decisions made in each group were discussed by all participants. At the beginning of the workshop, I presented a review of the process to date, the analysis of the output from previous workshops, and the aims and objectives for the current workshop, i.e. the classification and selection of driver concepts, and to propose actions to deliver each of the three land–use objectives. Findings related to the importance of hubs, and the identity of driver concepts were also presented. For workshop activities, driver concepts were described as 'levers' (Penn et al. 2014) that could be adjusted, in order to convey the purpose of the concepts as an approach blanket to peatland management.



	Lever	Controllable	Difficult to control	Not controllable
1	Rainfall			✓
2	Revegetation	✓		
3	Gully blocking	✓		
4	Agr environment schemes		✓	
5	Mosses		✓	
6	Land management	✓		
7	Game keeping	✓		
8	Climate change			✓
9	Funds for restoration		✓	
10	Deposition of eroded peat		✓	
11	Burning intensity	✓		
12	Afforestation	✓		

10. Deposition – site specific and engineering specific
4. Agri environment schemes – should be controllable but locally not controllable
5. Mosses – difficult to control due to infancy of techniques and timescales involved –
11. Burning intensity – can be controlled by management by land owner, not necessarily by policy.
2, 3, 5, 12 – These items are dependent on levels of external funding.
7. Game keeping reliant on economics of shooting activity.

Figure 4.10. Classification of driver concepts. Example output from Peak District workshop to classify driver concepts as as ‘controllable’, ’difficult to control’ and ‘not controllable’. The latter class was deselected for the remainder of the workshop.

In the first instance each group was given a list of driver concepts, and asked to classify concepts as ‘controllable’, ‘difficult to control’ and ‘not controllable’ . Each group was asked to make notes of the reasons for their selections (e.g. Figure 4.10). Next, with all participants working together, the reasons for selecting the controllability class for each concept was discussed so that groups could make changes to classifications before a final aggregated list was agreed as a group. The concepts identified as ‘not controllable’ were deselected for the second stage of the workshop.

Using the aggregated list of controllable driver concepts, and working with one land use objective at a time, each group was asked to discuss how each concept should be adjusted to deliver that specific objective. Initially three options were proposed as the method to describe how a driver concept should be adjusted; to promote (i.e. to increase the activity related to the concept), to protect (to continue with the current level of activity related to the concept) or to reduce. However, following a group discussion, a fourth category, to do nothing, was added to differentiate between the effort needed to maintain the activity related to a driver concept (to protect) and the abandonment of all activity (to do nothing). The ‘protect’ category was perceived by some participants to be somewhat ambiguous, and so the classification ‘maintain’ was adopted instead. A worksheet was provided for each land–use objective, and worked on in turn by each group. And although discussions were held within groups, group members were asked to select one of the four options of their own choice by placing a sticker in a box on the worksheet that would identify the

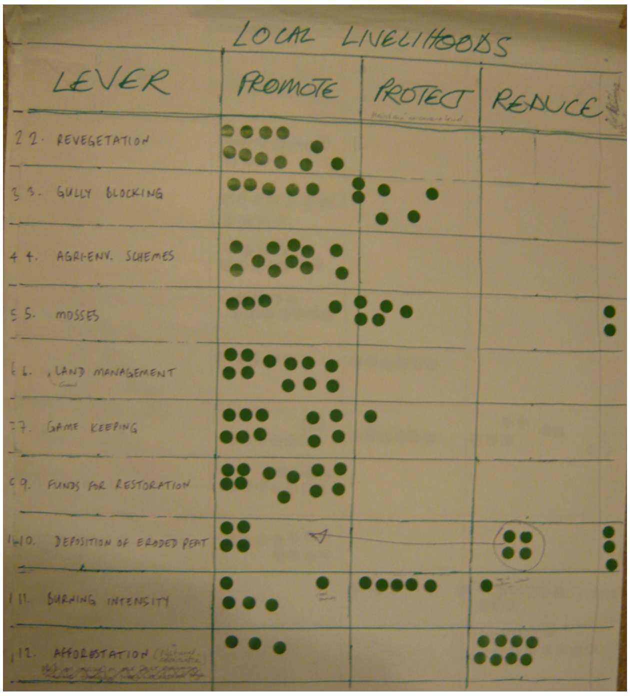


Figure 4.11. Promote, maintain, reduce. Proposed actions to achieve land–use objectives. Example output from Peak District workshop to identify actions to deliver an improvement in local livelihoods based on driver concepts (identified as 'Lever' in this example).

relevant adjustment option. This meant that each option could be selected a maximum of eleven times and the spread of opinion through all participants could be seen (Figure 4.11).

4.5.3 *Results*

Controllability

The final list of driver concepts included one 'not controllable' (climate change), 'three difficult to control' and six 'controllable' concepts (Table 4.3). One concept was split between 'controllable' and 'difficult to control' (deposition of eroded peat, Table 4.3). However following a discussion with all participants, both 'controllable' and 'difficult to control' sets of concepts were grouped together for the promote, maintain, reduce exercise, and climate change was excluded from further discussion[2]. Several themes evolved from worksheet comments, and subsequent discussion of the classification of driver concepts. Firstly, the availability of funding was seen as a significant factor in all concepts classified as controllable and difficult to control, except burning intensity and afforestation. The former was seen as a mix of the political will to make changes to the allowed practice of burning on blanket peatland and the desire of the land owner to increase productivity of a grouse moor, and the latter driven mainly by government policy. It was noted that although

[2]NOTE: A computational error resulted in the initial identification of 12 driver concepts which were the 11 included here plus total rainfall. This error was not discovered until after the results had been presented and discussed with participants. However, total rainfall was determined by participants as 'not controllable' and so was not used to determine management actions. The error was resolved and does not feature in the work presented in this thesis.

there may be funding available for restoration activities such as gully blocking and revegetation, implementation is dependent on landowner agreement. Although sources of funding themselves were perceived to be interconnected to most driver concepts, both agri–environment schemes and other funds used for peatland restoration, such as payment for ecosystem services (PES) schemes, were perceived to be difficult to control because of both the timescales required, and because the sources themselves are often not local (e.g. national government and European Union institutions). Two groups made specific notes on the technical difficulties associated with some driver concepts such as the impact of site, the infancy of techniques to restore mosses (including *Sphagnum* spp.), and the length of time needed to implement or realise change.

Table 4.3. Classification of driver concepts controllability. The three dots represent each group that took part in the Peak District workshop.

Driver concept	Controllable	Difficult to control	Not controllable
Burning intensity	•••		
Land management	•••		
Afforestation	•••		
Mosses		•••*	
Agri–environment Schemes		•••	
Game–keeping	•••		
Revegetation	•••		
Gully blocking	•••		
Climate change			•••
Funds for restoration		•••	
Trap†eroded peat	••	•	

†Workshop participants replaced deposition with trap

* = One group placed mosses between controllable and difficult to control

Promote, maintain, reduce, do nothing

Workshop participants agreed that regardless of management objective, there should be an increase in funding sources and activities to revegetate bare peat (Figure 4.12). All activities that were perceived to be beneficial to water quality were also perceived to be beneficial for increased carbon storage. Both water quality and carbon storage were perceived to benefit from; (1) an increase in gully blocking, trapping of eroded peat, and attempts to encourage more mosses; and (2) reductions in afforestation (either planting or ‘scrubbing up’) of blanket (deep) peat (rather than the planting of clough woodland), and burning intensity (Figure 4.12).

The main differences in the actions that were proposed between the three objectives were related to supporting local livelihoods. Almost all participants thought there should be an increase in game–keeping which resulted in a nuanced conversation about the future role of game keepers. Whilst game keeping was associated with burning intensity, which was seen to be detrimental to water quality and carbon storage, participants discussed reducing some game–keeping activities (such as burning) whilst increasing others such as of wildfire risk management. This point was highlighted in the difference between the game–keeping actions related to water quality, and those related to carbon storage. In the case of water quality, there was a fairly even split of opinion across the options to promote, maintain or reduce, whereas for carbon storage, the majority of participants proposed maintaining the current level of activity. Participants proposed an increase in general land management activities in order to increase local livelihoods, although it appeared that stakeholders only partially recognised that restoration activities, such as gully blocking and mosses, could fulfil this role because there was a split between promoting and maintaining these activities. Finally there was recognition by the majority of participants that burning intensity is currently linked to local livelihoods through game–keeping, with a split between an increase in burning intensity and maintaining the current frequency (Figure 4.12).

Discussions within and between groups, took place throughout the workshop which served to reinforce the complexity of decision–making in relation to the three land use objectives, but also demonstrated the collaborative approach adopted by participants despite differences in land–use position. Notably the group, prompted by a land owner, further developed earlier comments that game–keeping could perhaps, at least to some extent, be decoupled from managed burning with a greater emphasis placed on heather cutting, and by carrying out restoration activities which could promote local livelihoods, improve water quality, and increase carbon storage. Some participants also commented that it was beneficial to be involved in a process where work on problems was done in mixed groups to gain a greater understanding of the perspectives of other land users, and the wider implications of land–use decisions. These discussions support the suggestions of others that the benefits of participatory modelling extend beyond model outputs (e.g. Voinov and Bousquet 2010; Penn et al. 2013).

4.5.4 Summary of key findings from this section

Peatland stakeholders classified 10 driver concepts as controllable or difficult to control: climate change was classed as uncontrollable. In several cases there was unanimous agreement about how driver concepts should be managed. The group agreed that an increase in agri–environment funding, funds for restoration and revegetation would promote the joint achievement of all three land–use objectives. Actions for improving both water quality and carbon storage were generally

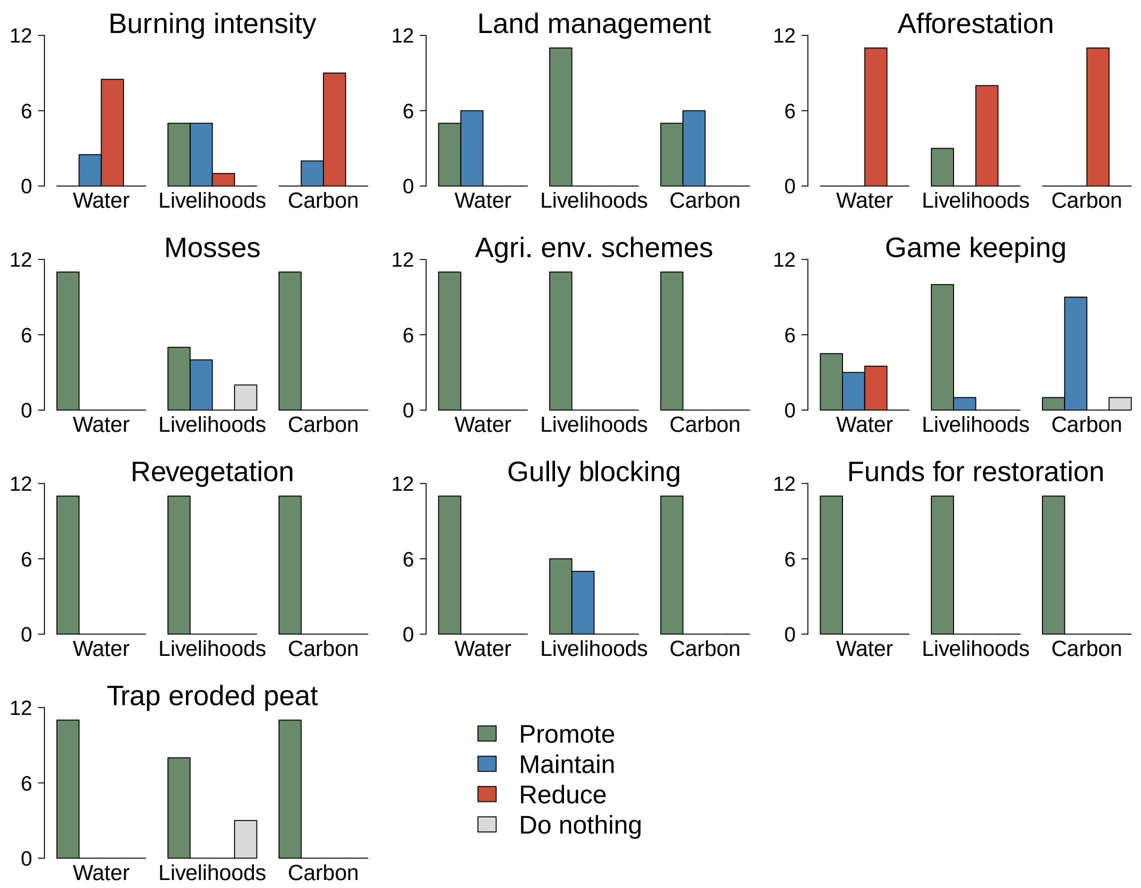


Figure 4.12. Land–use objectives: adjustments to driver concepts proposed during Peak District Workshop. Participants suggested changes to driver concepts in order to; improve water quality, support local livelihoods, and to maintain or increase carbon storage. Values on the 'y' axes represent the number of workshop participants (11).

coincident; but differences occurred in the changes that were suggested to support local livelihoods. This result suggests that some peatland restoration activities were not completely recognised as a means to deliver this objective, and may be related to the potential of these driver concepts to create shallower water tables. However, proposals were made and discussed that could decouple game keeping from managed burning, and reduce the perceived negative impact on both carbon storage and water quality objectives. These results provided the basis to amend how driver concepts interact with the blanket peatland network, and to investigate the impact of the choices made by stakeholders on hubs (which provide structural resilience to the blanket peatland network), and carbon storage.

4.6 The impact of land–use objectives on peatland carbon storage

4.6.1 Background

The aim of this section is to investigate how the decisions made by stakeholders to achieve three land–use objectives (maintaining or increasing carbon storage, improving the quality of water supplied, and supporting local livelihoods), are likely to impact carbon storage in blanket peatlands. A fuzzy cognitive map of the causal interactions within blanket peatlands was co–developed by blanket peatland stakeholders (Chapter 3), and has been used for the analysis of network structure in Sections 4.3 and 4.4. Using the tools of network analysis my results show that, (1) the framework of interconnections that form the blanket peatland network is maintained by a few hubs (Section 4.3), and (2) the controllability of the structure is reliant on a second class of concepts known as driver concepts, that were identified in Section 4.4.

I propose that both of these structural properties can be coupled to investigate how changes to driver concepts, suggested by stakeholders (Section 4.5), impact carbon storage in blanket peatlands, and network resilience by investigating the impact on hubs. I use a simple mathematical model to compare the steady–state output of the original co–developed cognitive model, to the outputs based on each land–use objective.

4.6.2 Methods

Fuzzy cognitive maps: generating model outputs

Fuzzy cognitive mapping (Kosko 1986) enables knowledge about a set of causal interactions to be modelled as a dynamical system (Chapter 3). Kosko (1986) proposed that a vector $\mathbf{C}$ (often called the state vector), at time t, should represent the current state of all concepts $(C_1 \cdots C_n)$ within the model. The causal interactions between the concepts are represented by a connectivity (or adjacency) matrix $\mathbf{W}$ which is used to iteratively modify the value of concepts according to the strength of their interactions. The connectivity matrix is shown as;

$$\mathbf{W} \equiv W_{ij} = \begin{array}{c c} & \begin{array}{cccc} C_1 & C_2 & \cdots & C_j \end{array} \\ \begin{array}{c} C_1 \\ C_2 \\ \vdots \\ C_i \end{array} & \begin{bmatrix} w_{11} & w_{12} & \cdots & w_{1j} \\ w_{21} & w_{22} & \cdots & w_{2j} \\ \vdots & \vdots & \ddots & \vdots \\ w_{i1} & w_{i2} & \cdots & w_{ij} \end{bmatrix} \end{array},$$

where *w* represents the strength (weight) stakeholders have assigned to the causal relationship between two concepts [-1, 1], and *ij* represents the *i*th row and *j*th column of a square matrix. In the example above, w_{23} defines the interaction strength between the concept in the second row and the concept in the third column. The connectivity matrix is square because rows represent the causal interactions that leave a concept C_i, and columns the causal connections that enter a concept C_j. All possible pairs of interactions are, therefore, represented in the connectivity matrix; although in many cases $w_{ij} = 0$ to signify that there is no interaction. In the state vector $\mathbf{C}$, each concept is given an initial value that can represent its activation status (Kontogianni et al. 2012), real values identified by stakeholders (Penn et al. 2013), or some fraction such as the amount of vegetation cover.

The original mathematical model by Kosko to represent a fuzzy cognitive map is given by Carvalho (2013) as,

$$C_j^{t+1} = f\left(\sum_{\substack{i=1 \\ i \neq j}}^{n} w_{ij} \times C_i^t\right), \tag{4.6}$$

where C_j^{t+1} is the value of the concept at the next model iteration. C_i^t is the current concept value at time t, w_{ij} are the weights between concepts, and f is a function to transform the model outputs into the interval [0, 1]. This process is repeated for all concepts and a new state vector $\mathbf{C}$ at time $t + 1$ is calculated. A solution is reached when a stable value is achieved for all concepts; which may require several model iterations. The approach assumed that a concept could not cause itself, hence $i \neq j$, but some concepts are self-reinforcing, such as different types of peatland vegetation that 'engineer' their own environment (e.g. Eppinga et al. 2009). Later variations of the model (Equation 4.7) added the current value of concepts to the calculation of the next model iteration (Stylios and Groumpos 1999) which incorporates self–causality (Carvalho 2013; Groumpos 2014). This approach to self–causality has since been used in a number of studies (e.g. Mei 2014; Gray et al. 2015), and is the method I followed (Carvalho 2013, Equation 3);

$$C_j^{t+1} = f\left(\sum_{\substack{i=1 \\ i \neq j}}^{n} w_{ij} \times C_i^t + C_j^t\right). \tag{4.7}$$

In Chapter 3 (Section 3.7.4), I discussed that variable interaction times were not considered in the model used here (e.g. Hagiwara 1992): all concepts are assumed to change at the same rate. However, because of the additional complexity of collecting this data from workshop participants and the challenging nature of the mapping process, time–based data was not collected. Although data about delays between concept interaction could have been estimated from published literature and added separately, it would have gone against the spirit of the participatory process, which was to co–develop the model in a transparent way with a mixed group of peatland stakeholders. Hobbs et al. (2002) also proposed that model outputs generated without time–based data were still useful if modelling focussed on the steady state solution rather than transient dynamics. Considering the aim to explore the end point of the impact of proposed changes on carbon storage, and the limited time available in workshops, I decided to continue with the simple model of Equation 4.7.

Activation function

Depending on the configuration of the connectivity matrix, some linear models may not converge into a stable solution. The use of an activation function f can resolve this issue, but studies have shown that the choice of activation function (or parameters) can alter the outputs of the fuzzy cognitive map (Penn et al. 2013; Knight et al. 2014). Although alternative methods or functions have been suggested (e.g. Wise et al. 2012; Rickard et al. 2015; Vogt et al. 2015), they are likely to require additional parameterisation (which may restrict their use by non–technical stakeholders) and further complicate the participatory process. I addressed this issue in two steps; firstly I randomly selected different concept input values and checked for differences in model outputs, and secondly I assessed the plausibility of model outputs by reviewing the literature published about the impact of peatland land uses (Chapter 2 and Figure 2.10) (comparable to the suggestion of Jetter and Kok 2014).

The activation function can take a number of forms, including linear (Penn et al. 2013), ramp (Hobbs et al. 2002), hyperbolic tangent, weighted power mean (Rickard et al. 2015), sensitivity functions (Vogt et al. 2015), or sigmoid (Papageorgiou et al. 2011; Groumpos 2014; Mei 2014), the latter being the most popular. I chose to use the sigmoid function defined by Papale and Valentini (2003),

$$f(x) = \frac{1}{1 + e^{\frac{-x}{\rho}}}, \tag{4.8}$$

where x is the weighted input and ρ, the slope of the sigmoid curve. Both Hobbs et al. (2002) and Knight et al. (2014) state that the parameters selected for activation functions can result in more than one final solution for the fuzzy cognitive map depending on the initial concept values. Knight et al. (2014) have defined the parameter space, depending on the number of fuzzy cognitive map concepts, for the sigmoid function under which there is a unique solution. As different values of ρ result in different model outputs (even if unique), multiple solutions were tested for by creating and comparing the results of 100,000 random sets of initial values for concepts of 0 or 1. In all cases the final output was identical, and although many more permutations were possible, the result was deemed acceptable. Added to this, model outputs also appeared to be plausible (Sections 4.6.3 and 4.7), and therefore ρ was set to 1 following Papale and Valentini (2003).

Land–use objectives: new interaction weights for driver concepts

The model of blanket peatland concepts (highly important subset), was run to a stable condition for; (1) the original set of values that were finalised at the validation workshop, and (2) each of the three land–use objectives based on driver concepts from Section 4.5. In many instances fuzzy cognitive maps have been used as simple models to explore system dynamics in order to support land–use decisions by changing initial concept values or interaction strengths, or where new concepts and/or interactions have been added (e.g. Kontogianni et al. 2012; Soler et al. 2012; Penn et al. 2013). To investigate the impact of the changes to driver concepts proposed by workshop participants, the value of the interactions from each driver concept was recalculated according to the following conditions,

$$w_{ij}^{new} = \begin{cases} w_{ij}^{orig} & \text{if } M > |P - R - DN|, \\ 0 & \text{if } DN > |P + M + R| - DN, \\ x \times w_{ij}^{orig} + w_{ij}^{orig} & \text{otherwise,} \end{cases}$$

where $-1 \geq w_{ij}^{new} \leq 1$, $x = \frac{-R+|P-M-DN|}{n}$ is the fraction that a driver concept was promoted or reduced by participants, and $P, M, R, DN =$ the sum of the votes for each of the categories; promote, maintain, reduce or do nothing (Table 4.4). The interval [-1, 1] was used because I reasoned that if, for example, 10% of a peatland had been revegetated then there would be scope to revegetate a further 90%. If however, 90% had been revegetated, any increase could only incorporate an additional 10%.

Table 4.4. Changes made to driver concepts interactions for the blanket peatland fuzzy cognitive map.

Driver concept*	Water	Livelihoods	Carbon
Burning intensity	-0.55	–	-0.64
Land management	–	1	–
Afforestation	-1	-0.46	-1
Mosses	1	–	1
Agri–environment Schemes	1	1	1
Game–keeping	–	0.82	–
Revegetation	1	1	1
Gully blocking	1	0.1	1
Funds for restoration	1	1	1
Trap eroded peat	1	0.46	1

'–' = no change was made to the interaction value

1 or -1 = maximum increase or decrease

< 1 or > -1 = the fraction to increase or decrease the interaction value

* = climate change was classed as 'uncontrollable' and excluded from discussions

4.6.3 Results and discussion

Fuzzy cognitive maps are semi-quantitative and as such, the output should be considered in relation to the current state model (Penn et al. 2013) (Figure 4.13). The model produced a steady–state value for each concept in < 20 iterations (Figure 4.13). I interpreted the impact of land–use objectives on carbon storage, and other concepts, by comparing the current steady–state rank of a concept, to the change in rank as a result of each land–use objective (Table 4.5). Water and carbon storage land–use objectives resulted in similar rank changes, which was expected because most of the changes made to driver concepts to achieve these two objectives were the same (Table 4.4). The livelihoods objective also resulted in changes to the current state output which, in general, favoured concepts related to game keeping; although some changes also mirrored the direction, if not magnitude, of those for water and carbon (Table 4.5). Because the aim of this modelling was to investigate the impact of changes to the interaction of driver concepts on ordinary concepts (including hubs), I removed driver concepts from the calculation of changes in rank.

I interpreted a higher position in the rankings (Table 4.5) to indicate an increase in the concept as a result of the proposed changes for that particular objective. Figure 4.14 shows the impact of changes to concept ranking following the introduction of the water quality objective after the original model outputs had reached a steady state (i.e. > 20 model iterations). In some cases, careful interpretation of the change in rank was required. For example, the rank of water–table depth fell in response to all land–use objectives, which equated to a shallower water table (a reduction in water–table depth). And a reduction in water–table depth is often one of the key goals of blanket peatland restoration (e.g. Blundell and Holden 2015). The most notable change to any concept across all objectives was to *Sphagnum* cover which was ranked 18 in the current state results, 15 in the livelihood results, and 7 and 6 in the water and carbon storage results respectively. Peat accumulation was ranked 14 in the current state, 11 in the livelihood results, 6 in the water results, and 7 in the carbon results. Notably the increase in peat accumulation did not result in a large increase in carbon storage ranked 20 in the current state, 18 and 17 in the water and carbon storage results, but remained at 20 in the livelihoods results. I suggest that this is because an increase in other concepts, to support livelihoods, act to reduce carbon storage. Erosion was also reduced from 8 in the current state results to 11 in the water and carbon results, and 9 in the livelihoods objective (Table 4.5).

The objectives to increase water quality and improve or maintain carbon storage resulted in increases of *Sphagnum* cover and peat accumulation. There were reductions in water–table depth, bare peat, wildfire, heather cover, CO_2 emissions (8 to 11) and global warming potential (9 to 12), and no change in the rank of managed burning (19). The objective to support local livelihoods resulted in a small increase in managed burning (from 19 to 18), *Sphagnum* cover (from 18 to 15) with a subsequent increase in peat accumulation (14 to 11), but without any overall increase in carbon storage (20). According to the cognitive model, an increase in managed burning has negative impacts on *Sphagnum* cover, peat accumulation and depth of peat, whilst increasing erosion and bare peat. Water tables also became shallower (from 17 to 19, see comment above regarding interpreting changes to water–table depth), but to a lesser degree than the other objectives. The livelihoods objective resulted in the largest decrease in wildfire (from 13 to 16), although heather cover remained the same as the current state (12). Carbon dioxide CO_2 emissions remained the same as the current state (7), but global warming potential increased under the livelihood objective (9 to 8).

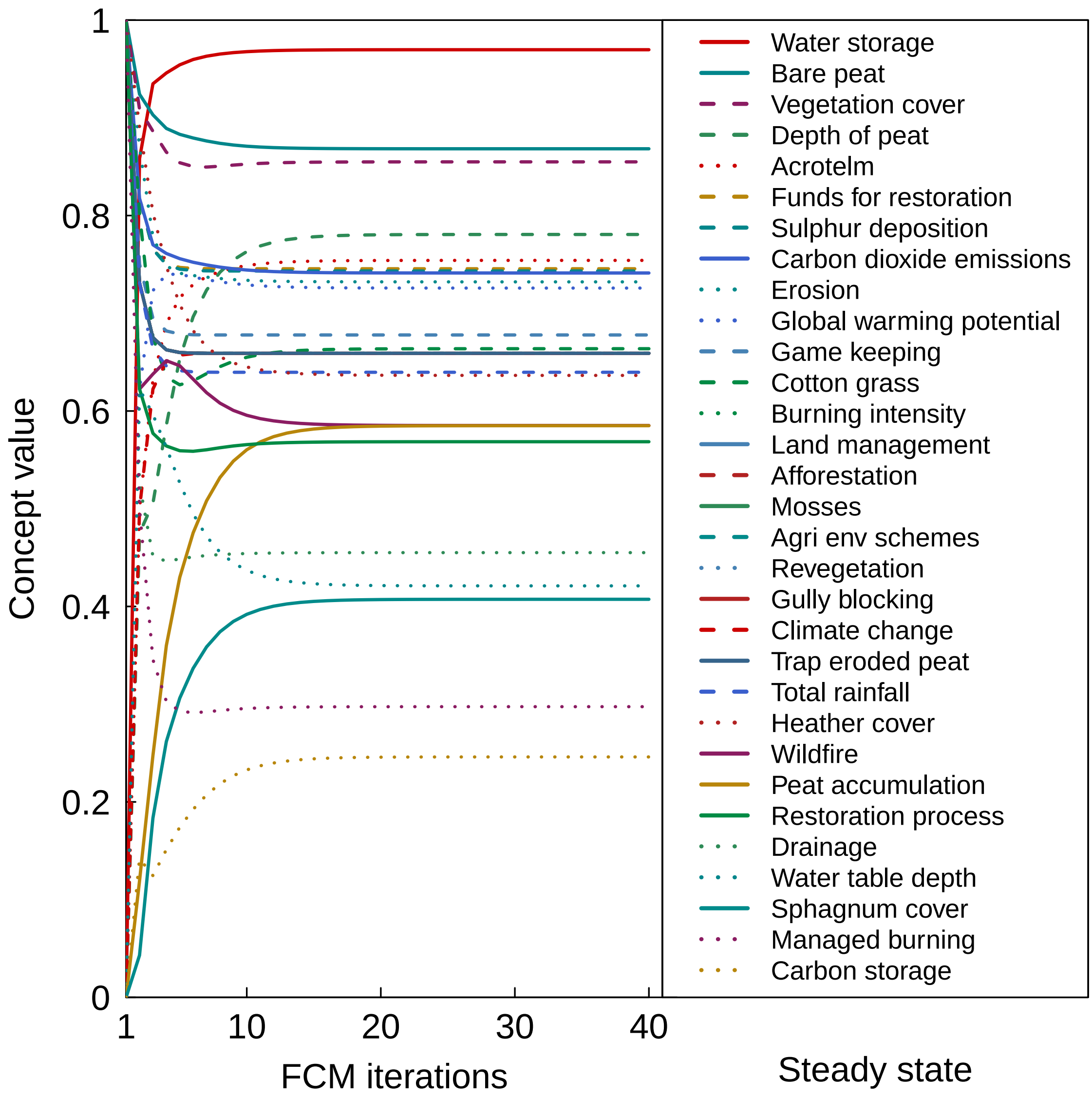


Figure 4.13. Blanket peatland current state fuzzy cognitive map (FCM): highly important concepts. The output shown uses the original interaction weights finalised at the validation workshop and incorporates a sigmoid activation function.

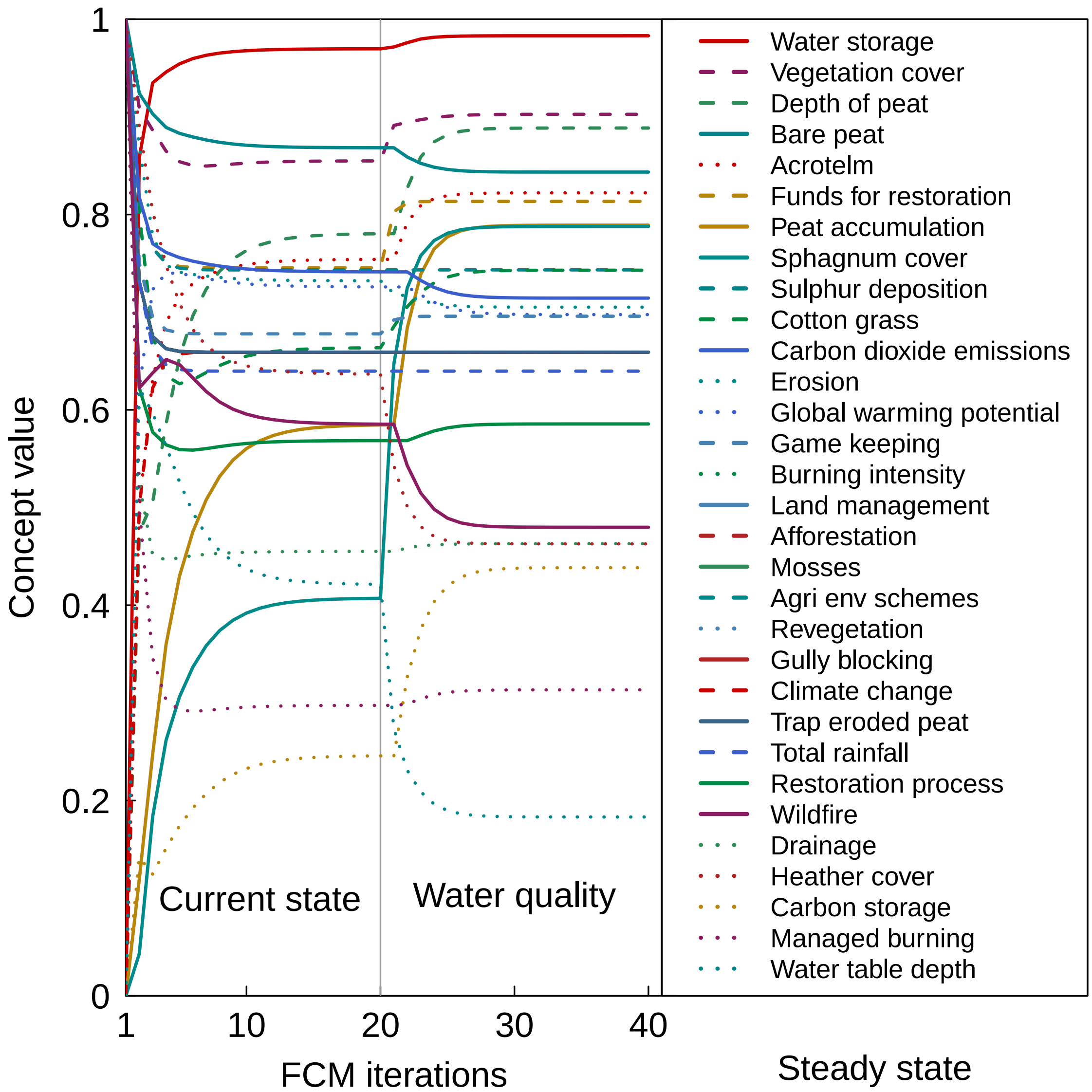


Figure 4.14. Blanket peatland future state fuzzy cognitive map (FCM): highly important concepts. Water quality land–use objective The output shown uses the current state interaction weights until iteration 20 where the interaction weights determined from the land use decisions workshop to deliver improved water quality were introduced.

Table 4.5. FCM final stable concept ranking for all model outputs.

Concept	Current state	Water	Livelihoods	Carbon
Sphagnum cover	18.0	7.0	15.0	6.0
Vegetation cover	3.0	2.0	2.0	2.0
Acrotelm	5.0	5.0	5.0	5.0
Sulphur deposition	6.0	8.0	6.0	8.0
Carbon storage	20.0	18.0	20.0	17.0
Carbon dioxide emissions	7.0	10.0	7.0	10.0
Managed burning	19.0	19.0	18.0	19.0
Restoration process	15.0	14.0	14.0	14.0
Heather cover	12.0	17.0	12.0	18.0
Water storage	1.0	1.0	1.0	1.0
Depth of peat	4.0	3.0	4.0	3.0
Water table depth	17.0	20.0	19.0	20.0
Peat accumulation	14.0	6.0	11.0	7.0
Total rainfall	11.0	13.0	13.0	13.0
Erosion	8.0	11.0	9.0	11.0
Wildfire	13.0	15.0	16.0	15.0
Cotton grass	10.0	9.0	10.0	9.0
Drainage	16.0	16.0	17.0	16.0
Bare peat	2.0	4.0	3.0	4.0
Global warming potential	9.0	12.0	8.0	12.0

Note: (1) in case of ties concepts are ranked according to mean order, and (2) driver concepts are not included in rank comparison.

In order to understand the impact of land–use objectives on the resilience of blanket peatlands, I classified concepts with with ≥ 15 interactions as hubs. There were five hubs in the highly important network (Table 4.6) that represented approximately 16% of the total number of concepts within the network. The same five concepts were also among the 16% of the most connected concepts in the complete network, where their number of interactions (degree) was between 19 and 51. Although this classification might appear to be somewhat arbitrary (Barabási et al. 2011), it can be justified because the role of these concepts in maintaining network structure has been tested (Section 4.3). My results show that the changes made to driver concepts maintained or increased the rank of hubs that positively affect peatland resilience (*Sphagnum* cover, depth of peat, and peat accumulation), whilst the rank of hubs that that negatively impact blanket peatlands (increased

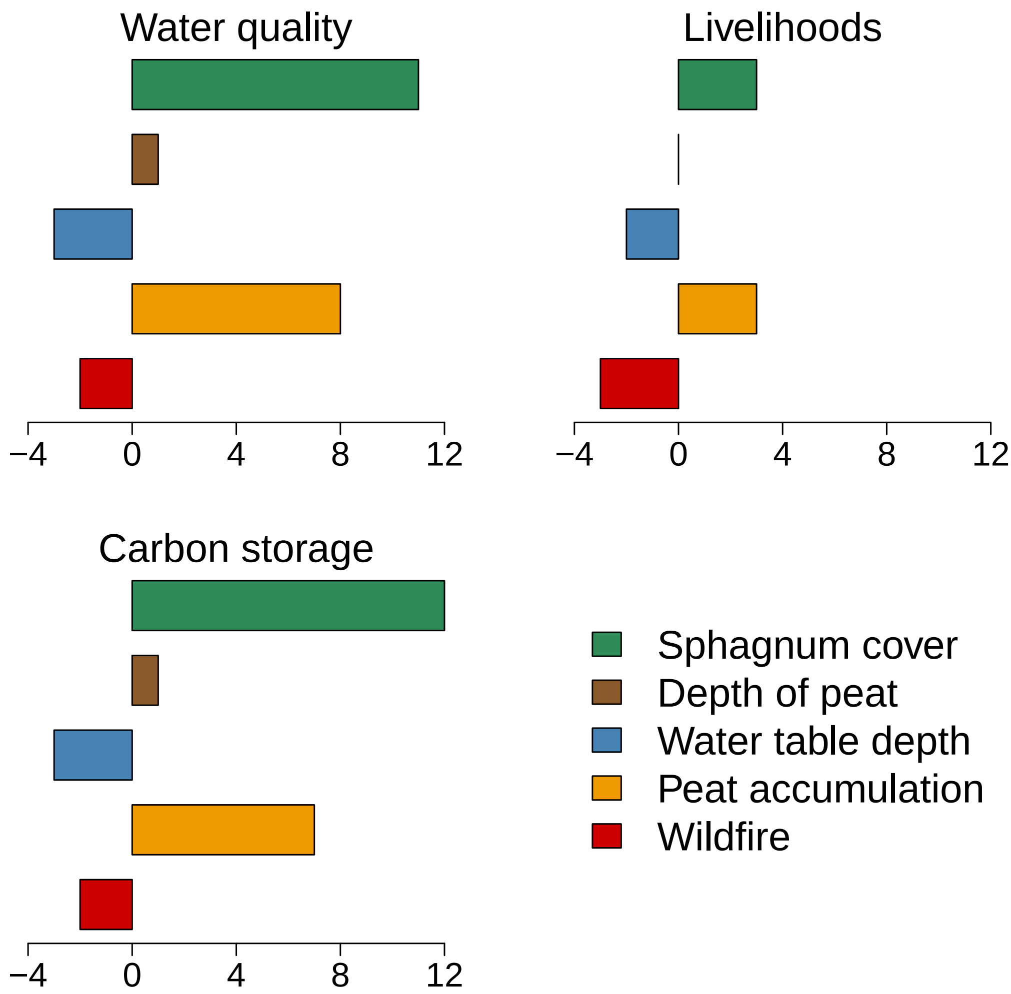


Figure 4.15. Impact of land–use objectives on hubs: blanket peatland network. The change in rank position of hubs for each land use objective compared to the original fuzzy cognitive map rank for non-driver (ordinary) concepts (represented by the 'x' axis). Hubs were defined as concepts with a number of connections ≥ 15. The network comprised highly important concepts only.

water–table depth and wildfire) were reduced (Figure 4.15).

Table 4.6. Blanket peatland network hubs: number of interactions

Hub	Interactions (total)
Sphagnum cover	22
Wildfire	21
Water table depth	21
Depth of peat	15
Peat accumulation	15

Hubs are defined as concepts with ≥ 15 connections

4.6.4 Summary of key findings from this section

The simple model used here produced a set of plausible results that enabled the effect of land–use objectives on carbon storage, and blanket peatland network resilience, to be investigated. I found that; (1) land–use objectives that aimed to increase the quality of water supplied and maintain or increase carbon storage resulted in a relative increase in carbon storage (and peat depth), whilst the livelihoods objective resulted in no change in carbon storage (or peat depth). A number of changes that were made to driver concepts to achieve the livelihood objective increased peat accumulation (such as wildfire reduction). However, an increase or continuation of managed burning, and only partial support for some peatland restoration measures (such as gully blocking) effectively negated any beneficial effect on carbon storage. This result implies that greater support could be given to local livelihoods without increasing carbon losses, but it seems unlikely that that it would be acceptable to all stakeholders, if there is no net benefit to carbon storage. (2) The second key finding was that the proposed changes to driver concepts had a positive or no effect on the hubs that promote resilience of the blanket peatland network, and a negative impact on those that had a detrimental effect. Both results demonstrate that a structural representation of stakeholder mental models, encoded in a network of interactions, can be integrated into a participatory process to help understand the impact of land–use objectives.

4.7 Discussion

My results demonstrate how the structure of a co–developed cognitive network of the interactions between social and ecological concepts, can be used in a participatory process to both propose how to achieve land–use objectives, and assess the impact of those changes on the original model. Using a fuzzy cognitive map of blanket peatlands in the UK, co–developed and validated by stakeholders with differing land–use positions, two classifications of concepts were identified (hubs and drivers). These two classes of concept were used to; (1) focus stakeholder discussions on the changes that should be made to achieve both local and wider societal land–use objectives for blanket peatlands, and (2) assess the impact of the decisions on blanket peatland carbon storage.

4.7.1 Hubs and drivers: two important classes of concept

Connectivity within the blanket peatland network was dominated by a few concepts. These hubs were found to connect a large proportion of the network (five connected to > 60% of concepts), and were key to maintaining the framework of interactions between blanket peatland concepts. A similar result was found by Albert et al. (2000, Figure 1) in a study of complex networks. Once hubs are damaged or removed, perhaps by land–uses, the structure of the network quickly breaks apart. However, hubs included concepts that have a positive impact on carbon accumulation (i.e. mutualistic) such as *Sphagnum* cover, and those that have a negative impact (i.e. antagonistic) such as wildfire: these difference have important implications for the management of driver concepts.

I found that the blanket peatland network was more resilient to the loss of hubs than a number of real–world networks such as the internet (e.g. Cohen et al. 2001). Because the cognitive model was, in fact, a sample of the interactions between peatland concepts, there will be many more interactions, across different scales that were not included by stakeholders who took part in participatory workshops (in common with other interaction networks, e.g. Olito and Fox 2015). Clearly the knowledge base of stakeholders will determine the interactions included (Penn et al. 2013), but because the network is incomplete it is difficult to say whether network connectivity reflects the true distribution of interactions within blanket peatlands, or is a result of the model building process. Nevertheless, some peatlands have been shown to exhibit homeostatic responses to some perturbations (e.g. Swindles et al. 2012), which may be reflected in my results.

Mutualistic hubs imparted both resilience and vulnerability to the structure of the blanket peatland network (Figure 4.6). This apparent paradox occurs because the majority of concepts have few connections, and therefore loss of these concepts has little effect on network structure (Jeong et al. 2000). However the loss of, or damage to, a hub or set of hubs can rapidly break apart the structure (Morone and Makse 2015). This vulnerability can also be exploited as part of land–use strategies by targeting antagonistic hubs to disrupt the part of the structure that causes damage to the peatland (or negatively affects ecosystem function). A similar strategy has been proposed to prevent the spread of epidemics (Newman 2010, page. 614).

Blanket peatland driver concepts had fewer interactions than other concepts (Figure 4.9). These findings agreed with the results of Liu et al. (2011), and provide the second classification of network structure. Drivers can be manipulated to increase those interactions that promote the maintenance or development of mutualistic hubs which are important for blanket peatland resilience. Alternatively, stakeholders could develop strategies to reduce the impact of antagonistic hubs (Figure 4.16). For example, by making changes to driver concepts that reduce the depth of a peatland water table, the negative impact of wildfire (an antagonistic hub) can be reduced (Turetsky et al. 2015). It could be argued that this approach to hubs already takes place as part of blanket peatland management, where prescribed burning of vegetation takes place to in order to prevent wildfire (Allen et al. 2013). However, I used an integrated approach that takes into account how changes affect all included interactions of the peatland. And although the changes made to driver concepts to support local livelihoods produced the greatest reduction in wildfire (Section 4.6), there was no benefit to carbon storage because burning intensity was not reduced, which perpetuates deeper water tables (Holden

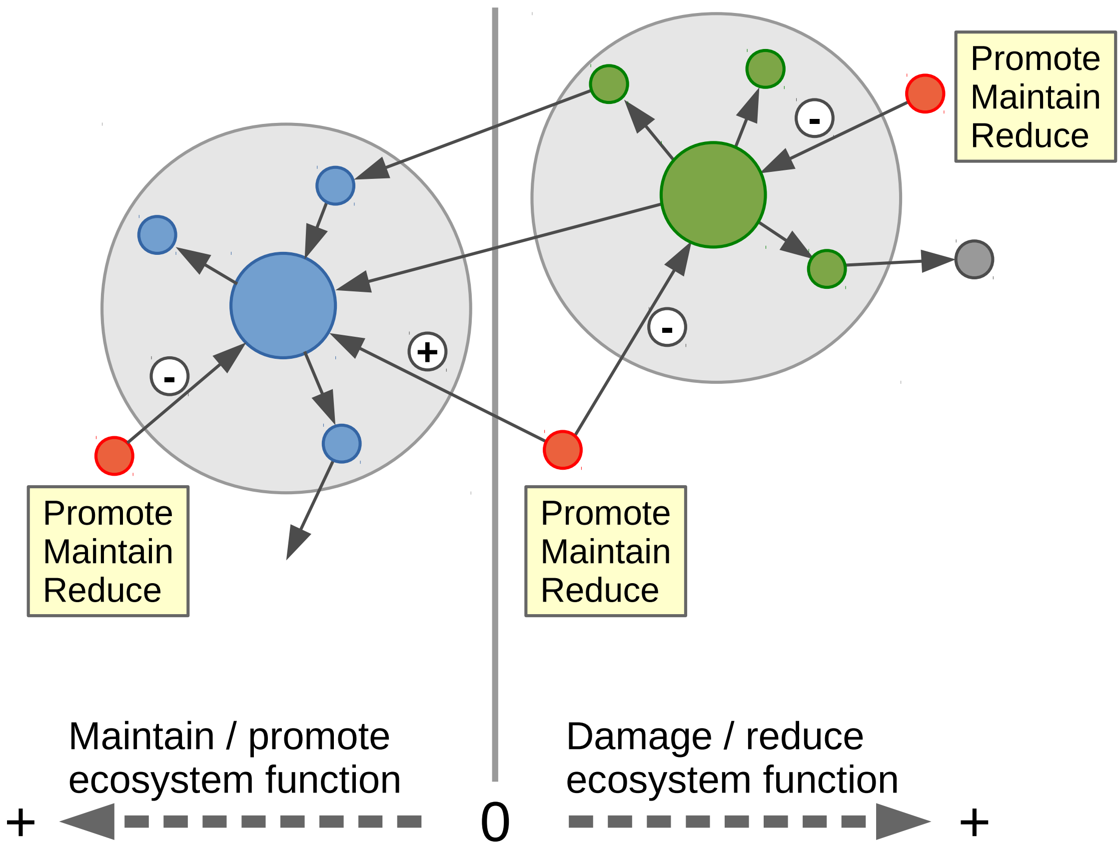


Figure 4.16. Conceptual diagram of the interaction of hubs, drivers and land–use objectives. Large grey circles represent the immediate neighbourhood of hubs. The blue mutualistic hub promotes ecosystem function and the green antagonistic hub reduces ecosystem function. Orange circles represent driver concepts with options to promote, maintain or reduce the effect on hubs. Land uses that result in an increase in the blue (mutualistic) hub will maintain or promote ecosystem resilience, especially in conjunction with actions that reduce the impact of the green (antagonistic hub), and vice versa.

et al. 2015), and also makes the carbon stored in peatlands more vulnerable to wildfire (Turetsky et al. 2015). The actions taken for carbon and water objectives also reduced wildfire (although not to the same extent), as well as increasing carbon storage, predominantly because of actions that reduced water–table depth.

My results suggest that because there are multiple objectives for blanket peatland land use, it is likely that there will need to be a balance between how driver concepts are managed. The most appropriate balance should be determined in decision–making processes, supported by approaches such as the one described here, that take into account knowledge from all relevant sources within the stakeholder community (Maltby 2010; Voinov and Bousquet 2010). It would be relatively straightforward to adjust the values of driver concepts in a workshop setting, so that different combinations could be tested in an iterative process with stakeholders present until the desired impact on carbon storage was achieved. Whilst it is tempting to conclude that refining model outcomes in this way will provide a clear set of actions to follow, the model used here has limitations. All models have limitations, but they should be considered in terms of usefulness (*sensu* Box 1976).

It must be remembered that the changes in concept rank are relative, not absolute, and the proposed increases or decreases in driver concepts are not quantified in any way that can be measured, which must be done through fieldwork and process–based models based on empirical data (e.g. Baird et al. 2011, and explored in more detail in Chapters 5 and 6). The primary objective of the model used here was to provide a framework to bring together different sources of knowledge, create the opportunity to align mental models, and engage stakeholders in a bottom–up process where they could share understanding, and discuss the impact of land–use decisions. The workshop activities and discussions provided insights into how the different driver concepts were perceived in terms of their impact on land–use objectives (e.g. Table 4.12).

4.7.2 Delivering multiple land–use objectives: issues and opportunities

The impact of changes proposed by stakeholders were consistent with the literature reviewed in Chapter 2 Section 2.6, and suggest model outputs were plausible. For example, reductions in water–table depth resulted in increases in *Sphagnum* cover, peat accumulation, and peat depth (Figure 4.15), and these increases were in decreasing order of magnitude (i.e. a large increase in *Sphagnum* cover produced a smaller increase in peat accumulation rates). The results suggest that whilst the objectives for carbon storage and water quality produced the greatest benefit for blanket peat carbon storage, there can be an increase in support for livelihoods without a detrimental effect on current levels of carbon storage. The proposed changes to several driver concepts were of the same or similar magnitude for all land–use objectives (Figure 4.12), which indicates that there are several broad areas of agreement between stakeholders. These results imply that there could be an increased participation in peatland restoration by farmers and gamekeepers. However a single focus on livelihoods would not increase carbon storage and therefore does not produce a win–win situation, but it may be possible for land–use objectives to complement each other. A route to how this might be achieved emerged during a stakeholder workshop.

A review of the changes to driver concepts proposed by stakeholders, and model results, suggests the framework used here has made clear one of the main challenges to combining land–use objectives. Although there are many areas of agreement, some aspects of peatland restoration were not perceived by all to be an opportunity to improve the support for local livelihoods. Gully blocking, increasing the cover of mosses and, to some degree, trapping eroding peat were only partly seen as routes to support local livelihoods, but according to my results (Table 4.5), these driver concepts are likely to result in an increase in the amount of carbon stored. The workshop was not designed to capture the occupation of individual participants in relation to their views and so this information is unavailable. However, it would be helpful to understand if those who favoured peatland restoration (including some land owners) have discounted those in farming or game keeping occupations from restoration activities, or if it is those in farming and game keeping occupations who object to restoration; or a mixture of both. The three driver concepts mentioned are likely to raise water–tables which may be the crux of any objection from game keepers and farmers because of uncertainty of the impact on grouse *Lagopus lagopus* (L.) productivity and sheep *Ovis aries* (L.) health; a concern that has since been raised in other forums (*Peatland restoration - what's in it for me?* 2015).

The majority of stakeholders thought that game keeping should be increased to support local livelihoods, but there was disagreement about burning intensity which most stakeholders thought should be reduced to deliver carbon storage and water quality objectives (Table 4.5). Notably though, a grouse moor land owner proposed that the role of game keepers could be adapted over time to become more involved in activities to restore peatlands and hence become less coupled to managed burning. In addition, a number of workshop participants perceived that more could be done by those leading the restoration of Peak District blanket peatlands to source workers locally rather than from out of area. Whilst no effort was made to validate this comment, there may be an opportunity to create closer working partnerships between restorers and those in farming and game keeping occupations, and to develop shared objectives or to experiment with different treatments based on the local knowledge of farmers and gamekeepers (where SSSI legislation allows). Achieving this result will of course depend largely on the willingness of land owners (both that favour peatland restoration, and those that currently do not) and conservationists to engage in this process against a background of increased managed burning on UK blanket peatlands (Douglas et al. 2015). Notwithstanding these differences, there are clearly opportunities for approaches to land–use objectives to complement each other (also proposed by Reed et al. 2013b), but changes in the mindset of conservationists as well as traditional land managers may be required (Chapter 3). These results highlight the complex nature of the challenge faced by stakeholders when trying to achieve what have been perceived as competing land–use objectives.

McShane et al. (2011) identified a failure to think about the 'hard choices' implicit in any trade-off, and an overemphasis on 'win–win' as one of the key reasons that conflict between local livelihoods and conservation objectives remain unresolved. Redpath et al. (2013) also highlighted the need for trade–offs to make progress where 'win–win' solutions remain elusive. However, Reed et al. (2013b, Appendix 1) identified a set of adaptation strategies, based on ecosystem services, that could reduce the need for trade–offs in the UK uplands. But there is also a need to address the issues between stakeholder groups that lie at the root of the conflict (Redpath et al. 2015) that are often linked to livelihoods (Young et al. 2010), which could include a failure to engage bottom–up with a broad spectrum of knowledge from relevant peatland communities (refer to the discussion in Chapter 3).

The discussions in the stakeholder workshops reported here also indicate that proposals made to achieve wider societal benefits from blanket peatlands would mean adapting current approaches to management and restoration, and need not damage support for local livelihoods in farming and gamekeeping. These adaptations could provide feasible alternatives, which may make the removal of obstacles to progress more likely (Noordwijk et al. 2014). Resolving this aspect of peatland land use is likely to require difficult decisions on the behalf of both the proponents of peatland restoration, and those who are concerned that their livelihoods could be jeopardised. It is essential that land user communities are genuinely involved in decision–making processes about ecosystem land use (Maltby 2010; Young et al. 2016a), and the result of workshop discussion and modelling does suggest one plausible route forward: the next steps are not the domain of models but require stakeholders to continue to work together in what, to say the least, would be challenging meetings; after all, the contestation in peatland land use has all the hallmarks of a 'wicked' problem' (Rittel and Webber 1973).

The decisions made by stakeholders in Section 4.5 are a function of group composition and the context of blanket peatland land–use in the South Pennines (e.g. Voinov and Bousquet 2010). Therefore, the aggregated cognitive model, and modelled outputs, represent some of the range of inputs and outputs that could result from similar workshop discussions, in this and other locations, about future blanket peatland land–use. There are also several suitable alternatives for participatory modelling of complex systems to the ones incorporated into the process reported here (e.g. Hobbs et al. 2002; Ritchey 2013; Bommel et al. 2014; Sedlacko et al. 2014). Nevertheless, the approach used here, builds on previous participatory modelling studies (Penn et al. 2014), by coupling mental models as knowledge networks (Kosko 1988), with the implications of network structure (e.g. Albert et al. 2000; Liu et al. 2011), in a bottom–up collaborative process that could be applied in other social–ecological systems where land use is contested. The flexible and intuitive nature of building, visualising, and working with networks (Gershenson and Niazi 2013; Pocock et al. 2016), means that a wide range of participants can contribute their knowledge of the social and ecological interactions in question and discuss issues of contested land–use (*sensu* Pocock et al. 2016). I propose that it is this simplicity and accessibility that promotes engagement and the discussion that subsequently takes place around how network concepts interact and the implications of network structure, that can facilitate new thinking about how land–use objectives could be achieved, which may make solutions to previous obstacles more likely, and stakeholder meetings more impactful.

Chapter 5

Connecting participatory and process–based models of complex systems

5.1 Chapter summary

I have shown how the co–development of a network from individuals' mental models, and two simple concepts, hubs and drivers, can be used by stakeholders to propose how multiple land–use objectives could be achieved, and to explore the impact of those proposals on carbon storage. The approach described in Chapters 3 and 4 was intuitive and transparent, so that it was relatively simple for stakeholders to participate, share mental models, and incorporate their experiential knowledge into a model of the causal interactions of blanket peatlands. The use of this combined knowledge to guide discussions about land use gave insights into one possible route to address issues of contestation, but the mathematical model used to compare the impact of the proposed changes was based on semi-quantitative data, and its steady–state outputs were relative. Whilst the outputs of the aggregated cognitive network could help shape the direction of land–use decisions, and provide a framework for discussions between stakeholders, in order to quantify the impact of driver concepts, in particular over centennial to millennial timescales, and to enrich the cognitive model, a 2D process–based peat accumulation model was developed. I modified the ecological and hydrological models from the **`DigiBog`** peatland development model (Baird et al. 2011; Morris et al. 2011a) to simulate blanket peat accumulation on slopes and plateaus. The scope of this chapter is outlined in Figure 5.1.

In conjunction with Chapter 6, this chapter addresses research question 4;

What is the predicted impact of social and ecological factors on the centennial to millennial storage of carbon in blanket peatlands when conceptualised as a complex system?

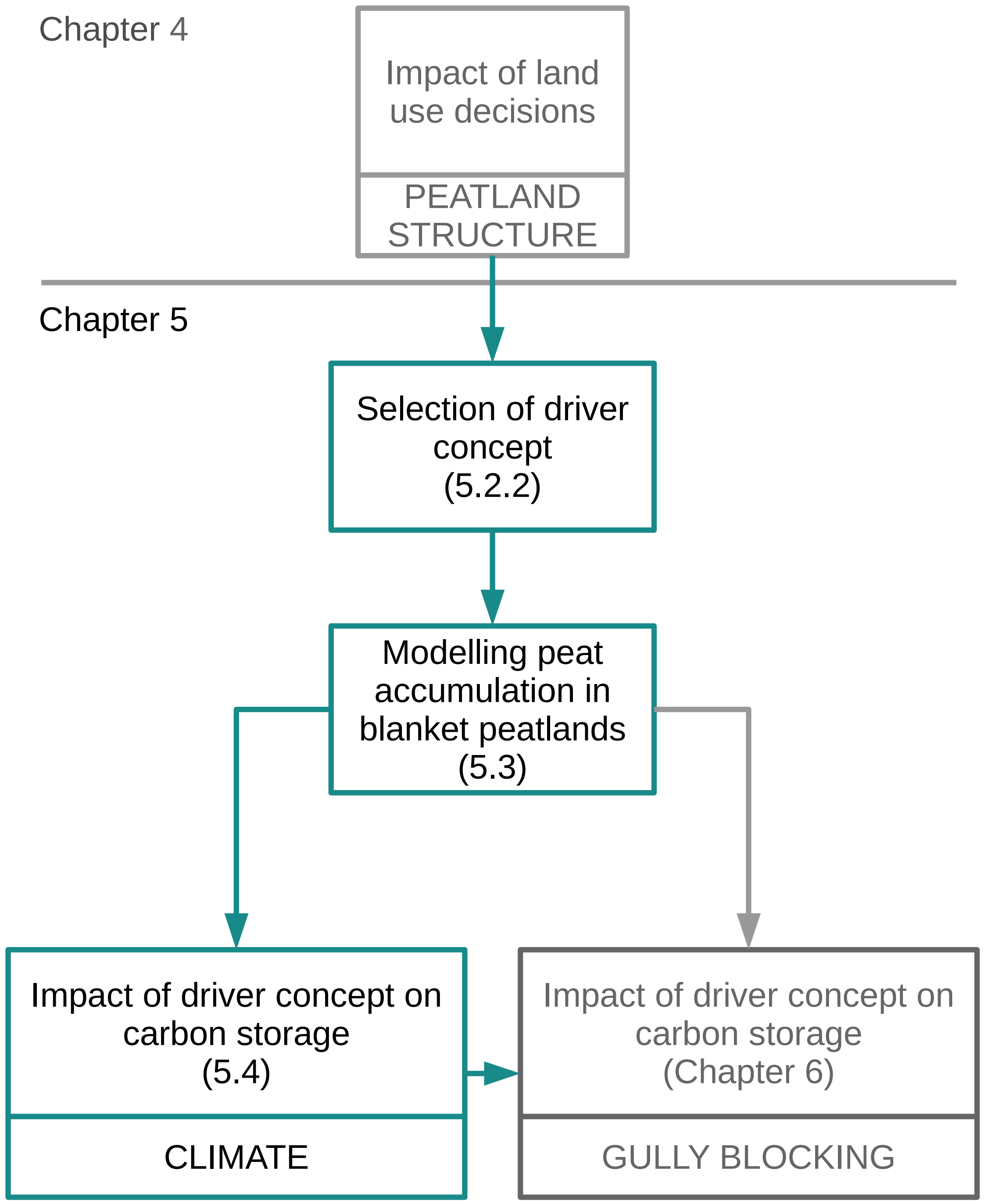


Figure 5.1. Scope of chapter 5. Green boxes represent the scope of this chapter and figures in brackets are chapter or section numbers.

5.2 Introduction

5.2.1 Background: coupling models of complex systems

Process–based models are often integrated into decision–making processes for the management of natural resources: the previous chapter highlighted some of the benefits of including a model based on a wide range of knowledge sources in a participatory process. Voinov and Bousquet (2010) identified two objectives for participatory modelling; (1) to share and increase knowledge of system dynamics, and (2) to support decision making by clarifying the impacts of solutions. But there is a wide choice of model types (e.g. Hobbs et al. 2002; Ramirez et al. 2015; Wood et al. 2015) and a single model may not be sufficient to achieve either of these objectives: for example, if there are gaps in stakeholder understanding of system dynamics, new knowledge may need to be acquired and a different model used for this purpose. Models can be coupled in a nested or iterative approach where a conceptual model of the ecosystem under investigation is used to generate scenarios for future land use and the implications investigated with process–based models (e.g. Walker et al. 2002; Reed et al. 2013b).

I have previously argued that complex systems are difficult to understand and predict and because of this complexity, that modelling is the most suitable method to capture the long–term accumulation of carbon in blanket peatlands. As part of a stakeholder engagement process, Reed et al. (2013b) demonstrated how a number of process–based models can be used to quantify the impact of land use on vegetation, grouse populations, downstream movement of sediment and nutrients, and 10 year carbon fluxes for blanket peatlands, in the Peak District, UK. Models that aim to explain specific aspects of a system (blanket peatlands in this case) have been categorised as minimal models for a system (Evans et al. 2013b). In contrast, synthetic models for a system (also known as tactical models, Evans et al. 2013b) aim to allow the dynamics of a system to develop from the local interaction of resources. Because process–based models enable system behaviour to emerge from local interactions, predictions are not restricted to statistical observations, and different futures can be explored (Evans et al. 2012).

In order to quantify the long–term impact of external forcing on blanket peatland carbon storage, such as from the driver concepts identified in Chapter 4, the internal processes, and external conditions that mediate the accumulation and decomposition of peat need to be suitably represented in a process–based model of peatland development. The challenge, therefore, is to incorporate the key processes that make model outputs plausible when compared to real systems (Grimm et al. 2005; Evans 2012). For example, key peatland processes include; the interaction between decomposition and hydraulic conductivity that can vary in vertical and horizontal space; and the flow of water between neighbouring deposits of peat (Belyea and Baird 2006). Such a model would allow the coupled impacts of future climates and management decisions to be simulated. Following Belyea and Baird (2006), Baird et al. (2011) and Morris et al. (2011a) developed such a complex system model, known as **`DigiBog`**, where virtual peatlands can be 'grown'. As yet blanket peatland simulations (e.g. Heinemeyer et al. 2010) have not included the autogenic mechanisms (such as between litter production, peat decomposition, and water movement) that can mediate how peatlands respond to external forcing which is likely to alter modelled peatland responses. However, although the current version of **`DigiBog`** does include this important feedback mechanism, it needed to be modified so that peat could be accumulated in two or three dimensions on slopes as well as plateaus.

5.2.2 Rationale for selection of driver concept: climate

The discussions and output of the land–use objectives workshop (Chapter 4) were used as the basis for the selection of two driver concepts; climate change and gully blocking. In this chapter, I explore how the temporal resolution of climate variables (rainfall and temperature) affects carbon storage in peatland development models.

Climate change

Climate change was selected as the first driver concept to investigate because it was classified by stakeholders as uncontrollable and, as a result, was excluded from proposals to deliver land–use objectives. Understanding the impact of climate on carbon stored in peatlands and of the reciprocal feedback to the climate system of greenhouse gas emissions, is an issue that affects peatlands across the globe, and is one of the most researched topics in peatland science (e.g. Ise et al. 2008; Fenner and Freeman 2011; Gallego-Sala and Prentice 2012; Charman et al. 2013; Busch et al. 2015; Charman et al. 2015; Witze 2015). The carbon stored in peatlands in many areas of the world may become more vulnerable to oxidation and fluvial losses as a result of climate change (e.g. Li et al. 2015). For example, increased drought is predicted to have a impact on certain terrestrial ecosystems (including forested peatlands) in Asia (IPCC 2014, page 7), Swindles et al. (2015) proposed that warming will cause permafrost peatlands to collapse into Arctic Fens, and the suitable climatic conditions for UK blanket peatlands are predicted to shrink northwards (Gallego-Sala et al. 2010). This increase in vulnerability of carbon stocks includes the blanket peatlands of the South Pennines (Clark et al. 2010). When coupled with the effects of land use, the impact of climate change could be more significant; for example many European and Asian peatlands have been drained for agriculture and forestry, which has increased the vulnerability of carbon stores

(Moore et al. 2013; Turetsky et al. 2015). In the UK, the Committee on Climate Change (2015) has identified an increased risk from climate change to water quality, and carbon storage, predominantly because of the managed burning of vegetation that takes place on blanket peatlands (an activity that has increased in both frequency and extent, Douglas et al. 2015).

Studies of the response of peatlands to climate change are often contradictory (e.g. Laiho 2006; Fenner and Freeman 2011; Wang et al. 2015). And whilst peatland models provide an important way to evaluate the effect on carbon storage of future climate scenarios under controlled input conditions, previous modelling studies have also proposed quite different outcomes. Ise et al. (2008) found that an increase in temperature would result in the loss of significant quantities of carbon, whereas Morris et al. (2011b) predicted a largely homeostatic response to climate forcing. These differences may be due to the treatment of peatlands as a complex adaptive system and the inclusion of cross–scale feedbacks by Morris et al. (2011b), or the instantaneous increase in temperature of Ise et al. (2008) that acted as a shock to shift the peatland to an alternative state (*sensu* Scheffer et al. 2001). More recently, Morris et al. (2015a) in a modelling study of climate signal preservation in raised bogs, found that whilst changes in rainfall caused water tables to respond homeostatically, temperature changes resulted in a permanent alterations to peat structure. A finding supported by Li et al. (2015) who predicted increased blanket peatland erosion was controlled mainly by temperature.

In addition to these differences, existing peatland development models (e.g. Frolking et al. 2010; Morris et al. 2015a) ignore the intra–annual (seasonal) variation of climate variables which could have a significant effect on carbon storage under predicted future climate regimes (Gallego-Sala et al. 2010; Charman et al. 2015), especially for blanket peatlands (Charman 2002). These predictions include more hot and less cold daily temperatures, an increase drought events and summer rainfall, and more high intensity rainfall events (Committee on Climate Change 2015; IPCC 2014; Charman et al. 2015). Some of these predictions such as higher temperatures and increased rainfall, are likely to lead to increased carbon accumulation in higher northern latitudes where peatlands lie within suitable climatic conditions (Charman et al. 2015). Therefore, it could be important to take into account the seasonal distribution of rainfall and temperature in peatland development models, in particular for blanket peatlands where these distributions are thought to be principal factors in peat development (Lindsay et al. 1988; Charman 2002). However, whilst a recent study has investigated the effect of different temporal resolutions of rainfall on the outputs of landscape evolution models (Coulthard and Skinner 2016), the effect of changes in seasonal patterns of temperature and rainfall have not yet been tested in peatland development models.

Peatland models that do not account for seasonal variation in rainfall and temperature may overestimate annual water inputs which in turn affects the balance of plant productivity and

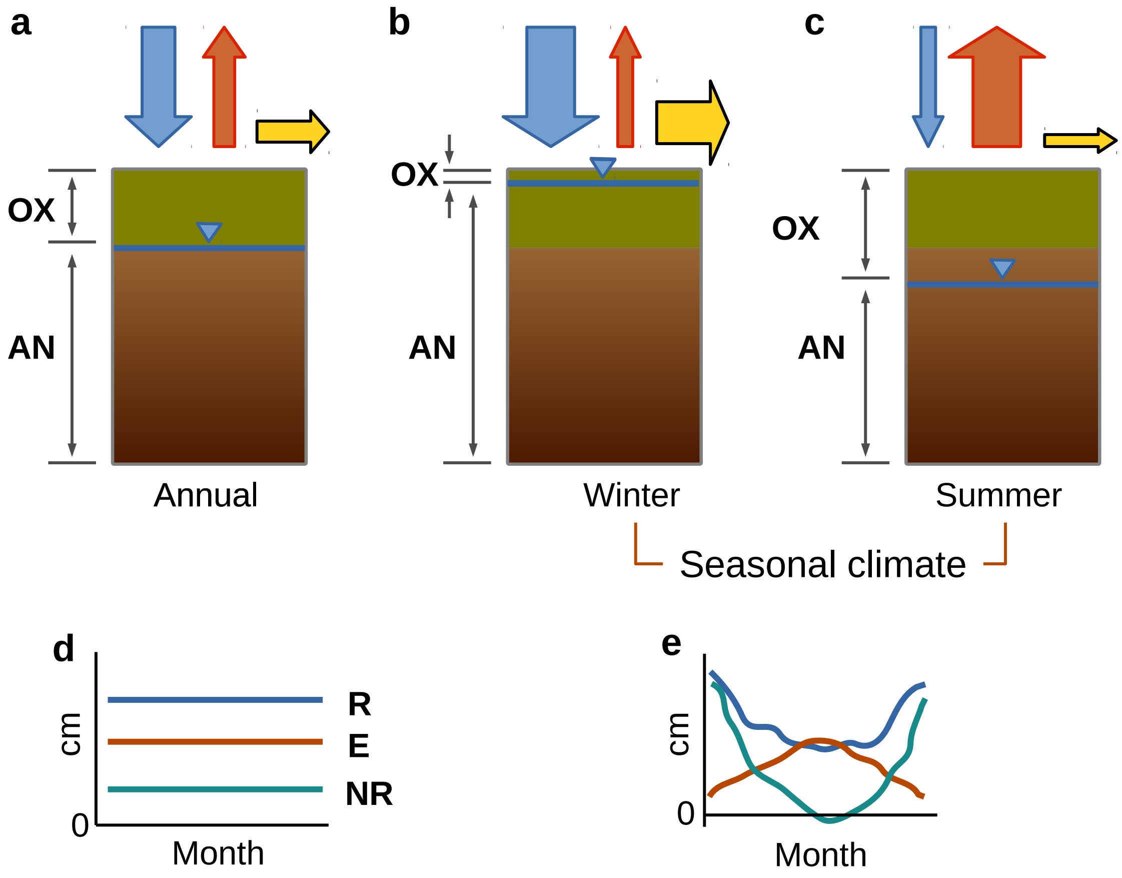


Figure 5.2. Hypothesised effect of a seasonal climate on intra–annual carbon accumulation in a virtual peatland. a – **c** The blue line and inverted triangle represents the water table position, yellow arrows represent overland flow, blue arrows and lines represent rainfall (R), orange arrows and lines represent evapotranspiration (E) and the green line in **d** and **e** represents net rainfall (NR) = rainfall - evapotranspiration. The size of arrows is indicative of magnitude. OX = oxic zone of increased decomposition, AN = anoxic zone where decomposition is several orders of magnitude lower. **a** Annualised weather results in a steady balance of net rainfall. **b** Winter conditions of high rainfall and low evapotranspiration lead to low rates of productivity and decomposition. **c** Higher summer temperatures and lower rainfall lead to lower water tables and increased decomposition which includes peat proper previously in the anoxic zone. **d** and **e** Monthly inputs for R, E and NE under a annualised and seasonal climate regimes.

decomposition. As this balance determines the rate of carbon accumulation, an evaluation of the importance of intra–annual climate variables in peatland development models is needed. Figure 5.2 conceptualises the potential effect of seasonal weather patterns in a peatland development model. When climate variables are annualised (Figure 5.2a), constant levels of net rainfall continue to top–up peatland water tables which is one of the main controls on plant productivity and decomposition. However, the introduction of seasonality will increase the amount of water input in the winter (Figure 5.2b), which may decrease decomposition rates, because of shallower water tables, whilst additional excess rainfall would to be lost through overland flow. In comparison, the effect of annualised inputs is to apportion much of this excess rainfall evenly throughout the year, and so total annual water input to the peatland may be greater (although some net rainfall may still be lost to overland flow). In the summer, (Figure 5.2c) less rainfall and higher evapotranspiration may lead to deeper water tables which could expose previously decomposed peat that, in the winter, resides in the anoxic zone to increased decomposition. The balance of productivity may also change if annual average water tables are deeper. Deeper summer water tables combined with higher temperatures, are also likely to increase rates of decomposition.

As a result, I decided to investigate the effect of different temporal resolutions of rainfall and temperature on modelled blanket peat accumulation over a 5,000 year timescale. The appropriate resolution could then be combined with other driver concepts from the network model and, in future, stakeholders could choose to investigate the coupled effects of different climate regimes and management processes on blanket peatland carbon storage.

5.3 Model and method

5.3.1 Rationale for a new blanket peatland model

The **DigiBog** peatland development model (Baird et al. 2011; Morris et al. 2011a) was introduced to incorporate the important autogenic feedback mechanisms that link hydrological and ecological interactions that mediate the accumulation of peat. The model can be configured to work in over two or three dimensions. Peatland models of undifferentiated peat (0D) or those that vary vertically (1D) (e.g. Clymo et al. 1984; Hilbert et al. 2000; Frolking et al. 2010; Swindles et al. 2012) do not take account of the spatial variation that occurs across peatlands. For example, blanket peatland thickness can vary from a few centimetres to several meters depending on the topography of substrate (e.g Blundell and Holden 2015, Fig. 1). Models that ignore spatial variation may not allow ecosystem dynamics to develop (Baird et al. 2011), which may affect how the system responds to external forcing, and develops self–organised behaviour such as spatial patterning (such as hummocks and hollows in the case of peatlands).

Self–organisation drives spatial heterogeneity because of changes in the flows of energy, nutrients and materials (Levin 1998) which in turn drives ecosystem resilience (Levin 1998; Scheffer et al. 2012; MacDougall et al. 2013). Self–organisation, a characteristic of complex systems, is a result of the interaction between ecosystem components which includes feedback mechanisms that operate across temporal and spatial scales (Belyea and Baird 2006). Feedback mechanisms mediate how peatlands respond to external disturbances such as the climate system and land use (e.g. Morris et al. 2015a). It is also important to recognise that these responses are likely to be non-linear (Belyea 2009; Swindles et al. 2012), and as a result, peatlands may show little change to external disturbances for long periods of time followed by periods of rapid change (Belyea et al. 2004; Holden 2005b; Belyea 2009).

The adaptive response of peatlands, regulated by feedback mechanisms, to external disturbances is not straightforward. Fenner and Freeman (2011) proposed that severe drought will lead to increased carbon losses from UK peatlands, whereas Wang et al. (2015) concluded that for a type of peatland in the USA (pocosin), the same conditions result in an increase in carbon accumulation. Depending on the nature and severity of disturbance, the long–term response of peatlands is likely to be different from that in the short term (Laiho 2006): adaptive mechanisms driven by autogenic negative feedbacks tend to stabilise water tables in the long term and partially decouple peatland water tables from external influence (Swindles et al. 2012; Waddington et al. 2015). However, severe disturbances such as those that have resulted large areas of bare peat and the deeply incised gullys of the South Pennines may result in a shift to a persistently degraded ecosystem state (Scheffer et al. 2001) that requires management intervention to prevent continued loss of stored carbon (Evans et al. 2014).

The complexity of the interaction of peatland feedbacks (Waddington et al. (2015) identified a set of seven hydrological feedbacks) that link together plant litter production, litter and peat decomposition, plant succession, hydrology and peatland size and shape (Baird et al. 2011), provide the rationale for further developing **`DigiBog`** to represent blanket peatlands in two and three dimensions to investigate how they respond to forcing. In the next section I discuss the progress that has been made to achieve this aim.

5.3.2 Model description

`DigiBog` represents a peatland as a set of columns that are horizontally connected by water flow in, up to, four directions and vertically differentiated by individual peat layers (Figure 5.3). **`DigiBog`** comprises a set of submodels that govern plant litter production, water movement, decomposition, and hydraulic properties. The submodels interact bidirectionally with the size and shape of the peatland (Figure 5.4 and Baird et al. 2011). In this version of **`DigiBog`**, layers of peat are added at the end of a (model) year: the thickness of a new layer is determined by the mean water table position for that year, and represents the ecological timestep of the model. During each year, the water table position is established for each column by moving water between columns. Water movement is governed by the hydrological submodel and is based on; (1) the vertical variation of hydraulic conductivity (K) of each layer within a column, (2) the height of the column, and (3) the space available for water (drainable porosity s) within the layer pore space, or in a ponding layer above the peat surface. Next, each layer is decomposed according to the water table position and the oxic or anoxic rate of decomposition (a layer that is partially submerged has the relevant rate applied to the proportions of peat above and below the water table). Water movement occurs over much shorter timesteps than the ecological timestep, and is typically of the order of a few

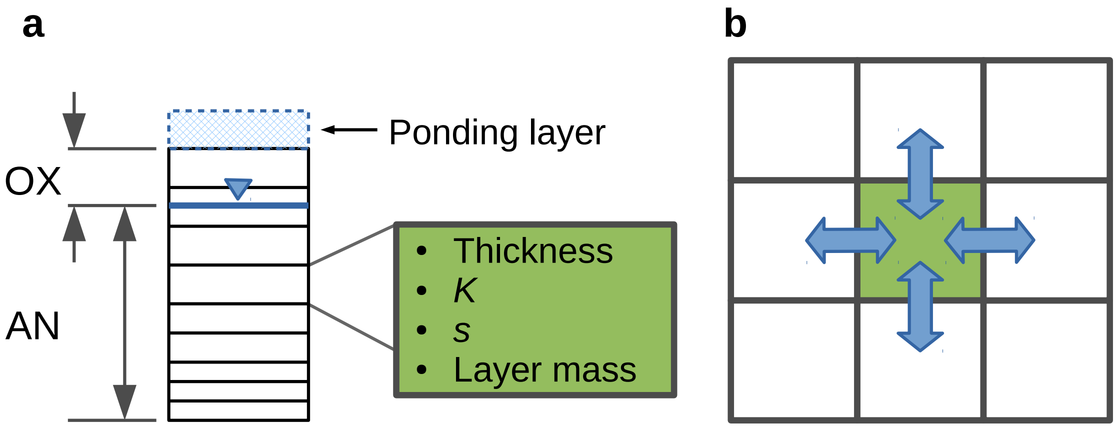


Figure 5.3. `DigiBog` Column and layer structure. a Cross section of a column. Each column is made up of layers of varying thickness. Each layer has a number of properties which include thickness, hydraulic conductivity K, drainable porosity s, and layer mass (original and that remaining after decomposition). The water table position is shown as a blue line with inverted triangle. OX = oxic zone, AN = anoxic zone. The ponding layer enables water to rise above the surface of the peatland. **b** Plan view of water movement into or out from a column.

minutes to ensure that the hydrological submodel remains stable: the hydrological submodel is nested within the ecological submodel (Baird et al. 2011).

Previous versions of **`DigiBog`** have been implemented in 1D (Swindles et al. 2012) and 2D (Morris et al. 2011a). Clearly in light of the previous discussion, and to take into account the hillslopes on which blanket peatlands form, it is the 2D version that is of interest here. The version demonstrated by Morris et al. (2011a) differs from the implementation described below in a number of ways: (1) ecological timesteps were set at 10 years; (2) each column measured 10 m $\times$ 10 m (although the effect of smaller column size on peatland shape was also investiagted); (3) the hydrological submodel was run until a steady state was reached (500 days) before being used in the other submodels (Baird et al. 2011); (4) layer decomposition was based solely on water table position; (5) plant litter production for the new peat layer was provided for a mixture of vascular plants and moss species determined by Belyea and Clymo (2001) but was not temperature dependent; and (6) net–rainfall and temperature were read into the model on an annual basis.

In the version of **`DigiBog`** described here, the following changes were made: (1) ecological timesteps took place on an annual basis, (2) each column measured 2 m $\times$ 2 m, (3) the main **`DigiBog`** program was re-written to enable the output of the hydrological submodel to be used to update other submodels during each hydrological timestep (i.e. every few minutes), so that water tables were transient and not stable. Since the original model was published, the authors have extended layer decomposition (4 above) and plant litter productivity (5 above) to include the effect of temperature which form new decomposition and plant litter production submodels (described in Morris et al. 2015a, supplementary material), both of which were included in this version of

DigiBog. (6) For intra–annual climate simulations, net rainfall and temperature were read into the model on a monthly or weekly basis, and decomposition parameters recalculated for the relevant time period.

Plant litter production for the new peat layer is now given by,

$$p = 0.001 \times \left[9.3 + 133z - 0.022 \times (100z)^2\right]^2 \times f(T_{ann}), \tag{5.1}$$

where p is the new annual mass of litter (kg m^{-2} year^{-1}, later converted to thickness) to be added to a column, and z is the mean annual water–table depth calculated from $B - h$ where B is the height of a column's peat surface above the impermeable base and h is the height of the water table; $p = 0$ for depths below 0.668 m and $f(T_{ann})$ is the function $(0.069 \times T_{ann} + 0.004)/(0.069 \times T_{\mathrm{BC}})$ which simplifies to $0.1575 \times T_{ann} + 0.0091$, and where T_{ann} is the mean annual temperature.

The oxic and anoxic decomposition parameters now incorporate the effect of temperature as follows;

$$\begin{aligned} \alpha_{\mathrm{OX},\bar{\mathrm{T}}} &= \alpha_{\mathrm{OX},\mathrm{T_{BC}}} \times Q_{10}^{\left(\bar{T}-T_{\mathrm{BC}}\right)/10} \\ \alpha_{\mathrm{AN},\bar{\mathrm{T}}} &= \alpha_{\mathrm{AN},\mathrm{T_{BC}}} \times Q_{10}^{\left(\bar{T}-T_{\mathrm{BC}}\right)/10}, \end{aligned} \tag{5.2}$$

where $\alpha_{\mathrm{OX},\bar{\mathrm{T}}}$ and $\alpha_{\mathrm{AN},\bar{\mathrm{T}}}$ are the oxic and anoxic annual decomposition parameters applied to the mean annual air temperature $\bar{\mathrm{T}}$ for all simulations; $\alpha_{\mathrm{OX},\mathrm{T_{BC}}}$ and $\alpha_{\mathrm{AN},\mathrm{T_{BC}}}$ are the annual decomposition parameters applied to a mean air temperature T_{BC} (6.29 °C), measured by Belyea and Clymo (2001) and used as a baseline to define the temperature dependency of decomposition; and Q_{10} is the rate of change for a 10 °C increase in temperature (Morris et al. 2015a).

Layer decomposition includes the effect of temperature (Equation 5.2), water table position and the effect of recalcitrance on increasingly decomposed older peat (e.g. Clymo et al. 1984). Recalcitrance was incorporated by the multiplication of the anoxic decomposition rate by proportional mass remaining θ:

$$lm^{t+1} = \left[lm^t \times \mathrm{OX} \times e^{-\Delta t \alpha_{\mathrm{OX},\bar{\mathrm{T}}}}\right] + \left[lm^t \times 1 - \mathrm{OX} \times e^{-\Delta t \theta \alpha_{\mathrm{AN},\bar{\mathrm{T}}}}\right], \tag{5.3}$$

where lm^{t+1} is a layer's new mass after decomposition, lm^t is the current layer mass, OX is the proportion of the layer above the water table, $1 - \mathrm{OX}$ is the proportion of the layer below the water table, Δt is the model timestep and the terms $\alpha_{\mathrm{OX},\bar{\mathrm{T}}}$ and $\alpha_{\mathrm{AN},\bar{\mathrm{T}}}$ are derived from the new decomposition submodel given by Equation 5.2. The original mass of each layer added to a peat column is tracked so that the proportion of mass remaining can be calculated, $\theta = lm_{t+1}/lm^{orig}$.

In addition, a thin layer (0.02 m) of mineral soil with hydraulic conductivity of 1×10^{-5} m s^{-1} was added between the base substrate and the initial peat layer: water movement occured in this layer but no decomposition took place. The ponding mechanism present in the hydrological model (Baird et al. 2011), but not used by Morris et al. (2011a), was activated to a depth of 2.5×10^{-3} m for all columns.

Five equations are used in **`DigiBog`** which, including Equations 5.1, 5.2 and 5.3, govern plant litter production, decomposition rates, the layer mass after decomposition, water movement and the feedback between K and decomposition (Figure 5.4). The remaining equations used by Baird et al. (2011) are shown here for completeness. The set of parameters and their units used in the simulations for this study are detailed later in this chapter. Water movement between columns is described in the hydrological model by a solution of,

$$\frac{\partial h}{\partial t} = \frac{\partial}{\partial x}\left(\frac{K(d)}{s(d)} d \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(\frac{K(d)}{s(d)} d \frac{\partial h}{\partial y}\right) + \frac{P(t) - E(h,t)}{s(d)}, \tag{5.4}$$

where h represents water–table height, t is time, x and y are column dimensions, d is the height of the water table above the impermeable base, K is the depth–averaged hydraulic conductivity below the water table, P represents the addition of rainfall during a timestep, E represents water loss through evapotranspiration during a timestep t, and s is drainable porosity.

The feedback between layer K and decomposition is recalculated based on the mass of the newly decomposed layer;

$$K = \alpha e^{b\theta}, \tag{5.5}$$

where α and b are parameters chosen to represent the decline in hydraulic conductivity as a function of the proportion of layer mass that remains θ following decomposition.

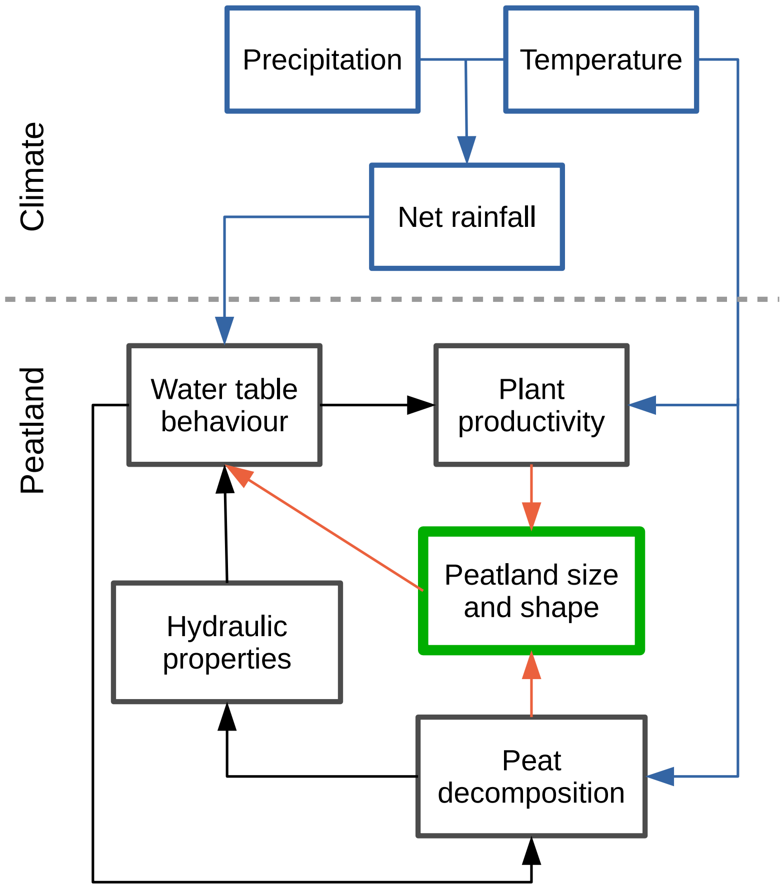


Figure 5.4. Conceptual model of the version of `DigiBog` used for simulations. Blue boxes represent climate inputs and grey boxes represent model algorithms. The green box represents peatland development and orange arrows are cross–scale feedbacks. The dashed grey line represent the climate–peatland boundary. Adapted from Baird et al. (2011) and Morris et al. (2015a).

5.3.3 Model setup: peatland size and shape

This version of **`DigiBog`** was configured as a transect of 25×2 m columns of peat draining into a stream. Boundary cells are used to specify how water exits the model. A von Neumann zero–flow condition was used to define a drainage divide at the top of the slope and to define the edges of the transect. A Dirichlet (set water level) condition was used to allow water to flow out of the model at the bottom of the hillslope (Figure 5.5), and was set to the combined height of the mineral and first peat layer in each column ($\approx 2.086 \times 10^{-2}$ m). At the start of each model run, the water table was set to the top of the first layer of peat added to each column. Although blanket peatland can form on all but the steepest slopes (Charman 2002), peat coverage is likely to be thin and therefore to enable deeper peat to develop, simulations were carried out on a constant slope of 3° (Figure 5.5). To compare how peat accumulated on hillslopes to accumulation on plateaus, a selection of simulations were implemented with a flat base. Sections of a hillslope that include flat and sloping terrain in combination with features such as basins (e.g. Tipping 2008, Fig. 4) are envisaged in future simulations.

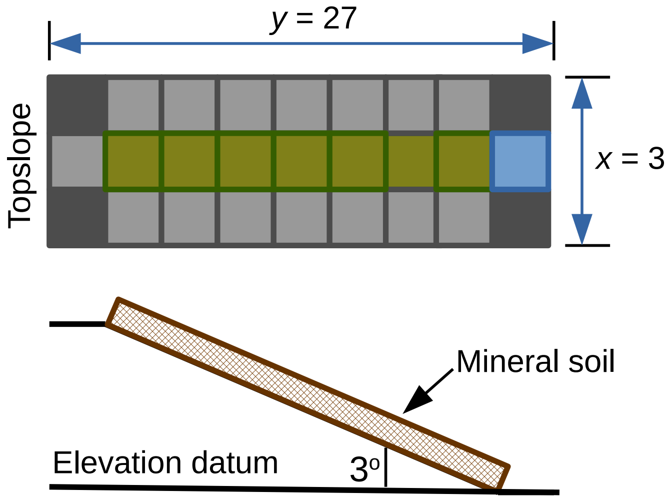


Figure 5.5. Schematic of the `DigiBog` setup used for simulations. Top; green cells represent active columns of peat (25 × 2 m), dark grey cells are boundary columns that are switched off, light grey cells are specified as a zero–flow von Neumann condition and the light blue cell is specified as a Dirichlet condition. Bottom; sloping models were configured with a shallow slope of 3 °. All model runs used a 0.02 m thick mineral layer.

5.3.4 Model parameters: initial peat properties

In Section 5.2.2, I hypothesised that the introduction of an intra–annual climate may affect the accumulation of carbon because changes to the distribution of net–rainfall and temperature are likely to affect both plant productivity and decomposition. As both the new productivity and decomposition functions are sensitive to temperature, the rate of decay and water–table depth, model parameters for oxic decay α_{OX} and Q_{10} were chosen to be manipulated (Table 5.1). A set of default values were selected for the remaining input parameters which were used for all simulations; (1) model run time, (2) the anoxic decay parameter α_{AN}, (3) bulk density ρ, (4) drainable porosity s, and (5) the initial saturated hydraulic conductivity K of a new peat layer. The rate of anoxic decay was kept constant so that the effect of water–table behaviour and the exposure of previously submerged peat to oxic conditions could be explored. All parameters were chosen to be plausible when compared to measured data (discussed below) and were applied to the three sets of climate inputs (annual, monthly, weekly).

Default parameters for all simulations

The default values used for parameters 1–5 were as follows; (1) simulations ran for 5,000 years, which was the approximate age of South Pennine peats proposed by Moore (1975); (2) the base rate for the proportion of peat mass lost through anoxic decay α_{AN} was set to 10^{-5} year^{-1} (Morris et al. 2011b). Currently **`DigiBog`**, treats bulk density ρ and drainable porosity s as constant throughout a peat column (Baird et al. 2011). (3) Bulk density was set at 100 kg m^{-3} , which was comparable

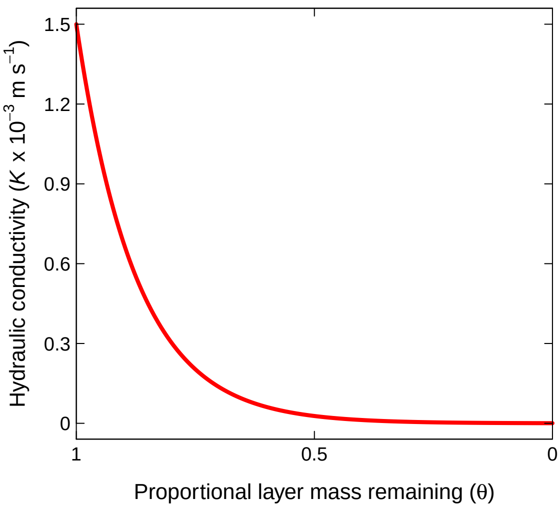


Figure 5.6. Feedback between hydraulic conductivity and decomposition.

to the range of bulk density reported by Wallage and Holden (2011, Table 1), for an intact area of blanket peat, of 95–121 kg m^{-3}, mean $= 108$ kg m^{-3}. (4) Holden et al. (2001) reported $s = 0.35$ at depths of 0.10–0.25 m and $s = 0.55$ above that: because s is non–varying I assumed a value of 0.3 (unitless), identical to that of Morris et al. (2011a,b). (5) The saturated hydraulic conductivity of the first and newly added peat layers was set to 1.5×10^{-3} m s^{-1} ($\alpha = 15.87, b = 8.0$, Equation 5.5) which thereafter declined with increasing decomposition (Figure 5.6). A recent study of water movement in a blanket peatland by Turner et al. (in preparation) suggests this is a suitable value for near–surface hydraulic conductivity.

Manipulated parameters

Three values were chosen for both the oxic decay parameter (α_{OX}), and Q_{10} for hillslope models: to reduce the number of simulations, two values were used for each plateau model (Table 5.1). As both α_{OX} and Q_{10} of have been shown to vary widely (Baird et al. 2011; Kleinen et al. 2012), selection was based upon a set of values that appeared credible in relation to reported measurements. Clymo et al. (1984) estimated oxic decay rates of 1.5×10^{-2} and Hogg (1993) reported rates of 2×10^{-2} – 5×10^{-2} year^{-1} for *Sphagnum* mosses in hummocks and hollows by analysing peat cores. Although higher rates of oxic decay have been reported for different *Sphagnum* species from litter bag experiments (e.g. Belyea 1996), I chose α_{OX} parameters consistent with Morris et al. (2011b) and within the above reported ranges of 2.5×10^{-2} year^{-1}, 3.5×10^{-2}, and 4.5×10^{-2} year^{-1}.

Values of Q_{10} vary according to peatland process: Helfter et al. (2015, supplementary material) reported a median Q_{10} of 3.36 ($n = 11$) for a UK blanket peatland, Freeman et al. (2001) observed values of 1.33 and 1.72 for UK upland peat soils (presumably mainly blanket peat), whilst values of between 3.1 and 4.4 have been reported for different peatland types (Hogg 1993; Dioumaeva et al. 2003). Based on these values, I assumed values of 2 and 3 for both oxic and anoxic Q_{10}.

Table 5.1. DigiBog intra–annual climate simulations

Peatland base	Climate	Q_{10}	Oxic decay (α_{ox} year^{-1})
Hillslope	Annual	2	0.045
Hillslope	Annual	2	0.035
Hillslope	Annual	2	0.025
Hillslope	Annual	3	0.045
Hillslope	Annual	3	0.035
Hillslope	Annual	3	0.025
Hillslope	Monthly	2	0.045
Hillslope	Monthly	2	0.035
Hillslope	Monthly	2	0.025
Hillslope	Monthly	3	0.045
Hillslope	Monthly	3	0.035
Hillslope	Monthly	3	0.025
Hillslope	Weekly	2	0.045
Hillslope	Weekly	2	0.035
Hillslope	Weekly	2	0.025
Hillslope	Weekly	3	0.045
Hillslope	Weekly	3	0.035
Hillslope	Weekly	3	0.025
Plateau	Annual	2	0.045
Plateau	Annual	2	0.025
Plateau	Monthly	2	0.045
Plateau	Monthly	2	0.025
Plateau	Weekly	2	0.045
Plateau	Weekly	2	0.025

Q_{10} values were applied to oxic and anoxic decomposition

5.3.5 Climate data

I investigated the effect of intra–annual climate inputs on peat development of annual (i.e. constant), monthly, and weekly weather inputs based on four years (2010–2013) of precipitation and temperature records from Keighley Moor, a blanket peatland in the north of England ($53^\circ\ 85'\ 31''$ N, $-02^\circ\ 02'\ 13''$ E), where a recent survey has recorded peat depths of up to 3.7 m (Blundell and

Holden 2015). Climate data from the University of Leeds weather station on Keighley Moor (2010–2013) suggests mean annual rainfall and temperature of 1.15 m and 7.57 °C respectively (temperature was recorded every 10 minutes at a height of 1 m: daily temperature was calculated from the mean of the maximum and minimum values. Rainfall data was collected using a tipping bucket rain gauge). Site vegetation comprises *Calluna vulgaris* (heather), *Eriophorum vaginatum* (hares tail cotton grass), *Eriophorum angustifolium* (common cotton grass) and *Vaccinium myrtillus* (billberry). *Sphagnum* mosses are less well represented but include *S. fallax, S. capillifolium, S. fimbriatum* and *S. cuspidatum.* A complete description of past and present vegetation and management of Keighley Moor is given in a palaeoecological study by Blundell and Holden (2015).

The climate variables used in **`DigiBog`** are net rainfall U (i.e. precipitation P minus potential evapotranspiration E), and temperature. Potential evapotranspiration was calculated using the Thornthwaite method (described in Palmer and Havens 1958), which is based on precipitation (cm), mean monthly temperature (°C), daylength (hours) according to site location, number of days in a month, and a monthly index $i = (T_{ave}/5)^{1.514}$, where T_{ave} is the mean monthly temperature. The result was a set of monthly values for potential evapotranspiration which were summed to give an annual value. Weekly values were calculated by apportioning the annual potential evapotranspiration to each week according to the sum of daylengths for that week (Figure 5.7). Net rainfall addition (removal) U from the virtual peatland was calculated from $U = P - E$.

Climate variables interact with the virtual peatland in two ways: firstly net rainfall adds or removes water (depending if potential evapotranspiration is greater than precipitation) which affects the position of the water table. Secondly, temperature is used to calculate the rate of plant litter production and also to determine the oxic and anoxic decay parameters used to decompose peat layers: both litter production and decomposition are also dependent on water table position. These processes take place over over two different time loops; annual and sub–annual. New litter addition and the properties of the new peat layer are set annually (Equation 5.1), whereas decomposition (Equation 5.3), water movement (Equation 5.4), and the feedback between layer K and decomposition (Equation 5.5) take place sub–annually. Because plant litter production takes place on an annual basis, the annual mean temperature is used, whereas the temperature used to determine decomposition varies monthly or weekly. Sub–annual timesteps can be fixed, at say 5 minutes, and in a model where climate varies only inter–annually, it is relatively simple to keep track of the sum of sub–annual timesteps: when the value of one year is reached the next year's net rainfall and temperature values can be read into the model. At this stage the new annual oxic and anoxic decay parameters are recalculated based on the new temperature value. Obviously when climate varies intra–annually these values need to be read in at the appropriate interval (monthly or weekly in this case).

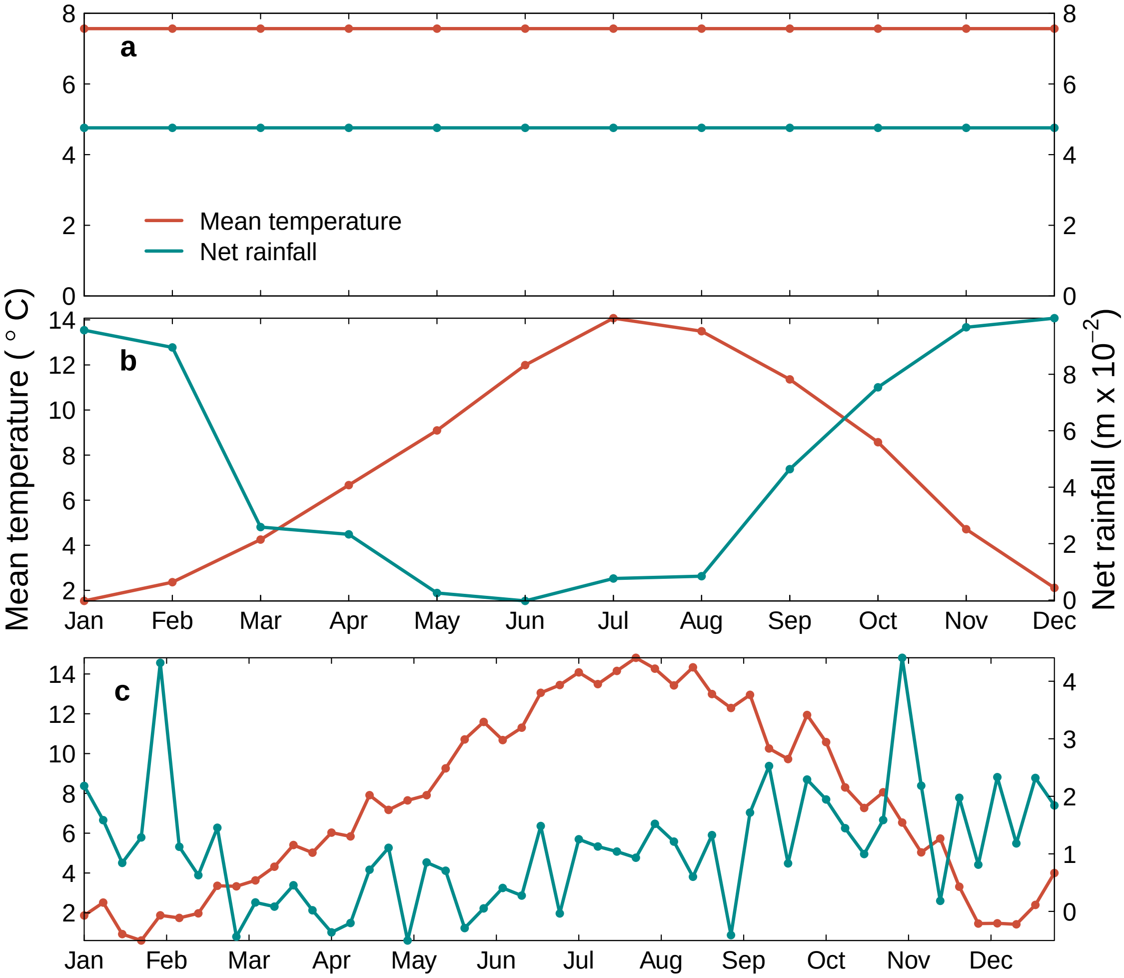


Figure 5.7. Intra–annual climate data for simulations. Keighley Moor mean temperature and net–rainfall (precipitation minus potential evapotranspiration) data from 2010–2013 for **a** annual, **b** monthly and **c**, weekly time periods.

5.3.6 Time management

Each month or week comprises a number of sub–annual timesteps; during a timestep, the water table position is calculated for each column of peat and all layers are decomposed. Using a fixed timestep of a few minutes for a complete simulation would be computationally expensive and so to speed up model execution, variable sub–annual timesteps were calculated according to Baird et al. (1998),

$$\frac{Kh^{max}\Delta t}{s(\Delta x)^2} \leq 0.5, \tag{5.6}$$

where Kh^{max} is the maximum transmissivity of all peat columns, Δt is the calculated timestep, Δx is the size of a peat column (2 m in this case) and s is the drainable porosity applied to all peat layers. To improve the numerical stability of the hydrological submodel in the first few years of a simulation, timesteps were set to 5 minutes for years 1–50 and afterwards allowed to vary up to 50 minutes.

Due to the introduction of variable timesteps, two time management routines were implemented to read in climate data at the appropriate interval. One routine was used for both the annual and monthly modules: the same values for temperature and net rainfall were used for each month for the annual climate. And a second routine was used for weekly climate data. In the same way as for fixed timesteps, these routines keep track of sub–annual timesteps but because timesteps vary, the final timestep of the period is set to ensure time is not under– or over–accumulated.

5.4 Results

The results of intra–annual climate simulations are discussed in two stages. Firstly the three parameter sets for sloping peatlands were combined to compare peat accumulation between annual, monthly and weekly intra–annual climate variables (i.e. nine simulations, Table 5.2). A comparison was also made between peat accumulation on hillslopes and plateaus (Table 5.2). Secondly, the dynamics of peatland development between climates is explored for one parameter set (i.e. three simulations). All simulations parameterised with an oxic decay rate of 2.5×10^{-2} year^{-1} produced unusual peatland surfaces which may represent the development of hummocks and hollows but requires further investigation beyond the scope of this thesis. Therefore the parameter combinations discussed here were chosen to represent peatlands with roughly uniform surfaces, which are often seen on blanket peatlands. Because of the time taken for each model run, the results presented here are a limited test of the sensitivity of the model to different representations of climate variables, and the accumulation of peat in the modelled peatland. All results for sloping peatlands are plotted without the slope, unless otherwise indicated, so that differences in height between the toe– and topslopes do not dominate 'y' axis scaling.

Table 5.2. Model results reported for climate resolution simulations

Peatland base	Simulation ID	Climate	Q_{10}	α_{ox} year^{-1}
Hillslope	H1, H2, H3	a, m, w	2	0.045
Hillslope	H4, H5, H6	a, m, w	3	0.045
Hillslope*	H7, H8, H9	a, m, w	3	0.035
Plateau	P1, P2, P3	a, m, w	2	0.045

a = annual, m = monthly, w = weekly

* = parameter set explored in greater detail

5.4.1 *The effect of climate representation on peat accumulation*

At the end of 5,000 model years, the intra–annual climate simulations show that for for both hillslope and plateau models: (1) net peat accumulation was greatest when climate is annualised; (2) the lowest rates of net accumulation occur when a monthly climate was used; and (3) weekly climate inputs result in the accumulation of more peat than monthly inputs and less than annual inputs (Table 5.3 and Figure 5.8). The mean peat depth for hillslope models developed with the annual climate was 1.7 m and 1.2 m greater than that for monthly and weekly climates respectively. When comparing plateau models, the mean peat depth for annual climate inputs was 1.2 m and 1.1 m greater than monthly and weekly inputs. For hillslope models, weekly climate inputs resulted in a mean peat depth that was approximately 0.5 m greater than the monthly climate, but this difference was reduced to less than 0.1 m for the plateau model (Table 5.3).

The effect of decomposition parameters (i.e. oxic decomposition and Q_{10}) on peat accumulation is reported here for hillslope models only. Increasing the temperature sensitivity of peat decomposition by changing Q_{10} from 2 to 3 for both oxic and anoxic decay parameters (i.e. from a doubling to tripling with every 10°C rise in temperature), whilst maintaining an oxic decay parameter of 4.5×10^{-2} year^{-1}, resulted in a small reduction in mean peat depth of 0.12 m for annual climate inputs, 0.17 m for the monthly inputs and 0.13 m for weekly inputs. Decreasing the oxic decay rate to 3.5×10^{-2} year^{-1} whilst maintaining a value for Q_{10} of 3, resulted in an increase in mean peat depth for annual inputs from 2.17 m to 3.35 m (an increase of 1.18 m), whilst there were smaller increases for both monthly and weekly inputs of 0.19 m and 0.33 m respectively. With this parameter set, the annualised climate simulation accumulated a maximum peat depth that was 2.37 m and 1.77 m greater than monthly and weekly climate inputs respectively. Interestingly, the difference in peat accumulation for monthly inputs was very small (Δh = 0.02 m) for parameter combinations $Q_{10} = 2, \alpha_{\mathrm{OX}} = 4.5 \times 10^{-2}$ year^{-1} and $Q_{10} = 3, \alpha_{\mathrm{OX}} = 3.5 \times 10^{-2}$ year^{-1}, (Figure 5.8a), but increased for weekly (Δh = 0.2 m), and annual (Δh = 1.08 m) climate inputs.

Table 5.3. Final accumulated peat depths for simulations using different climate inputs

Climate	Hillslope		Plateau	
	Mean depth (m)	Max depth (m)	Mean depth (m)	Max depth (m)
Annual	2.60	3.67	2.59	2.90
Monthly	0.91	1.15	1.40	1.63
Weekly	1.41	1.79	1.54	1.78

Values for hillslopes are based on all columns from three simulations for each climate, whilst values for plateaus are based on all columns from a single simulation for each climate.

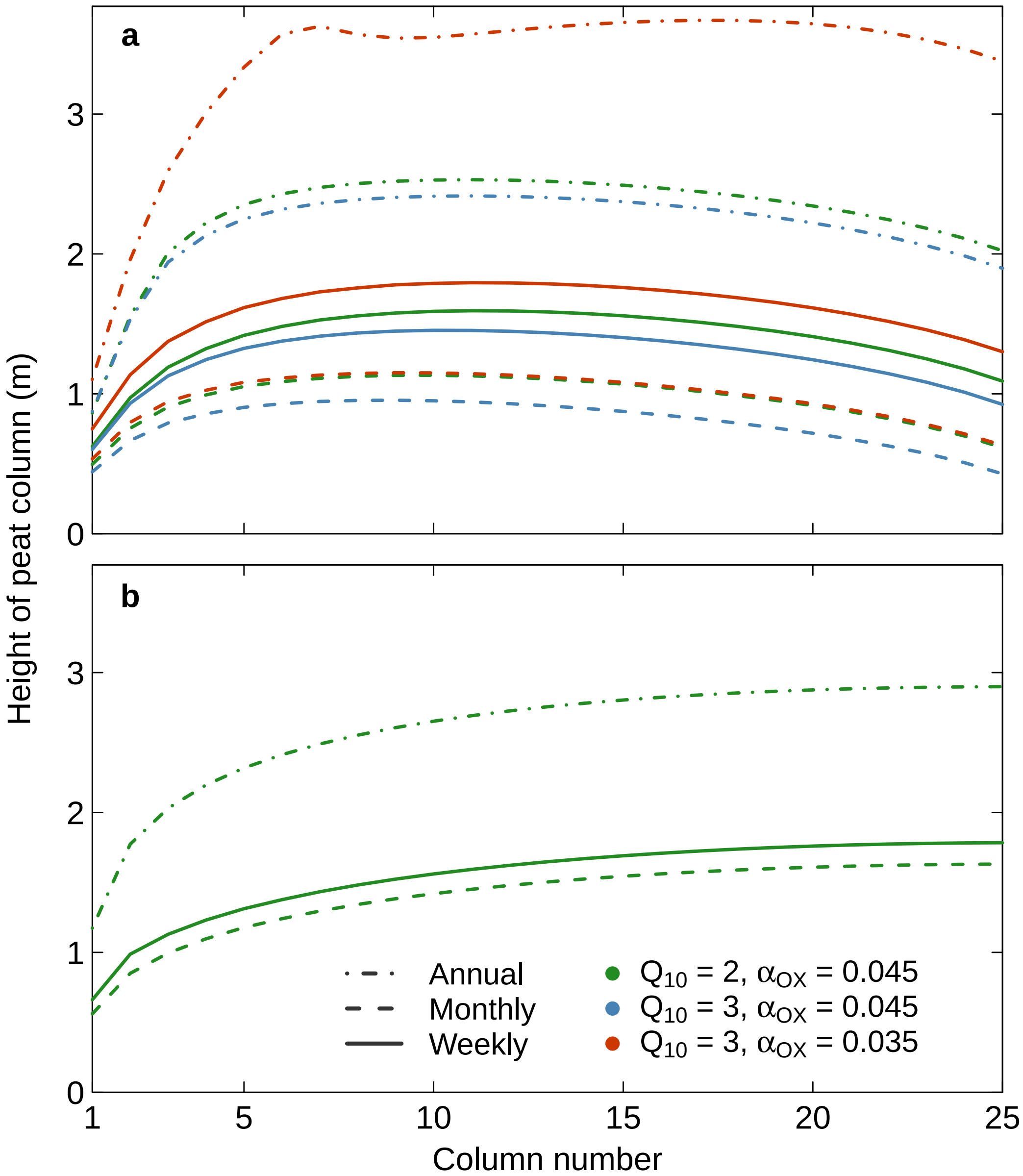


Figure 5.8. Effect of intra–annual climate on peat accumulation. Lines represent the height of the virtual peatland surface after 5,000 years. Peatland development was simulated on, **a** a 3 ° slope (column 1 = toeslope); and **b** a plateau (column 1 = plateau edge).

5.4.2 Oxic zone thickness

The mean thickness of the oxic zone followed the same pattern as peat depth in relation to climate inputs and decomposition rate (Figure 5.9). For all simulations, the parameter set $Q_{10} = 3, \alpha_{\mathrm{ox}} = 4.5 \times 10^{-2}$ year^{-1} resulted in the thinnest mean annual oxic zones and $Q_{10} = 3, \alpha_{\mathrm{ox}} = 3.5 \times 10^{-2}$ year^{-1} results in the thickest. Mean annual oxic zone thickness for annualised climate inputs varied from 0.042 m (mean peat thickness = 2.17 m) to 0.22 m (mean peat thickness = 3.35 m). Monthly climate input simulations resulted in mean oxic zone thicknesses of between

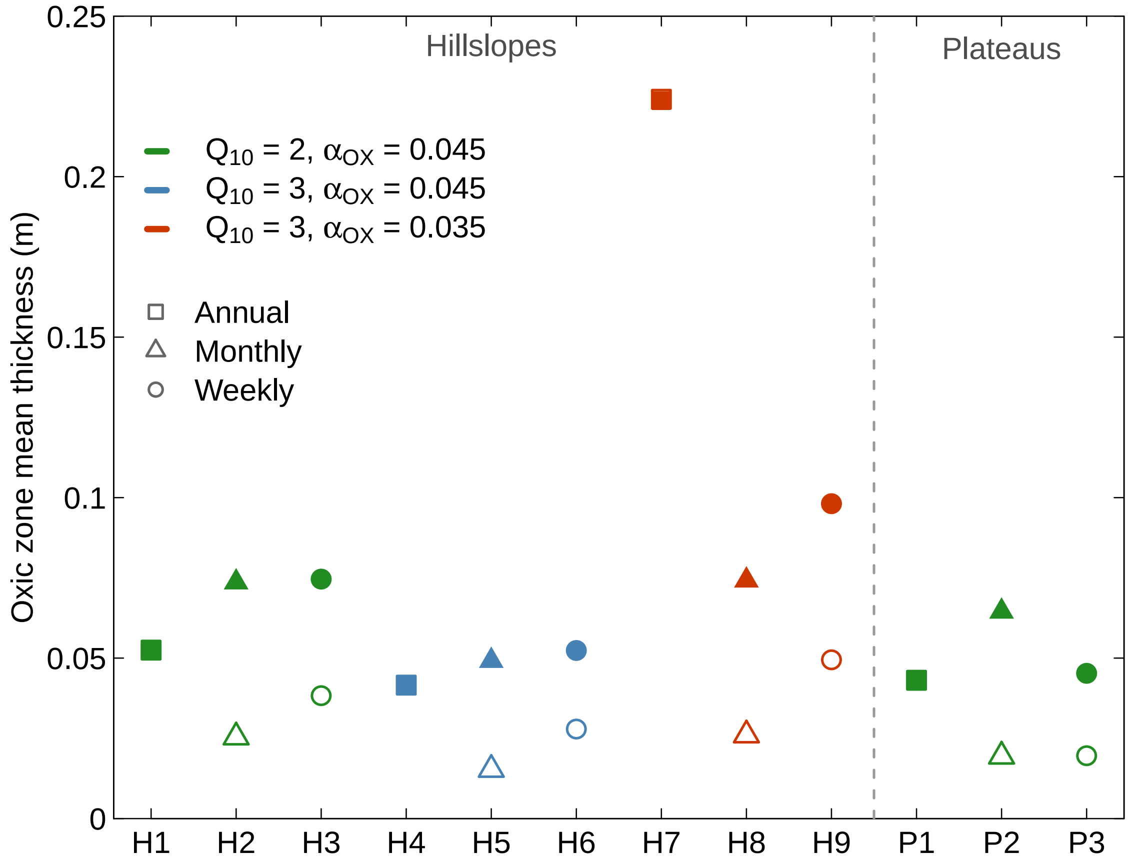


Figure 5.9. Oxic zone thickness. Values are the mean of all columns from a model in the final year of a simulation. Filled symbols are the mid–year mean thickness of the oxic zone throughout the months of May, June and July. Open symbols are the annual mean thickness. Filled and open symbols coincide for annual climate simulations. 'Y' axis labels refer to simulation identifiers; H = hillslope models, P = plateau models.

0.015 m (mean peat thickness = 0.79 m) and 0.026 m (mean peat thickness = 0.98 m). For weekly climate inputs, oxic zones varied from 0.03 m (mean peat thickness = 1.26 m) to 0.05 m (mean peat thickness = 1.59 m). Monthly and weekly inputs resulted in an increase in mid–year (May, June and July) mean oxic zone thickness that range from 0.05 m (monthly climate; $Q_{10} = 3, \alpha_{\mathrm{OX}} = 4.5 \times 10^{-2}$ year^{-1}) to 0.10 m (weekly climate; $Q_{10} = 3, \alpha_{\mathrm{OX}} = 3.5 \times 10^{-2}$ year^{-1}). On average mid–year oxic zones were 0.044 m (Monthly) and 0.036 m thicker (weekly) than annual oxic zone thicknesses. For monthly input simulations, both mean annual and mid–year oxic zones were approximately the same thickness for the parameter sets $Q_{10} = 2, \alpha_{\mathrm{OX}} = 4.5 \times 10^{-2}$ year^{-1} and $Q_{10} = 3, \alpha_{\mathrm{OX}} = 3.5 \times 10^{-2}$ year^{-1}. The greatest difference between mean annual and mid–year oxic zone thicknesses occurred in monthly climate input simulations. However, in all cases mean annual oxic zones were thinner using monthly inputs than the equivalent periods in weekly climate input simulations.

5.4.3 Peatland cross–sectional shape

All plateau simulations produced plausible cross sections for the modelled peatland (Figure 5.8 **b**). In these simulations the peat was thickest at the drainage divide (column 25), which represents the centre of the plateau, and gradually thinned towards the plateau edge. Similarly monthly and weekly hillslope models showed plausible cross sections. In this case, peat thickness was greatest between the centre of the hillslope and column five (10 m from the hillslope edge). Topslope columns were thinner than those at the centre in both monthly and weekly input simulations. Topslope thinning occurs because the downslope gradient causes water to flow quickly away from the top of the slope exposing the peat there to oxic decomposition. The only source of water for the column at the top of the slope was from rainfall, whereas downslope columns receive water from rainfall and from upslope columns. Whilst this was also true for the plateau simulations, the slope gradient results in the removal of water at a faster rate than it is replenished. Hillslope models combined with annual climate inputs, and a parameter set of with the $Q_{10} = 2$ or 3 and $\alpha_{\mathrm{OX}} = 4.5 \times 10^{-2}$ year^{-1} (H1 and H4) also showed the same cross section (Figure 5.8a). However, the hillslope model (H7) with annual climate inputs and the parameter set $Q_{10} = 3, \alpha_{\mathrm{OX}} = 3.5 \times 10^{-2}$ year^{-1}, developed a small mound at columns six and seven (12 and 14 m from the hillslope edge) with a shallow basin immediately upslope of the mound and warrants further investigation.

5.4.4 Three peatlands in more detail

In this section I explore in more detail the development of peatlands H7 (annual), H8 (monthly) and H9 (weekly), all hillslope models with temperature sensitivity $Q_{10} = 3$ and oxic decay rate $\alpha_{\mathrm{OX}} = 3.5 \times 10^{-2}$ year^{-1}. Mean peat depths for the three models were H7 = 3.35 m , H8 = 0.98 m and H9 = 1.59 m. Mean oxic zone thicknesses were H7 = 0.22, H8 = 0.03 (annual) and 0.07 (mid–year), H9 = 0.05 (annual) and 0.1 (mid–year). To further investigate the differences in peat accumulation, and the small mound feature of simulation H7, the height of each peatland and associated mean annual water table was reviewed at 1,000 year intervals throughout each simulation (Figure 5.11). In **`DigiBog`**, saturated hydraulic conductivity K_{sat} is central to the flow of water between columns and to the development of waterlogging which provides the conditions for peatland growth. Because K_{sat} declines with degree of decomposition, the proportion of layer mass θ remaining at the end of each simulation was also investigated for the three peatlands (Figure 5.12). Results for height, water table depth, and proportional layer mass are reported together for the relevant peatlands with the annual climate inputs simulation discussed separately from monthly and weekly simulations.

Annual climate inputs: simulation H7

After 1,000 years, peat accumulation outpaced the water table at both toe– and topslopes with less accumulation and shallower water tables. The water table was ≈0.2 m at 10 m from the toeslope, and ≈0.03 m at 16 m. Between 1,000 and 3,000 years, peat accumulation increased across the hillslope, but rates of accumulation were lower between 14 m and 20 m where a small basin formed with near–surface water tables (within 0.05 m), which may represent the formation of a pool. Accumulation slowed over the next 2,000 years and at the end of 4,000 years the small basin had infilled: the water table was at ≈0.21 m below the peat surface at this location, which was similar to the mean water table for the whole peatland of ≈0.22 m. The model appears to have reproduced the dynamics of pool formation and later infilling.

The proportion layer of mass that remained after decomposition shows that the two columns nearest the toeslope edge (0–4 m) were highly decomposed for the majority of their height with little of the original layer mass remaining (Figure 5.12a). The next four columns (4–12 m from the toeslope edge) were also highly decomposed in the top quarter of their profiles. Together, these columns inhibited the flow of water out of the hillslope. Within the centre of the peatland, peat layers were more decomposed as distance from the mineral base increased. The most striking feature of this decomposition profile was that the boundary between column seven (12–14 m from the toeslope edge) and its immediate downslope neighbour appeared to be quite abrupt with a mid–column section that was the least decomposed of the entire peatland. This boundary corresponds to the peak of the mound mentioned earlier.

Monthly and weekly climate inputs: simulations H8 and H9

The development of peatland height, and water table–depths for monthly and weekly climate simulations is shown in Figure 5.10. The height of both peatlands continued to increase throughout the simulations with mean annual water tables that were approximately five (weekly) and 10 (monthly) times shallower than that of the annual climate simulation. Water tables were initially above the peat surface but quickly (i.e. after a few years) fell below it. The water table in the monthly simulation reached a depth of 0.023 m by 500 years and remained approximately at that depth as the peatland grows so that by 5,000 years it was 0.026 m below the peat surface. The water table for peatland H9 was almost twice as deep, and over the same period continued to slowly deepen from 0.04 m to 0.05 m.

Peatlands H8 (monthly inputs) and H9 (weekly inputs) showed similar spatial patterning for the proportion of mass that remained for each layer (Figure 5.12b and c), which was dissimilar

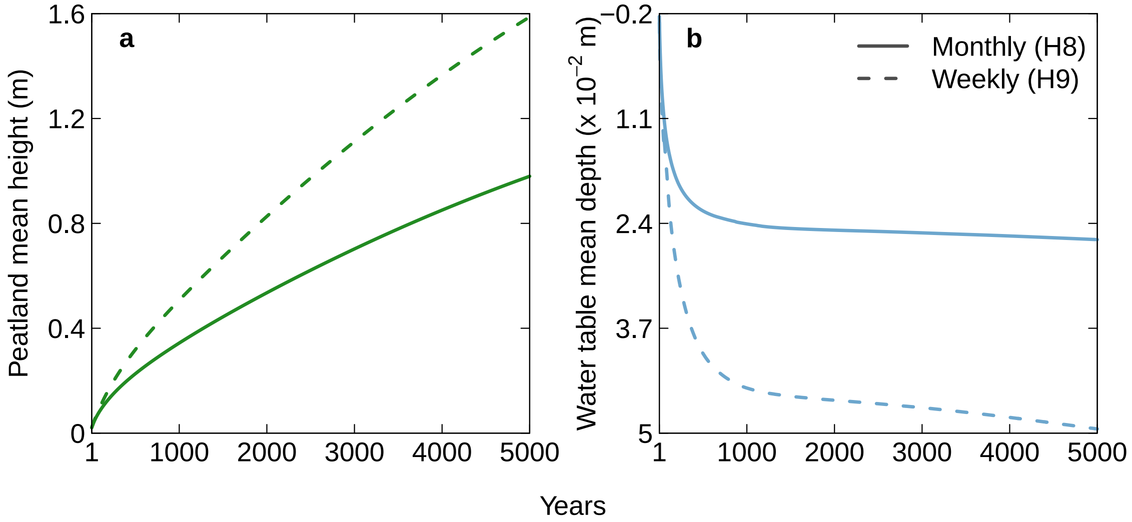


Figure 5.10. Comparison of monthly and weekly climates. a peatland height and **b** water table depth for monthly and weekly climate simulations. Peatland height is the mean of all columns at the end of each year. Water table depth is the mean annual depth of all columns for the year, where positive values are below the peatland surface.

to the annual climate inputs simulation. Although decomposition again increased with increasing distance from the peatland base, the abrupt transition observed in the annual inputs simulation at column seven was absent. There are a number of zones of decomposition that are shown as relatively symmetrical mounds. However, the columns in H9 were less decomposed than in H8; shown by larger mounds with higher proportional mass remaining. In peatland H8 the peat columns were more decomposed towards the topslope than the toeslope and the column at the top of the slope (48–50 m) was also highly decomposed. In both peatlands H8 and H9, the column at the edge of the toeslope (0–2 m) was highly decomposed.

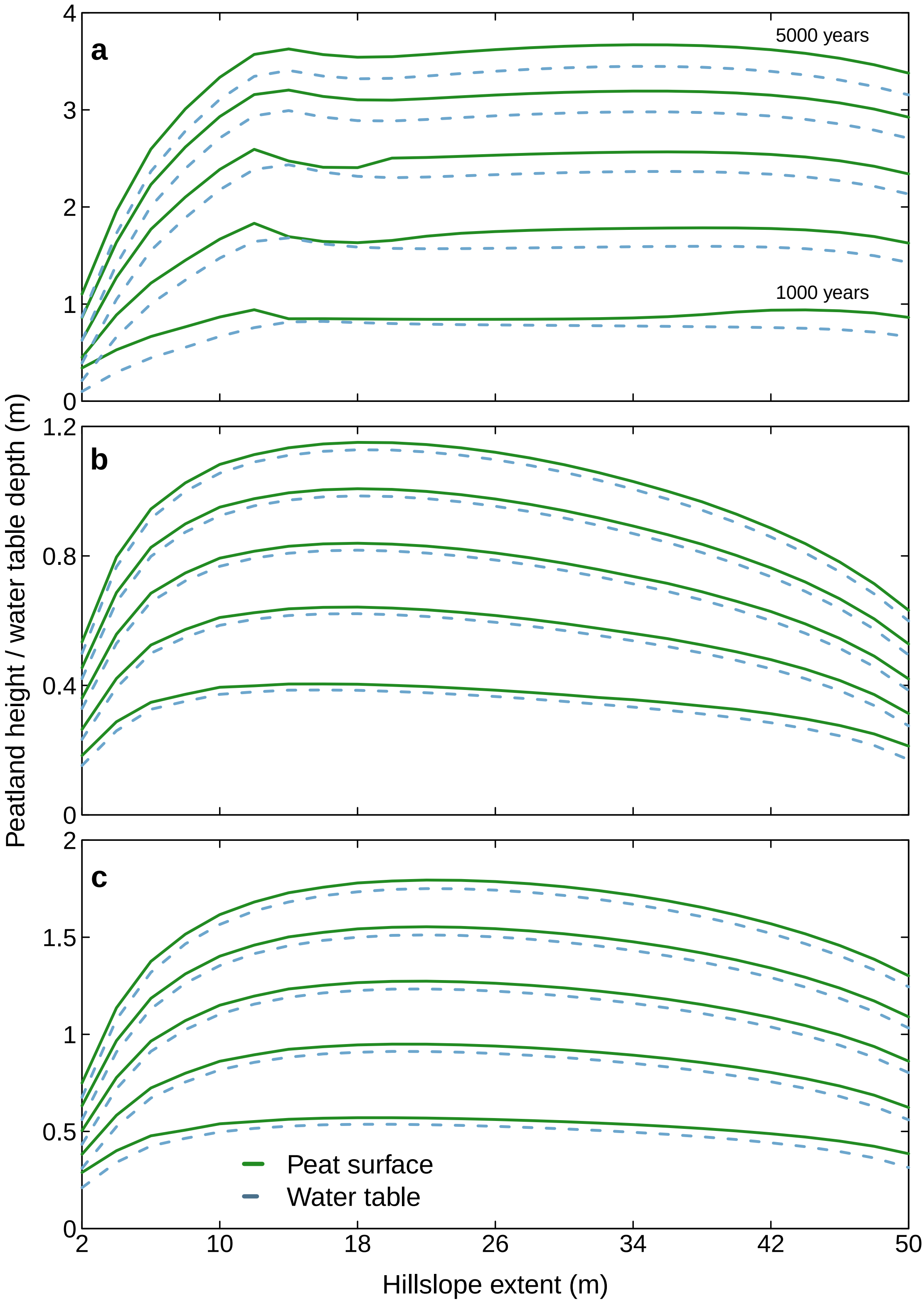


Figure 5.11. Peatland development over 5,000 years. Each pair of peat and water table heights (solid and dashed lines) represents the accumulation of peat every 1000 years for 5,000 years for **a** annual (H7), **b** monthly (H8) and, **c** weekly (H9) climate simualtions. Peat surface is the height accumulated at the end of each period. The water table is the mean annual depth for the year plotted. Note the difference in 'y' axis scales.

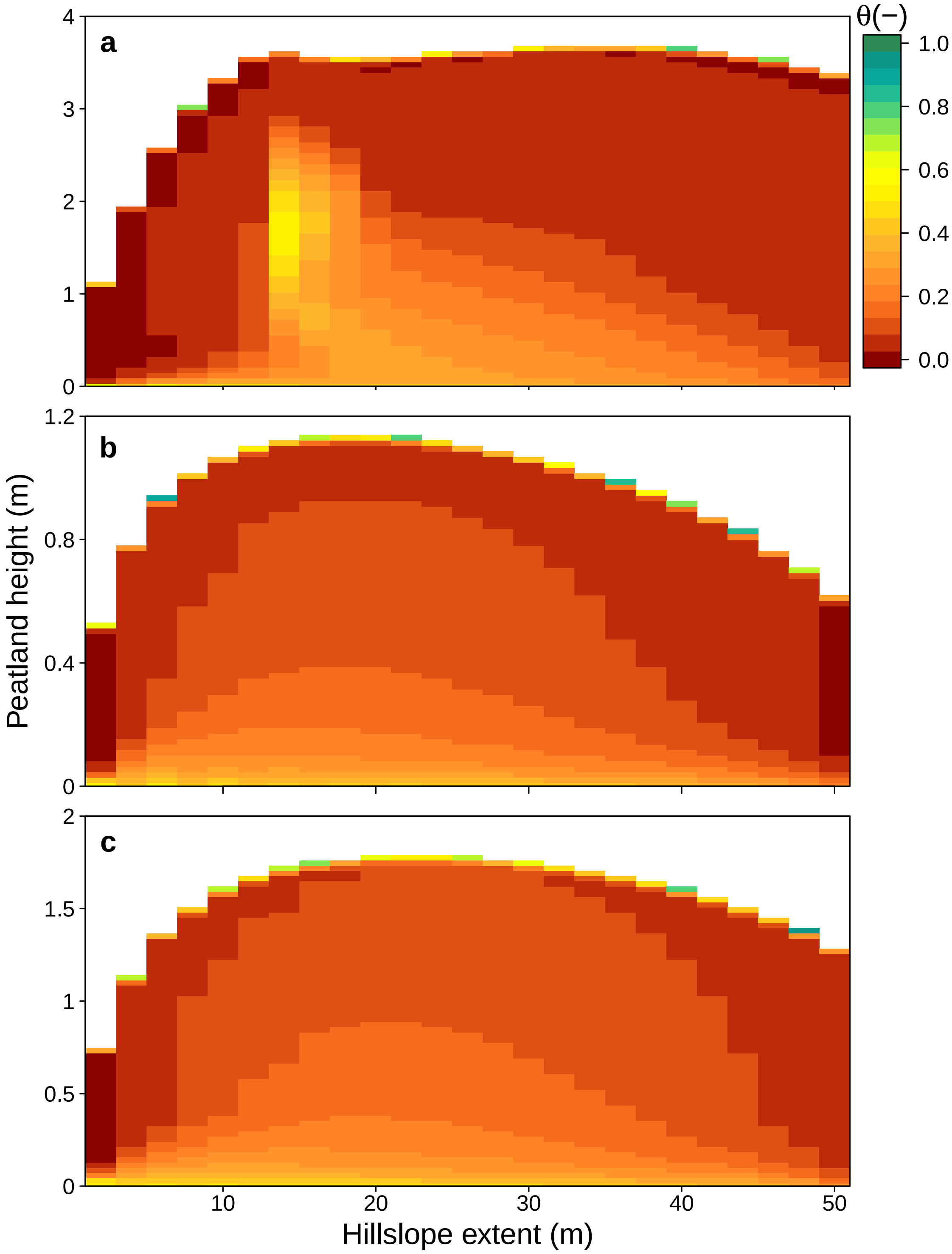


Figure 5.12. Peat layer decomposition. Colours represent the proportion of mass remaining θ (unitless) in a peat layer after 5,000 years of decomposition for **a** annual (H7), **b** monthly (H8) and, **c** weekly (H9) climate simualtions. Vales of θ near to 0 represent highly decomposed peat. Note the difference in 'y' axis scales.

5.5 Discussion

The work reported in this chapter represents the current progress on modelling the development of blanket peat as a complex adaptive system, over centennial and millennial timescales, on hillslopes and plateaus. The model used was a further development of the **`DigiBog`** peatland model first described by Baird et al. (2011) and Morris et al. (2011a) and represents the first attempt to simulate blanket bog development on hillslopes as a set of hydrologically connected columns that incorporates a feedback mechanism between plant litter production, water–table depth, and decomposition.

I assessed the effect of using annual, monthly and weekly intra–annual climate variables on peat accumulation in 2D hillslope and plateau models. The results demonstrate that annualised climate variables, such as those used in 1D peatland models reported by Frolking et al. (2010) and Swindles et al. (2012), accumulate greater quantities of peat in a 2D model than monthly and weekly climate variables. In addition, one of the annual climate simulations also shows how complex spatial behaviour can develop in a 2D model from the interaction of climate variables, model parameters, and the autogenic processes within the peatland. Simulation H7 may provide an insight into the mechanisms that allow pools to form and infill in blanket peatlands.

I hypothesised that including sub–annual variation in climate variables would change the rate of peat accumulation as a result of different distributions in net–rainfall and temperature (Section 5.2.2). Here I show for the first time that annualised climate inputs result in a significant difference in peat accumulation ($>$1 m mean peat height), and highlight the importance of taking into account the sub–annual distribution of climate variables in peatland development models. For simulations that used monthly and weekly climate inputs, different peatland heights are due to; (1) lower plant productivity because of shallower mean annual water–table depths (Figure 5.9), and (2) increases in mid–year temperatures, and reductions in net rainfall that incur higher rates of decomposition of new layers, and of previously submerged layers (Figure 5.13).

5.5.1 Litter production and decomposition

The temperature distribution of both sub–annual climate inputs is approximately the same; low in the winter, rising to a peak in the middle of the year and falling again towards winter. However, whereas monthly net rainfall distribution is highest in the winter and lowest in the middle of the year, the weekly distribution is much 'noisier'. Because litter production in **`DigiBog`** occurs annually and depends on mean annual temperature (which is the same for all climates) and mean annual water–table depth, differences in litter production between climates are driven by differences

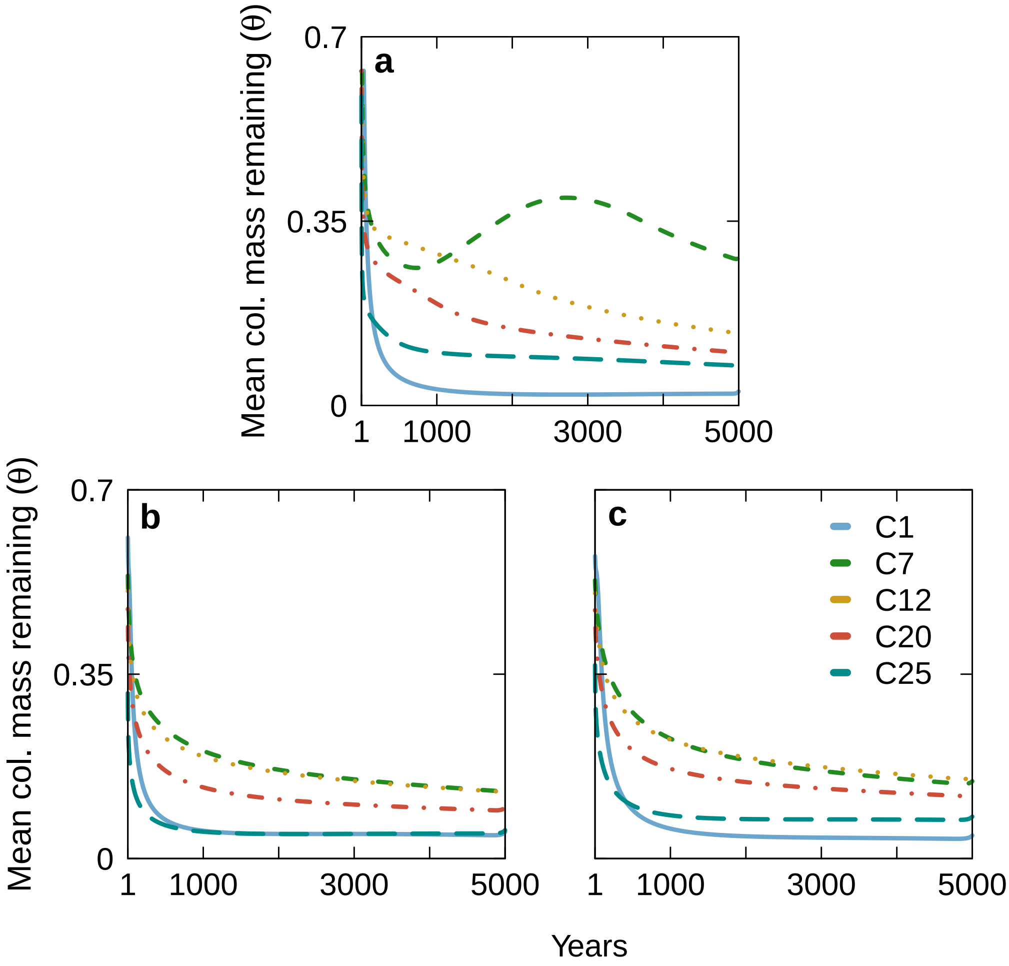


Figure 5.13. Time–based decomposition profile of selected peat columns. Annual mean proportion of column mass remaining for **a** H7 (annual climate), **b** H8 (monthly climate) and, **c** H9 (weekly climate). Columns (C) 1 (toeslope, 0–2 m), 7 (12–14 m), 12 (22–24 m), 20 (38–40 m) and 25 (topslope, 48–50 m). Peat becomes more decomposed as mass remaining values move from 1 to 0.

in water–table depth (oxic zone thickness). Litter production, shown in Figure 5.14, reaches a maximum of 1.04 kg m^{-2} $year^{-1}$ where the water–table depth is 0.30 m , but the growth in height of the modelled peatlands also depends on the how much litter is lost to depth integrated decomposition.

Unlike litter production, decomposition occurs during each model timestep (every 300–3,000 s, i.e. between 10,500 and 105,000 times a year) and depends on oxic and anoxic decay parameters, which are dependent on temperature, Q_{10}, and water table position. Higher rates of decomposition are applied to the proportion of peat above the water table and therefore, for the same parameter set, differences in decomposition between annualised and sub–annual climates are driven by both temperature and changes in water–table position. However, because of the similarity in temperature distribution between monthly and weekly climates, the differences in decomposition are mainly due to water–table position. In **`DigiBog`**, decomposition also results in a reduction in hydraulic conductivity K which determines the rate of water movement between columns, mediating water

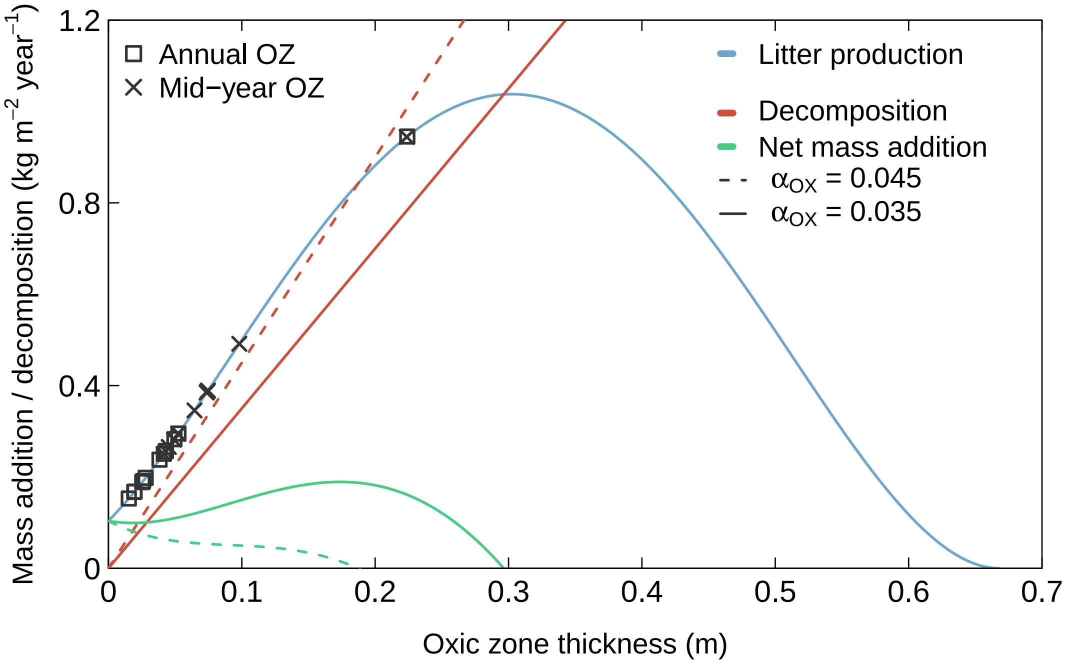


Figure 5.14. The effect of oxic zone thickness on peat accumulation in DigiBog. Annual values for litter production and depth–integrated decomposition re–calculated from Morris et al. (2011a) with temperature dependent litter production. Net mass addition is calculated from litter production minus decomposition. Squares and crosses represent the annual and mid–year (May, June and July) oxic zone mean thickness for sloping and plateau models at the end of 5,000 years. OZ = oxic zone.

table depth, and hence creating a feedback to litter production and decomposition. The effect that decomposition and K have on the development of the modelled peatlands presented here is discussed later in this section.

Peatland growth occurs because of the difference between litter production and decomposition. Figure 5.14 illustrates how litter production is affected by the different oxic decay parameters and oxic zone thickness for annualised climate inputs. For example, with the parameter set $Q_{10} = 3$ and oxic decay parameter $\alpha_{OX} = 3.5 \times 10^{-2}$ year^{-1}, the maximum annual litter addition is 0.19 kg m^{-2} year^{-1} when the oxic zone is 0.17 m; 0.13 m less than that for maximum litter production. The annual inputs simulation (H7) develops an oxic zone that is 0.22 m thick and accumulates 0.17 kg m^{-2} year^{-1}; the highest rate of all simulations. Although the litter addition curve shown in Figure 5.14 applies to all simulations, the dynamics of decomposition are more complex for monthly and weekly climates because new values of temperature and net rainfall are read into the model more frequently (12 or 52 times a year), and interact to affect the rate of decomposition and the position of the water table which, as stated above, affects decomposition.

The impact of intra–annual climate variables on peat accumulation can be further examined by considering the effect of changes to Q_{10} and to the oxic decay parameter α_{OX}. Of the three sets of climate inputs, peat accumulation in monthly simulations is the most sensitive to changes in Q_{10} (i.e. the greatest decrease in accumulation between Q_{10} values) and least sensitive to subsequent reductions in the rate of oxic decay (i.e. the smallest increase). This is intuitive because increasing Q_{10} to 3 leads to greater oxic decomposition during the mid–year when oxic zones are thickest, temperature is highest and water input is low. Lower oxic decay rates do increase peat accumulation, but because of the mid–year droughty conditions, greater amounts of litter are decomposed than for weekly climates.

5.5.2 Peatland development

Both monthly (H8) and weekly (H9) inputs simulations follow a similar developmental pattern (Figure 5.11). The water table in the toeslope column (0–2 m) is drawn down as the gradient to the peatland margin (boundary column) increases resulting in oxic decomposition. The toeslope column becomes highly decomposed (Figure 5.13) reducing K and impeding water flow out of the model: these toeslope columns accumulate the least peat. Because water flow is impeded, the water table in upslope columns remains close to the surface as water from both net rainfall and topslope columns drains down the hillslope. As a result, a mound of less decomposed peat gradually accumulates in columns where the water table is closer to the surface. These findings are supported by Lapen et al. (2005) who also found higher rates of decomposition at the toeslope margin of a blanket bog in modelling results and field observations. Lewis et al. (2012) also reported lower measured values of hydraulic conductivity at blanket peatland margins. My results also confirm the hypothesis of Lapen et al. (2005) that highly decomposed toeslope columns (peatland margins) lead to water table doming and increased peat accumulation within the blanket peatland.

The peatlands modelled here developed a range of peat depths that are within reported ranges for Peak District blanket peats (e.g. Conway 1954; Moore 1975; Tallis 1987). However, the annual climate simulation H7 is probably too deep for blanket peat on sloping terrain when combined with the modern climate variables used here. Peat depths on Keighley Moor were found up to $\approx$3.7 m (Blundell and Holden 2015) but these deepest accumulations occurred in underlying topographic hollows. Others have reported varying peat depths, for example Holden and Burt (2003) reported a mean catchment depth of 1.54 m for a blanket peatland in the North of England, Lapen et al. (2005) measured depths for an blanket peatland complex of $<$2m, and Tallis (1987) reported peat depths of between 2.0 and 2.75 m on 5 $^{\circ}$ slopes at Holme Moss in the Dark Peak District. There are a number of reasons that may account for these differences; (1) measured peat depths have accumulated under past climates, whereas the data used here were based on four years of recent climate that is repeated for all modelled years; (2) accumulation may have occurred over longer timescales (initiation dates of between 8,000 and 5,000 years ago have been proposed in the Peak District, Conway 1954; Moore 1975); (3) vegetation composition may differ from the modelled function; and (4) the modelled slope and selected parameters were general rather the based on the conditions of a specific location. Plateau models might be expected to accumulate more peat than sloping models (Conway 1954, reported depths of ≈2.4–4.0 m on plateaus in the Dark Peak) but those presented here show only a small increase in height. For example hillslope (H3) and plateau (P3) models combined with weekly climate inputs accumulated 1.39 m and 1.54 m respectively.

5.5.3 Recommendations

For the climate records used here, the monthly distribution of inputs resulted in very depressed rates of net rainfall for mid–year periods. This is because although the total net rainfall used is the same for each of the three climates, the variation in daylight hours used to calculate potential evapotranspiration, results in a different distribution of net rainfall for weekly climate variables. The temporal resolution of climate data is clearly an important consideration in models that include temperature dependent functions that are mediated by water table depth (such as oxic decay processes). Across the parameter sets explored, weekly climate inputs appear to produce the best estimation of peat accumulation, although annual simulations where $\alpha_{\mathrm{OX}} = 4.5 \times 10^{-2}$ year^{-1} give highly plausible results too. This means that within a perhaps narrower parameter space, models that use annualised weather also produce credible results for peat accumulation that has occurred under past climates.

However, because the sub–annual distribution of rainfall and temperature is key to blanket bog development (Lindsay et al. 1988; Charman 2002), and as much of the discussion of the impact of climate change on northern hemisphere peatlands has identified the importance of of changes in the sub–annual distribution of rainfall and temperature (e.g. Gallego-Sala et al. 2010; Charman et al. 2013), I recommend that weekly climate variables are used when exploring the impact of these changes. As all simulations took approximately the same amount of time to run (four weeks), the only obvious reason that might preclude the use of weekly climates is the availability of past or future climate data at this resolution.

5.5.4 Model limitations and further developments

The simulations shown here suggest that **`DigiBog`** is a suitable model to explore blanket bog peat accumulation on sloping terrain, although further developments are needed. The most significant limitation is the small section of 2D slope that was used for simulations. Blanket peatland complexes can extend over several km^2 but the computational time needed to model such landscapes with the current structure of 2D/3D **`DigiBog`** is prohibitive. Morris et al. (2011b) make a number of suggestions that might improve the output of **`DigiBog`** simulations such as the addition of plant functional types, depth dependent bulk density, and porosity, but the priority for 2D/3D modelling should be improving computational efficiency so that larger landscapes can simulated, and the differences between 2D and 3D models understood. Other developments of the model are considered next.

(1) The current 2D/3D version of **`DigiBog`** has produced interesting complex behaviours seen in real peatlands, but that have not been reported in other 1D or 2D peatland development models. One of the simulations reported here (H7), developed a large hollow where a pond could form because highly decomposed toeslope columns caused shallow water tables to develop in neighbouring upslope columns. As a result, peat accumulation in these upslope columns was lower than the others in the peatland and a basin developed. In addition, a number of simulations that were not discussed (where $\alpha_{\mathrm{OX}} = 2.5 \times 10^{-2}$ year^{-1}), showed model outputs that could represent the formation of hummock– and hollow–like microtopography. These are potentially exciting developments that require further investigation to determine if these features are a result of the interaction of autogenic processes and allogenic forcing, or are as a result of numerical artefacts of the solutions to model equations. If the results are due to the former, then the model may provide a mechanistic approach to understand how blanket peatland microtopography develops and persists. From limited investigation, it appears that impeded water flow occurs in some columns due to the interaction of low hydraulic conductivity and the rate of decay in newly added peat: but the reasons for this behaviour are not yet clear. Further testing at smaller spatial scales to see if the patterns are a result of column size would answer this question.

(2) Some peatland models include plant functional types based on sparse empirical data (e.g. Frolking et al. 2010; Heinemeyer et al. 2010), whereas the litter function in **`DigiBog`** was developed from empirical data. That said, alternative litter production and decomposition functions specific to blanket peatlands could be used to investigate management processes such as grazing or burning and could help to explore some of the concerns highlighted by peatland stakeholders reported in Chapter 4. These concerns were related to efforts to restore near–surface water tables, or to deliberately replace heather dominated areas with *Sphagnum* spp. mosses, and cannot be tested in the current version of **`DigiBog`**. One approach would be to reduce litter input to account for removal by grazing according to consumption patterns, alternatively there are a number of grazing models that could be used to parameterise the impact of sheep and deer (e.g. Palmer et al. 2004), but functions that describe plant succession and the impact on litter accumulation and decomposition would need to be developed.

(3) Above surface flow in **`DigiBog`** is limited. A ponding layer allows water to move above the peat surface, but water that exceeds the specified depth is lost from the model domain in a simple representation of overland flow. However, the incorporation of spatially distributed overland flow

when water tables exceed the pond (*sensu* Gao et al. 2015), may retain more water within the model domain whether from upslope columns or periods of high net rainfall.

(4) Daily distributions of rainfall and temperature were available for Keighley Moor but no attempt was made to estimate daily evapotranspiration needed to simulate peat development using these data because of the likely additional modelling time required. Whilst finer temporal resolution is unlikely to affect the overall distribution of temperature, experiments with daily rates of net rainfall could take place to ascertain the benefit of these data to understanding peat accumulation in blanket peatlands.

5.6 Conclusions

The effect of climate on peatland carbon accumulation has been one of the most researched topics in peatland science, but peatland development models have previously ignored the intra–annual distribution of climate variables. However, the distribution of net rainfall and temperature within a year may be more important for blanket peatland development than annual totals (Charman 2002), and studies have proposed that this distribution is likely to change in future climates. Here I show that it is important to take into account the sub–annual temporal variation of climate variables in peatland development models, even over millennial timescales. This study demonstrates that the use of annualised climate inputs (especially in relation to blanket peatlands), is likely to overestimate peat accumulation, and I recommend the use of weekly timeseries of net rainfall and temperature.

The simulations reported in this chapter do not attempt to model the development of blanket peatlands under projected climates. However, the simulations that use a monthly distribution of climate variables provide an indication of how peat accumulation rates are likely to be reduced in intact peatlands during warm dry periods, which could become a feature of future climates (e.g. Gallego-Sala et al. 2010). Because **`DigiBog`** incorporates a set of autogenic feedback mechanisms that produce resilience to external forcing, which is not a feature of bioclimatic envelope models (e.g. Gallego-Sala et al. 2016), the response of modelled peatlands to future climates coupled with management can be simulated. My results demonstrate the potential of **`DigiBog`** to investigate issues raised by stakeholders about how climate, in combination with future management, is likely to affect peat accumulation in the South Pennines. Having established how changes in climate distribution should be represented in the model, I now, in Chapter 6, couple weekly climate variables with peatland restoration, to investigate the impact of gully blocking on blanket peat carbon storage.

Chapter 6

The impact of gullies and gully blocking on blanket peatland carbon storage

6.1 Chapter summary

In Chapter 5, I described the development of a new version of the **`DigiBog`** peatland model in order to investigate impact of climatic forcing and management on blanket peatland carbon stocks. With the new model, I investigated the how the simulated development of blanket peatland carbon stores were affected by different temporal resolutions of rainfall and temperature. In this chapter, I now investigate how the model could be used to explore the effect on carbon storage of one of the peatland restoration options that was classified as a driver concept, and identified in a stakeholder workshop as a potentially important route to deliver carbon, water and livelihood objectives for South Pennine blanket peatlands. Three restoration options that were judged by stakeholders to increase carbon storage, and improve water quality, were perceived to be incompatible with the objective to increase the provision of local livelihoods (Chapter 4). Of the three options, I chose to investigate gully blocking (refer to Section 6.2.1 for the rationale for this decision). The scope of this chapter is outlined in Figure 6.1.

In conjunction with Chapter 5, the aim of this chapter is to address research question 4;

What is the predicted impact of social and ecological factors on the centennial to millennial storage of carbon in blanket peatlands when conceptualised as a complex system?

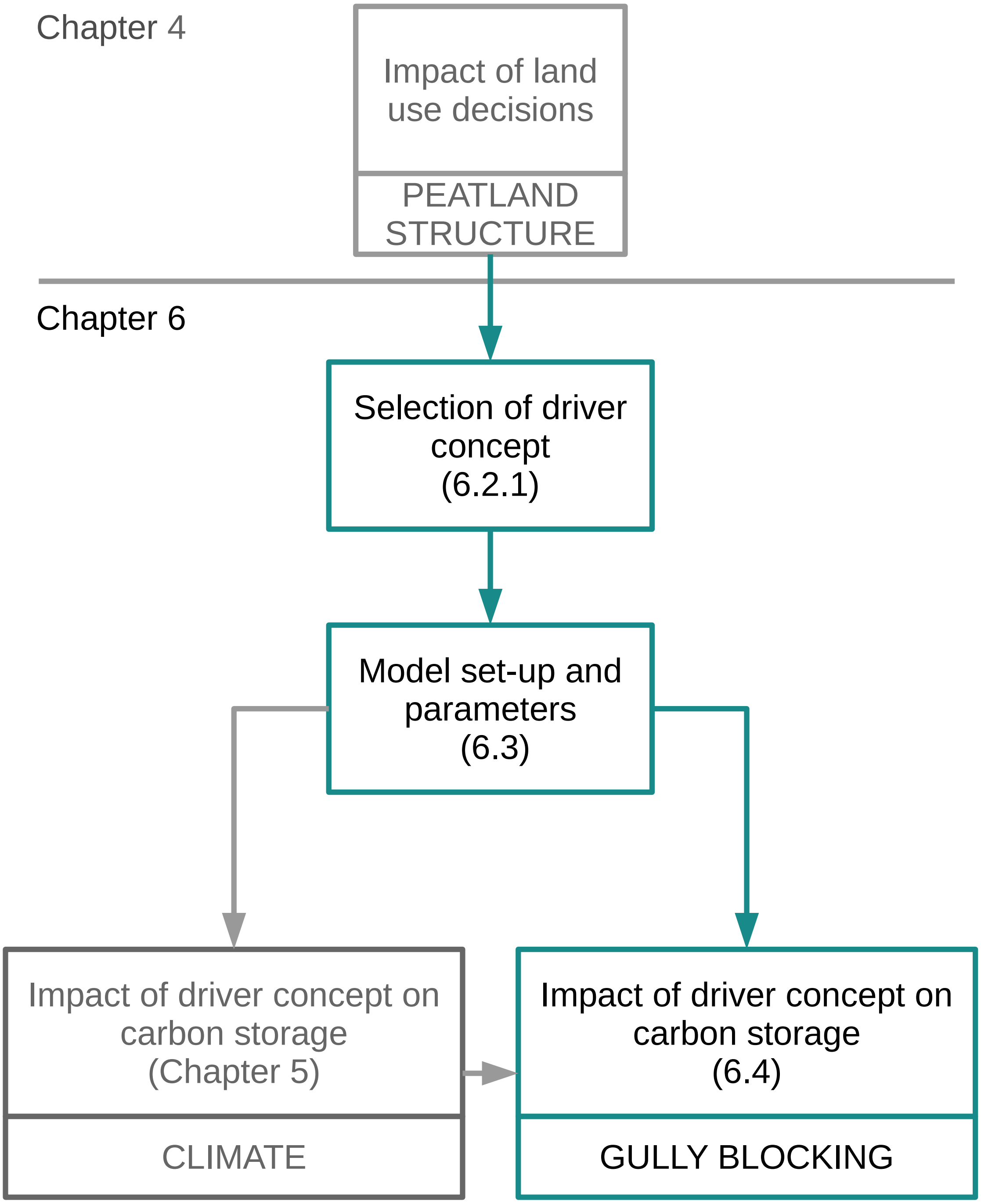


Figure 6.1. Scope of Chapter 6. Figures in brackets represent chapter section numbers. The findings from Chapter 5 are incorporated into the model used in Chapter 6.

6.2 Introduction

6.2.1 Rationale for selection of gully blocking

During the stakeholder workshop reported in Chapter 4, participants proposed an increase in restoration activity to block gullies in order to increase carbon storage and improve water quality. However, three driver concepts (gully blocking, mosses and trap eroded peat) were not fully recognised by stakeholders as opportunities to support local livelihoods. This may have been because the result of increasing these drivers is perceived to be shallower water tables and associated uncertain consequences for livestock and game birds (*Peatland restoration - what's in it for me?* 2015). However, the engagement of local land users in these restoration activities is a potential route to adapting local livelihoods in a movement away from activities that damage to those that restore peatland ecosystem function (Chapter 4), and therefore these driver concepts warrant further investigation. I chose to investigate gully blocking because; (1) according to the highly important subset of the stakeholder network model (used in Chapter 4), gully blocking was the most influential of the three concepts and is connected to nine other concepts (whereas mosses and trap eroded peat were connected to one concept each); (2) in combination with revegetation, gully blocking is one of the most widespread restoration techniques used in the South Pennines Special Area of Conservation (e.g. Pilkington et al. 2015); and (3) the long–term effect of gully blocking on the spatial and temporal dynamics of carbon storage in blanket peatlands is poorly understood.

6.2.2 The impact of gullies

Erosional gullies are one of the most significant and longstanding features of ecosystem degradation of peatlands in the UK and Ireland (Bragg and Tallis 2001). Because of predicted climate change, blanket peatlands that have established in some UK locations, that are now forecast to become unsuitable for future peat accumulation (Gallego-Sala et al. 2010), are likely to show increased degradation (Li et al. 2015): marginal areas for peatland growth include the South Pennines where erosional gullies dominate plateaus and slopes in some locations (Tallis 1987; Evans and Lindsay 2010). The initiation of gullying has been linked to combinations of allogenic and autogenic processes that have taken place over centuries (Tallis 1985; Tallis 1987). These process may have begun with rapid litter production that resulted in mechanically unstable peat which became susceptible to break–up under harsh weather conditions (Conway 1954), bog bursts around the peat margin (Tallis 1985), and anthropogenic impacts such as forest clearances and fire (Tallis 1987). The process of gullying is thought to have been later exacerbated by grazing, draining, managed burning, wildfire and industrial pollution (Tallis 1987; Evans and Lindsay 2010). Tallis (1985)

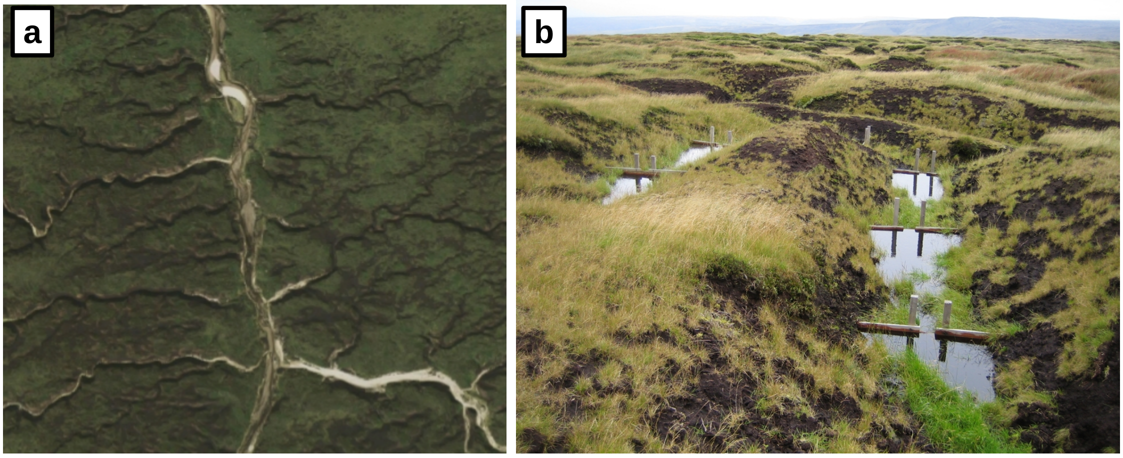


Figure 6.2. Gullies on Kinder Scout in the Peak District, UK. a Aerial image of a gully network showing gullies perpendicular to the main gully. Source; Map data ©2015 Google Imagery ©2015 DigitalGlobe, Getmapping plc, Infoterra Ltd & Bluesky, The Geoinformation Group. **b** Blocked gullies. Photograph courtesy of Mike Pilkington, Moors for the Future.

suggested that gullying began in the Southern Pennines > 1,000 years ago. More recent periods of erosion beginning in the last 200–300 years were probably initiated by air pollution and occurred in conjunction with periods of intensive grazing which, over the last 100 years, has prevented natural revegetation of bare peat (Tallis 1985).

Gullies can form closely–packed branching networks and become several metres wide and deep (Evans et al. 2005) where smaller gullies can form perpendicular to the main gully and hillslope (Figure 6.2a and b): as a result, gullies contribute significantly to the loss of blanket peatland carbon. Fluvial erosion is thought to account for the majority of these losses (Evans and Lindsay 2010; Grayson et al. 2012) although bare gully sides are also exposed to wind–driven erosion (Foulds et al. 2007) and other weathering processes such as dessication (Grayson et al. 2012). Water–table drawdown at the edges of gullies can have a significant effect on the carbon balance and hydrological function of blanket peatlands (Daniels et al. 2008; Allott et al. 2009; Dixon et al. 2014). Gully sides may, (1) expose down–profile peat, that may have been below the water table for hundreds of years, to secondary oxic decomposition (e.g. Tipping 1995; Borgmark and Schoning 2006; Morris et al. 2015a); (2) create preferential pathways for stormflow (Daniels et al. 2008); and (3) increase carbon lost during wildfires and managed burning (Turetsky et al. 2011). For example, Evans and Lindsay (2010, Table 3) reported that the effect of gully erosion was to switch the carbon budget of a Peak District blanket peatland plateau from a sink of -20.3 g C m^{-2} $year^{-1}$ to a source of 29.4 g C m^{-2} $year^{-1}$. It was estimated that gully sides represented ≈34% of this increased loss of carbon.

Gullies in blanket peatlands also reduce water quality (Daniels et al. 2008). In combination, water quality and carbon storage (Daniels et al. 2008; Parry et al. 2014) represent two highly important ecosystem services that provide benefits over national and global scales (Maltby 2010), making the reversal of these negative impacts key targets for peatland restoration by gully blocking. The aim of restoration is to prevent continued erosion and reduce peat oxidation by revegetating bare gully sides and by repairing hydrological function (e.g. Evans et al. 2006). Various techniques and materials are used to block gullies depending on the nature of the site, size of gully, prevailing hydrological conditions and estimated costs (Parry et al. 2014); the materials used include timber, peat and stone dams, and plastic piling (Evans et al. 2005; Parry et al. 2014; Pilkington et al. 2015). Where gullying is present, gully blocking is often combined with revegetation as the mainstay of blanket peatland restoration (e.g. Pilkington et al. 2015).

The return to near–surface water tables is seen as a measure of restoration success, but the impact of gullies and gully blocking on water tables, and the associated drawdown zone around a gully cannot be reliably modelled solely from a calibrated dataset of water–table depths (e.g. Allott et al. 2009), because water tables in soils are determined by hydraulic conductivity K, water storage capacity (porosity), and water–table gradient (Baird 2014). I could find no studies of gullying or gully blocking in blanket peatlands that took into account peat properties (Holden et al. 2006; Wallage and Holden 2011, did so in their assessment of peatland drains) even when the importance of these properties, such as hydraulic conductivity, is discussed (e.g. Daniels et al. 2008; Allott et al. 2009). Ballard et al. (2011) modelled the hydrology of a drained UK blanket peatland over a hillslope using coupled 1D models that incorporated peat properties, with drains being analogous to gullies. However, the model of Ballard et al. (2011) did not attempt to represent the effect of drains on peat accumulation across the hillslope. Given that secondary decomposition of gully sides is likely to further reduce K, which in turn affects water movement, I used the version of the **`DigiBog`** peatland development model constructed in Chapter 5, to explore the centennial effect of gullying and subsequent blocking on peatland hydrology and peat accumulation over a 50 m transect of virtual peatland.

6.3 Model and method

In this version of **`DigiBog`**, peat accumulation is simulated from the addition and decomposition of peat layers in hydrologically connected columns. The 1D peatland development and 3D hydrological models of Baird et al. (2011) and Morris et al. (2015a) were combined to simulate blanket bog development on hillslopes or plateaus. In the model, plant litter is added as a layer at the start of each year to each column, depending on the mean annual temperature and average thickness of the oxic zone of that column in the previous year. Depth integrated oxic or anoxic decomposition of each layer takes place sub–annually according to the proportion of the layer

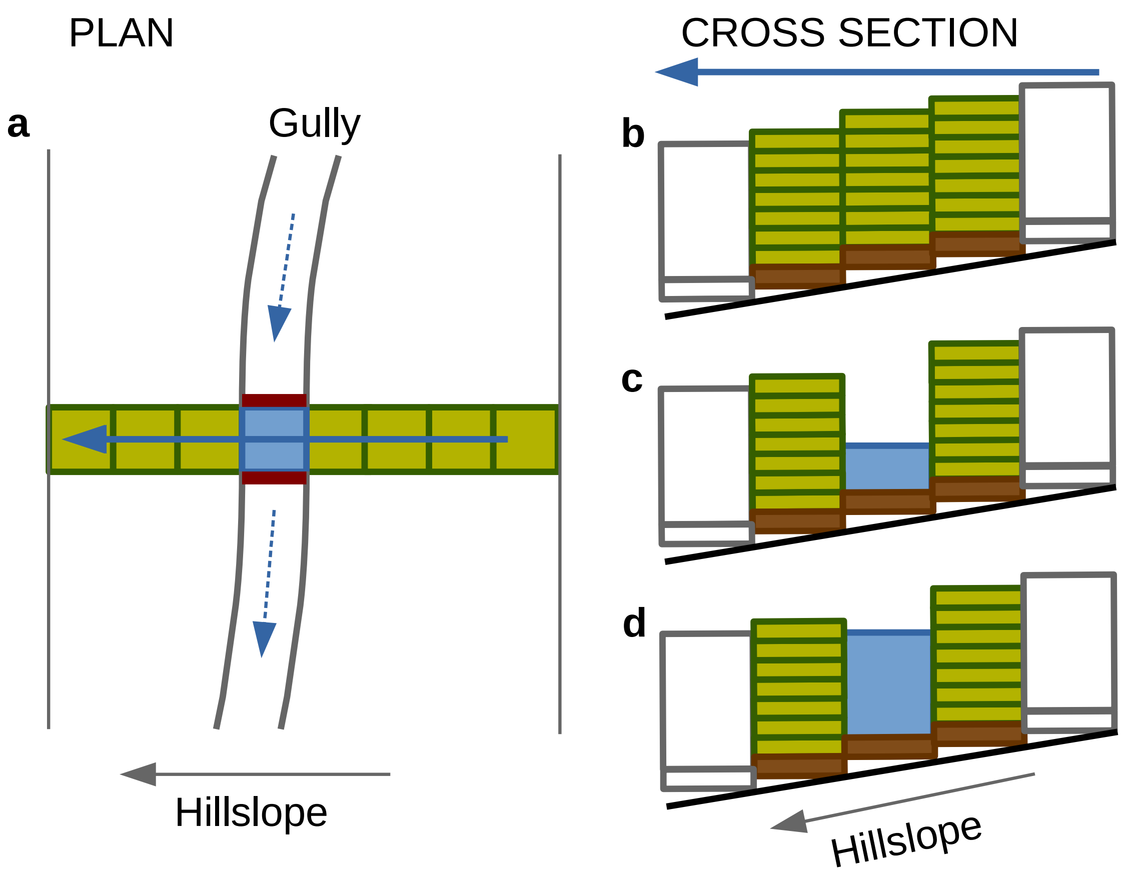


Figure 6.3. Conceptual model of peatland gully blocking. a Plan view showing gully perpendicular to the hillslope. Green squares are the model peatland columns, the solid blue arrow represents model downslope water flow, dashed blue lines represent gully water flow, thick red lines represent gully blocks and the vertical grey lines are the model boundary. Cross sections of the peatland **b** intact, **c** gullied and **d** gully blocks in place.

above or below the water table, oxic and anoxic decay parameters, and temperature sensitivity Q_{10}. The hydraulic conductivity K of each layer is updated at each sub–annual timestep according to the proportion of the original layer mass θ that remains after decomposition, and therefore declines with increasing decomposition (Morris et al. 2011a). Water tables are transient and so sub–annual timesteps are used to produce a numerically stable solution to water movement between columns. Water movement in **`DigiBog`** occurs horizontally between columns according to the Boussinesq equation described by Baird et al. (2011): a similar approach is used in the hydrological model of Ballard et al. (2011). The vertical movement of water in a column takes into account the pore space available for storage in each layer and a surface ponding layer. Overland flow is not modelled explicitly but once surface water exceeds the ponding depth, it is lost from the model domain akin to rapid overland flow. The model is described in detail in Chapter 5.

Model boundary conditions were based on (1) climatic inputs in the form of net rainfall (precipitation minus evapotranspiration) and temperature, (2) a Neumann no flow condition at the topslope, (3) a Dirichlet constant water water level condition at the toeslope, and (4) an impermeable base below a 0.02 m mineral soil layer. The modelled peatland used in all simulations was configured with 25 $\times$ 2 m^2 columns. A conceptual model of the configuration used for hillslopes is shown in Figure 6.3. Hillslope models were set on a 3 $^\circ$ slope. Because blanket bog development is sensitive to the sub–annual distribution of climate variables (Lindsay et al. 1988; Charman 2002), I used a weekly time series of net rainfall and temperature data from Keighley Moor, a blanket bog in the North of England ($53^\circ\ 85'\ 31''$ N, -02° $02'\ 13''$ E) (Blundell and Holden 2015). Climate data from the University of Leeds weather station on Keighley Moor (2010–2013) suggests mean annual rainfall and temperature of 1.15 m and 7.6 °C respectively (temperature was recorded every 10 minutes at a height of 1 m: daily temperature was calculated from the mean of the maximum and minimum values. Rainfall data was collected using a tipping bucket rain gauge). Using these data, mean weekly net rainfall and temperature values were calculated with each modelled year having the same weekly inputs so that the response of the simulated peatland to gullies and gully blocking was isolated from the effect of inter–annual climate variation.

Table 6.1. Default model parameters for gully blocking simulations

Parameter	Symbol	Value	Units	Established from source
Oxic decomposition	α_{OX}	3.5×10^{-2}	year^{-1}	(Hogg 1993; Morris et al. 2011a)
Anoxic decomposition	α_{ANOX}	10^{-5}	year^{-1}	(Morris et al. 2011a)
Temperature sensitivity*	Q_{10}	3	–	(Helfter et al. 2015),
Dry bulk density	ρ	100	kg m^{-3}	(Wallage and Holden 2011)
Hydraulic conductivity†	K	1.5×10^{-3}	m s^{-1}	(ET et al in Press XXX)
Drainable porosity	s	0.3	–	(Holden et al. 2001)
Peat column size	Δx	2	m^2	–
Pond depth	–	2.5×10^{-3}	m	–

* applied to both oxic and anoxic decomposition parameters

† Initial value of K before a new layer undergoes decomposition

Model parameters were chosen within the range of previously reported values (Table 6.1) that had been shown to accumulate a mean peat depth of $\approx$1.5 m on a 50 m hillslope over 5,000 years, using the climate inputs and version of **`DigiBog`** described above (and discussed in detail in Chapter 5). To investigate the impact of gully blocking on peat accumulation and peatland hydrology in the long term, the gullies modelled here were set perpendicular to the 2D peatland as shown in Figure 6.2. Gullies were simulated by the removal of peat layers from a column after 4,000 years, and a Dirichlet constant water flow condition set at half column height. After 4,100 years gullies were blocked with dams spaced at 2 m, and the simulations allowed to continued for

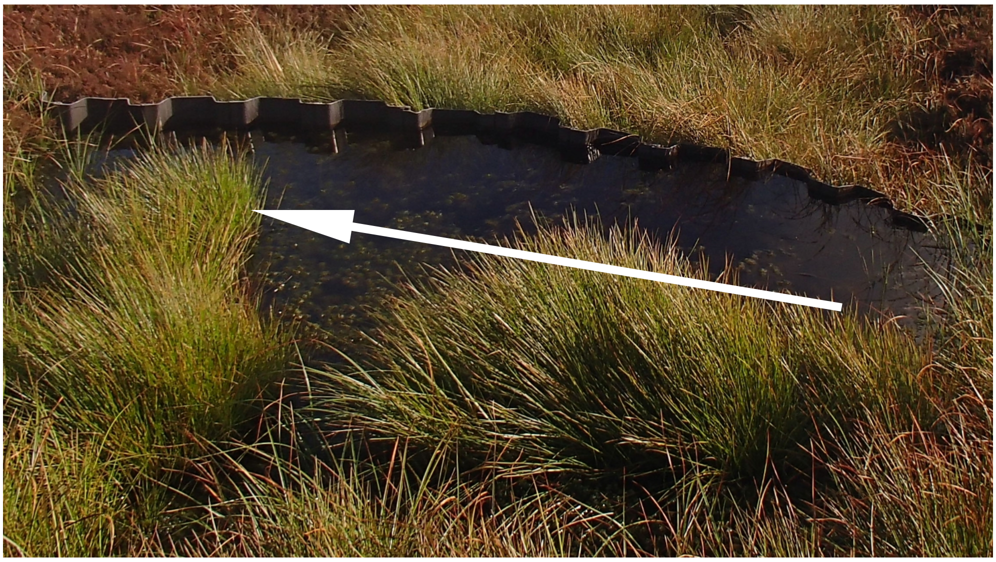


Figure 6.4. Gully infill on Keighley Moor, Yorkshire, UK. The white arrow indicates the direction of water flow in the model presented here (i.e. perpendicular to the gully). Photograph courtesy of Antony Blundell, University of Leeds.

another 200 years (Figure 6.3): a selection of gullied and blocked configurations were modelled for hillslope and plateau models (Table 6.2). Gully blocking was simulated by reestablishing a near–surface water table that was either fixed or increased with the height of the downslope gully side as time progressed. The latter approach was taken to simulate the sediment infilling and vegetation encroachment (Charman 2002, page 150) between gully dams that has been observed in some locations (Figure 6.4). The gully in one hillslope and one plateau model was not blocked (i.e. after 4,000 years of peat development a gully was created that remained open for 300 years). I assumed that the gully dams were impermeable (e.g. plastic piling), and therefore did not simulate water flow along the gully section. Although some gullied systems have begun to revegetate without intervention, because drainage has been impeded and colonisation by plants can occur (Parry et al. 2014), this process is not simulated here.

Table 6.2. Gully and blocking model configurations*

Peatland base	Blocked/not blocked	Block type
Hillslope	Blocked	Infilling
Hillslope	Blocked	Fixed
Hillslope	Not blocked	–
Plateau	Blocked	Infilling
Plateau	Not blocked	–

* gullies were located at 26 m from the modelled peatland margin

6.4 Results

Gully blocking prevented the continued loss of peat by oxidation as a result of gullying, and the balance between peat accumulation and decomposition was reversed in favour of accumulation (Figure 6.5). However, the loss of peat caused by the gully over 100 years did not recover following 200 years of gully blocking: most peat loss occurred during the 100 years following gully creation. Losses due to decomposition (i.e. ignoring peat removed from the gully column) for the hillslope model were 16% (of mean peat depth), rising to 23 % when the gully was allowed to remain open for a further 200 years (Figure 6.5): the plateau model lost 29 and 37 % of mean peat depth for the same time periods. Following simulated gully blocking, peat accumulation varied according to the type of gully block, whether the peat column was up– or downslope of the gully, and proximity of a column to the gully (Figure 6.6). For the hillslope model, peat accumulation was greatest using the infilling gully block: after 200 years, losses were reduced to 9 and 12 % of pre–gully mean peat depth for the infilling and fixed gully blocks respectively (Figure 6.5). Only the infilling gully block method was simulated for the plateau model which reduced losses to 26 % of the pre–gully mean peat depth.

The hillslope model lost the greatest amount of peat from the columns directly adjacent to the gully (i.e. the gully sides, Figure 6.6; C13 (downslope) and C15 (upslope)). These columns lost 30 and 35 % of their original depth before gully blocking and 33 and 41 %respectively if the gully remained open for 300 years. After gully blocking, losses were reduced to 18 % (infilling gully block) and 21 % (fixed block) for the downslope gully side, and 25 and 31 %respectively for the upslope gully side. As the downslope distance from the gully increased, peat accumulated to depths that equaled (column 7, 12–14 m) or exceeded (column 1, 0–2 m) pre–gully depths (Figure 6.6; C1 and C7). However, columns upslope of the gully accumulated new peat at a much slower rate after gully blocking and peat depth continued to decline (Figure 6.6; C20 and C25). For example, prior to gully blocking, column 20 (38–40 m, i.e. 10–12 m from the upslope gully side) had lost 12 % of peat depth which increased to 15 or 17 % (infilling and fixed block respectively) at the end of the simulations: although by this time the rate of loss was < 0.1 % year^{-1}.

Figure 6.7 shows the effect of the hillslope gully and the two gully blocking methods on water tables. Prior to gully creation, mean annual water–table depth for all columns was 0.04–0.07 m. Water tables across the hillslope rapidly became deeper on gully creation (falling to > 0.6 m for gully–side columns) but when the gully remained open, recovered to 0.08–0.16 m by the end of the simulation (i.e. 4,300 years) (Figure 6.7a). When the infilling gully block was simulated, downslope and both gully–side columns maintained near or at surface water tables whereas upslope water tables were slightly deeper at 0.06–0.08 m (Figure 6.7b). Peat accumulation outpaced the height of the fixed gully block after ≈50 years, and water tables were 0.06–0.09 m ≈300 years after gully blocking, which were similar to depths before gullying occurred, but the spatial distribution of water–table depths across the hillslope had changed (Figure 6.7c).

Secondary decomposition (Tipping 1995) occurs when peat that has been previously well preserved, under predominantly anoxic conditions, is exposed to high rates of oxic decay. Rapid lowering of water tables due to gully creation, has reproduced this effect in the modelled peatland

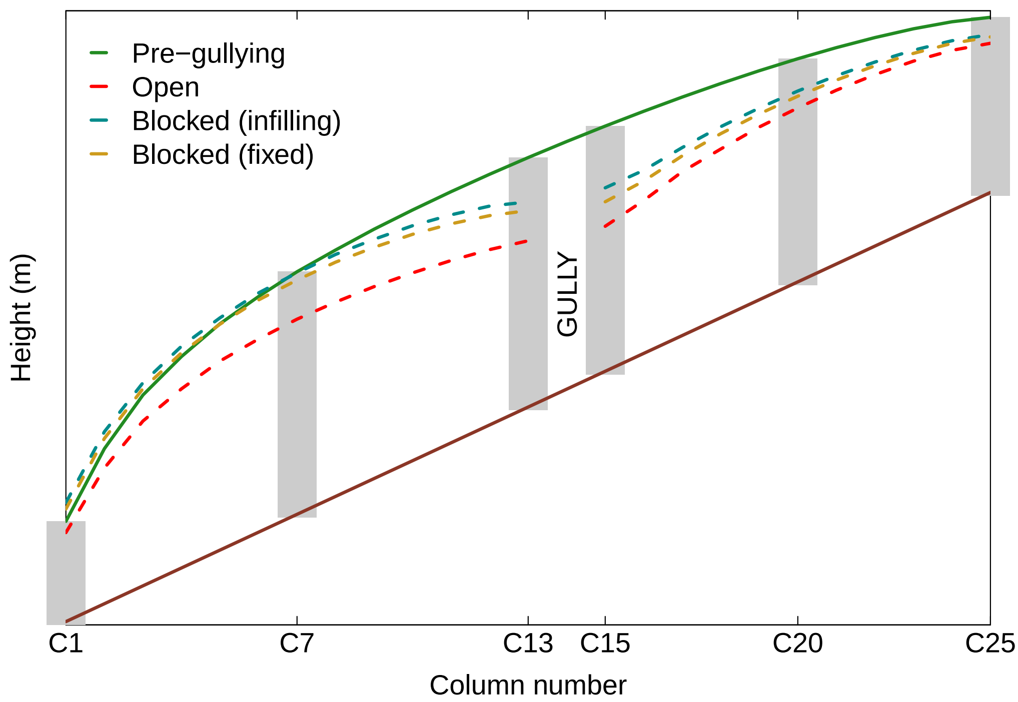


Figure 6.5. Modelled peatland surface pre– and post–gully blocking. Peat surfaces are plotted from mid–column heights, each simulated peat column was 2 × 2 m. Gullied peat heights represent the peat surface after 4,300 years. Shading indicates the location of peat columns discussed in detail. The maximum depth of peat accumulated was 1.5 m.

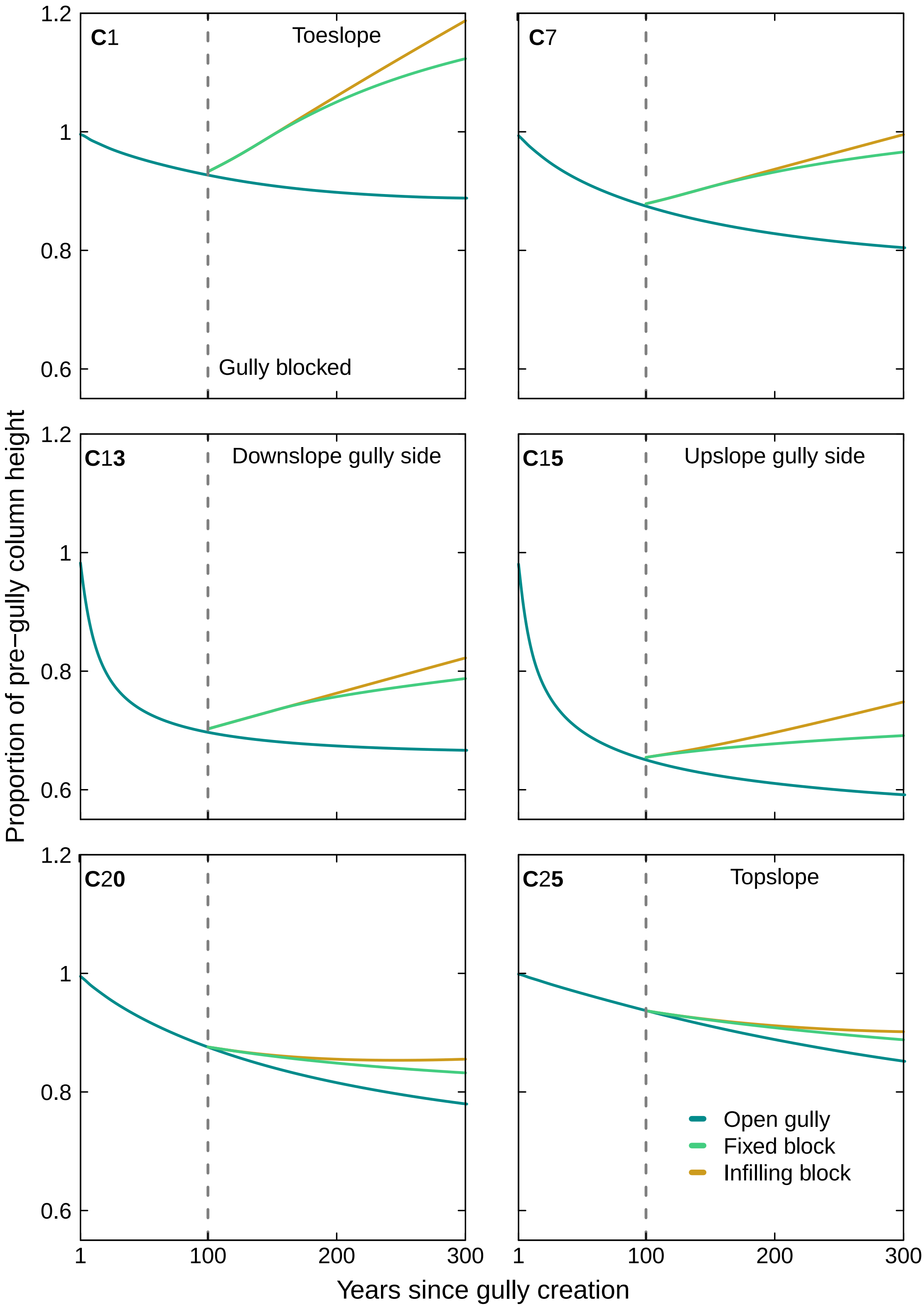


Figure 6.6. The effect of a gully and gully blocking on peat accumulation across a hillslope. Each panel represents one of 25 hillslope columns of dimensions 2 m $\times$ 2 m $\times$ h (where h is the accumulated peat height in m for a column). The gully was established in column 14 (at 26–28 m) and the depth was 0.5 $\times$ h $\approx$ 0.75 m in the example shown. Peat accumulation took place for 4000 years prior to the creation of a single gully. The gully was blocked after 100 years (shown by dashed vertical line). The simulation was allowed to continue until 4300 years had elapsed.

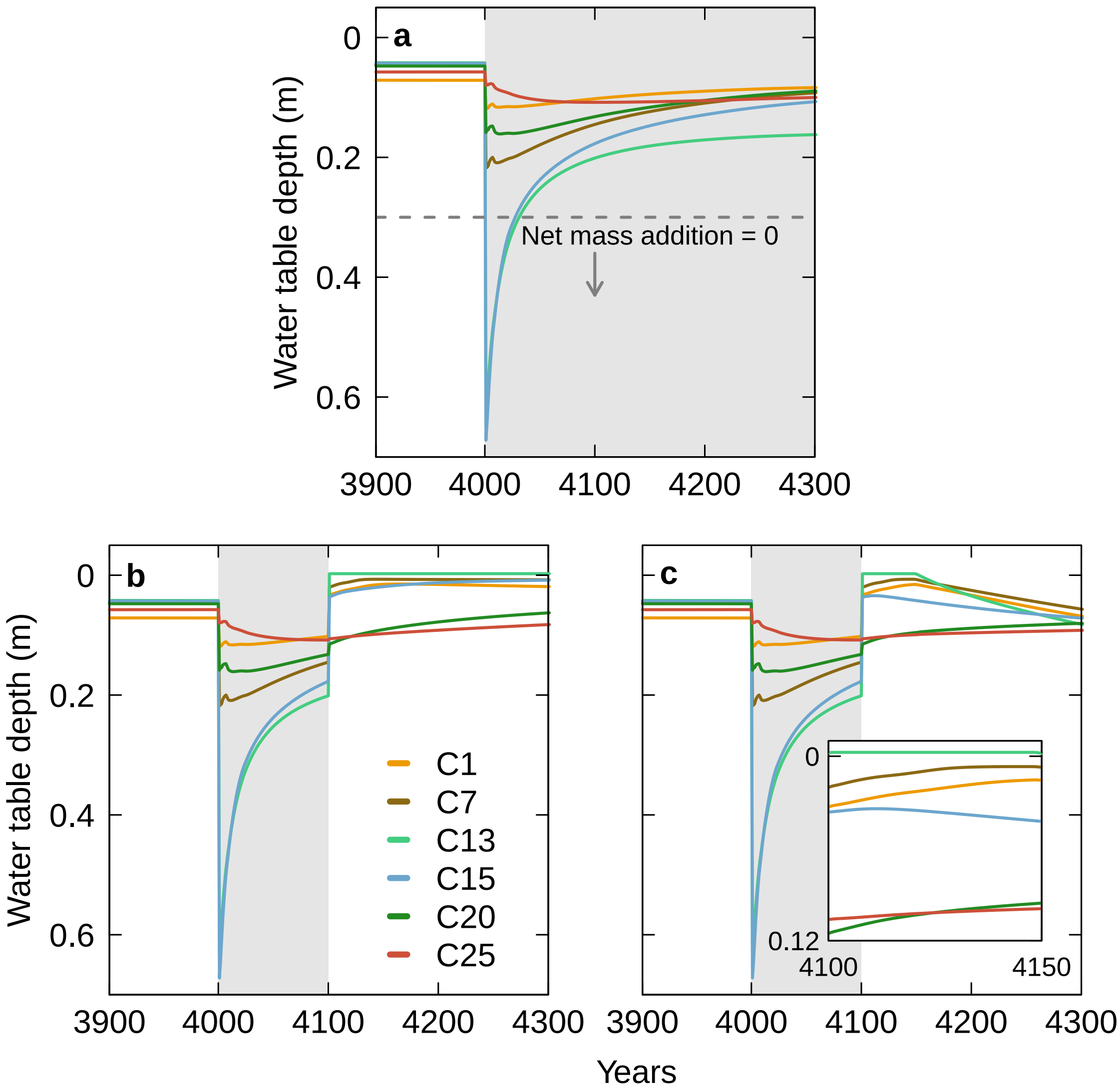


Figure 6.7. The effect of a gully and gully blocking on hillslope water tables. **a** Open gully, **b** blocked gully where the gully height was increased with increasing downslope peat height to simulate infilling (i.e. column 13 at 24–26 m), and **c** blocked gully with fixed height (inset shows water tables for 50 years after gully blocking). The gully was bocked after 4100 years. C1 (0–2 m) is the toeslope column and C25 the drainage divide at 48–50 m. C13 and C15 are downslope and upslope gully side columns respectively. The shading represents the time the gully remained open. No peat is accumulated when water tables are > 0.3 m.

(Figure 6.8a). Whilst the signal is most evident in gully–side peat accumulated approximately 2,000 years prior to gully creation (seen as a sharp drop in mass remaining θ), the effect can also be observed in the topslope column approximately 1,000 years before gully creation. Less decomposed peat accumulated in the saturated conditions that immediately followed gully blocking in all but the two columns upslope of the gully–side (C20 and C25, Figure 6.8b and c).

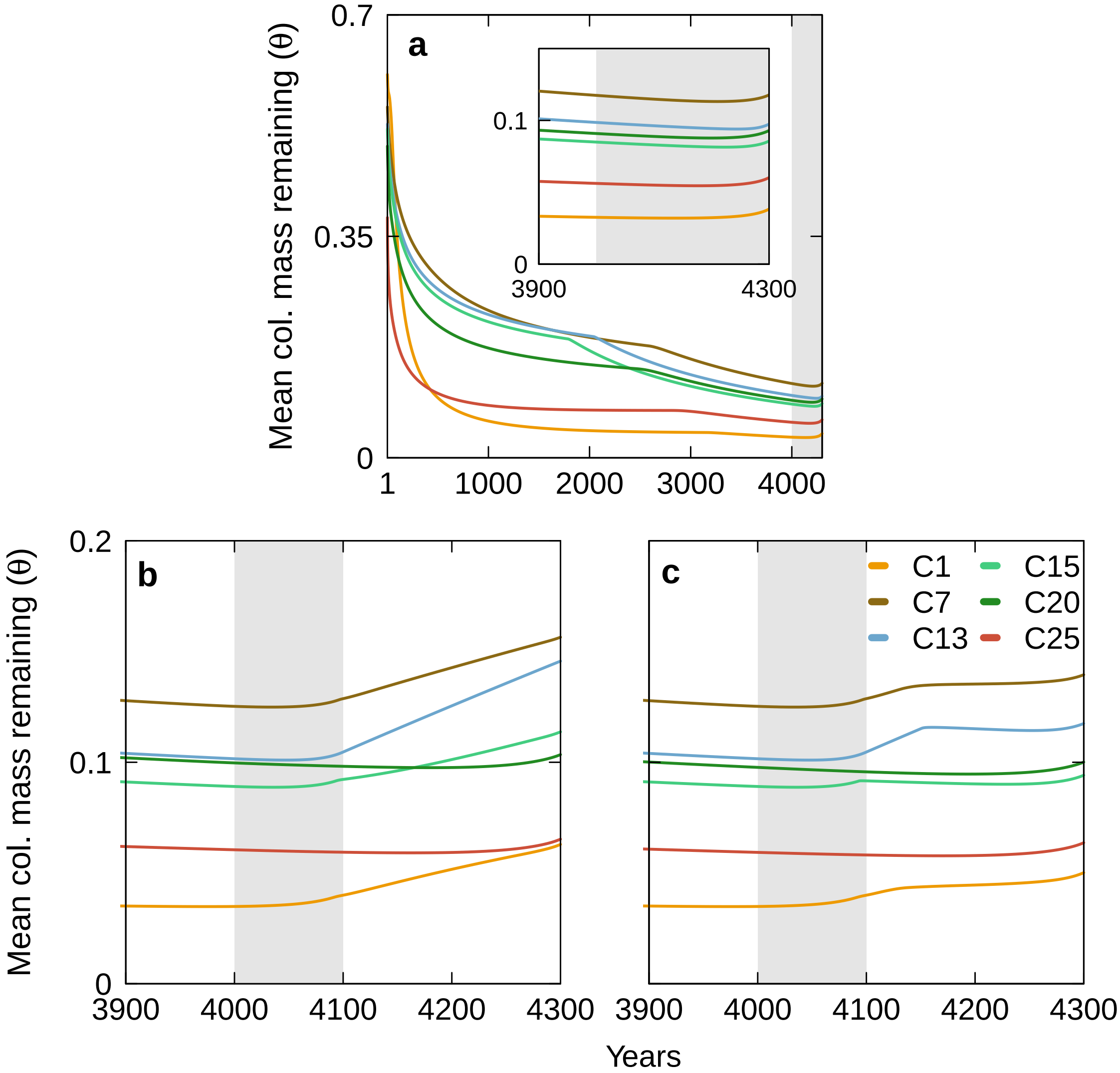


Figure 6.8. Peat column decompostion: hillslope model. a Open gully (inset decomposition between 4000 and 4300 years), **b** blocked gully where the gully height was increased with increasing downslope peat height and **c** blocked gully with fixed height (refer to Figure 6.7 for a description of peat columns C1–C25). Panels **b** and **c** are equivalent to the inset in **a**. Shading represents the time the gully remained open. The effect of the gully on the oxidation of peat can be seen as secondary decomposition of older well–preserved peat, evident in panel **a**, as the sharp decrease in θ.

6.5 Discussion

This chapter applied the version of **DigiBog** developed in Chapter 5 to a peatland restoration process that was identified from the analysis of a cognitive model, and during workshop discussions with peatland stakeholders. Gully blocking is commonly used in blanket peatland restoration with the aim of creating shallow water tables and encouraging typical peatland plant communities (Green et al. 2014; Blundell and Holden 2015), but the process can be contentious because of the perceived negative impact that shallow water tables may have on livestock and grouse.

The simulations reported here differ in two ways from previous studies that investigated the effects of gullies and gully blocking (e.g. Allott et al. 2009; Clay et al. 2012; Dixon et al. 2014). (1) Peat accumulation and water tables are simulated from the interaction of soil properties and the flow of water through the soil matrix in a dynamic 2D model. A set of ecological and hydrological feedback mechanisms govern the addition of plant litter and decomposition of peat according to temperature and water–table position. In turn, water–table position is determined by the flow of water between columns of peat, which is coupled back to decomposition through changes in hydraulic conductivity, and the exchange of water from climate inputs. (2) The modelled peatland was used to investigate gullying and subsequent gully blocking over 300 years, whereas the findings of empirical studies (including those that investigate the effect of drains in blanket peatlands) are restricted to very short periods of time (often < 15 years) (e.g. Holden et al. 2011; Bellamy et al. 2012; Dixon et al. 2014; Pilkington et al. 2015). Both differences can help provide new insights into the challenges of understanding the impact of gullies and gully blocking highlighted in empirical studies. These difficulties include; (1) there are differences in water tables between sites that have received the same restoration treatments (e.g. Allott et al. 2009; Pilkington et al. 2015), and (2) water tables have responded to revegetation and/or gully (or drain) blocking but more time is needed to assess if they are likely to achieve the shallower depths and smaller temporal variation of intact sites (e.g. Wilson et al. 2010; Wallage and Holden 2011; Holden et al. 2011; Bellamy et al. 2012).

The hillslope model shows how peat columns and water tables respond spatiotemporally to gully and gully blocking when soil properties are taken into account (Figures 6.6 and 6.7). Gully creation causes rapid deepening of gully–side water tables (0.63–0.67 m for downslope and upslope gully–side columns respectively) leading to secondary decomposition of older previously saturated and well–preserved peat (Tipping 1995), and lower hydraulic conductivity (K) across the hillslope (supporting the suggestion of Holden et al. 2006). Because of slope and varying decomposition, the response of the peatland to gullying is non-linear. The margin of a blanket peatland is highly decomposed (Lapen et al. 2005; Lewis et al. 2012) (also shown in Figure6.8a; C1), but the newly formed upslope gully side is well-preserved. The columns of peat that became the side walls of the gully (C13 and C15) developed under shallow water tables in the peatland centre, and have higher transmissivity ($K \times h$) than columns at the margin, which means that well–preserved columns transmit water more quickly. As a result, when the gully is created, upslope columns initially drain more quickly into the gully than downslope columns drain from the margin and therefore, upslope columns are subjected to a greater degree of secondary decomposition (Figures 6.6 and 6.8a; C15, C20 and C25).

Although gullying takes place 300 years before the end of the simulations, signals of increased decomposition are found in gully–side peat formed around 2,000 earlier, and not at the time of gullying (Figure 6.8a). Therefore, the onset of erosion could be misinterpreted from humification records, and precede other markers (e.g. plant macrofossils) used to identify events that cause deeper water tables to occur (Borgmark and Schoning 2006; Morris et al. 2015a). Using a 1D version of **`DigiBog`**, Morris et al. (2015a) also showed that signs of increased decomposition in peat due to climate change, can be offset from the events that caused them as a result of secondary decomposition. However, because the model used here links peat columns hydrologically, my results also demonstrate how the signals of gullying in blanket peatland humification records are also likely to vary in relation to the location of the gully, and hillslope position (Figure 6.8a).

The water table depths shown in Figures 6.7a–c should be interpreted in relation to the loss of peat across the hillslope: the overall height of the peatland decreases during gullying (Figure 6.5). Annual water–table position is a result of the combined effect of decreasing peat column height because of oxidation, and the negative feedback mechanism between increased decomposition and decreased hydraulic conductivity that causes simulated water tables to rise. However, when the gully remains open, the peatland shifts to a drier state with deeper water tables (Figure 6.7a) consistent with the observations made of intact and gullied peat sites by Allott et al. (2009). In common with empirical studies into the effect of peatland drains (e.g. Holden et al. 2006; Wilson et al. 2010), the peat column downslope of the gully (C13) remains drier even after 300 years.

Within 50 years of gully creation, the mean annual water–table depth of gully–side columns is < 0.3 m and < 0.2 m for other columns, which in an intact system provides the conditions suitable for net litter addition (Figure 6.7a; dashed horizontal line), but the gullied peatland continues to degrade because of drainage into the gully. In contrast to many degraded blanket peatlands in the South Pennines, the model used here assumes peat surfaces are vegetated. Given the continued loss of peat in vegetated open gully systems described above, this result supports the conclusions of Dixon et al. (2014) that revegetation alone is not sufficient to restore the hydrological function of blanket peatlands that are sources of carbon. However, Figure 6.6 shows that the rate of peat lost in the open gully system is much lower after 300 years, and in some columns has almost stopped. I suggest that where this occurs, net accumulation will restart without gully blocking even though the water–table regime is likely to be different from an intact peatland.

Peat columns show a varying response to gully blocking, which is also dependent on hillslope position. In downslope and gully–side columns peat accumulation is restarted by gully blocking. This effect is less pronounced above the gully where losses from increased decomposition are only stopped or slowed in the top 12 m of the hillslope (C20–C25). Figure 6.8b and c clearly show that downslope peat accumulated after gully blocking is less decomposed (increased θ) because of the imposed shallower water tables, whereas new peat accumulated upslope of the gully shows no change in the rate of decomposition. Interestingly, the simulations suggest that the rate of peat accumulation after gully blocking increases with increasing downslope distance from the gully. The toeslope column accumulates peat to a depth greater than before gullying (Figure 6.6; C1). This may be because net litter addition is a balance between production and decomposition, and the slightly deeper water tables in the toeslope column (C1) result in greater increases in peat thickness than columns with near surface water tables (e.g. Figure 6.7c, C13), where there is less decomposition but also less production. Water–table regimes in modelled peat show greater variation across the hillslope after the gully is blocked than before gullying took place (Figure 6.7b and c): i.e. gully blocking did not re-establish previous water–table dynamics on decadal timescales. These results are supported by the observations of Holden et al. (2011), who found greater spatial variation in peatlands where there were open and blocked drains, when compared to intact sites.

These simulations also show that because peat formed downslope after gully blocking remains saturated, hydraulic conductivity is higher than in upslope columns. This finding is supported by Wallage and Holden (2011) who also observed higher values of hydraulic conductivity after ditch blocking. This effect is also related to the length and angle of the hillslope, the position and depth of the gully, and the height of the gully block, and can therefore be expected to vary from site to site. In the case of the fixed block, rates of accumulation decrease as the difference between peat surface and gully block height increases, leading to increased decomposition of surface layers. As a result, by the end of the simulation, water table depths were slightly deeper than the intact peatland (Figures 6.7c and 6.8c). Ultimately this difference between the height of peat and the height of water in the blocked gully becomes a limiting factor in the recovery of peatland carbon storage (Figure 6.6).

The model used here is necessarily a simplification of peatland dynamics and has a number of limitations which includes a constant value for bulk density and drainable porosity. There are conflicting reports of how bulk density varies in blanket peats: for example Lewis et al. (2012) report increased bulk density at the margin of a blanket bog where peat is more decomposed (although they suggest this is due to compression from forces within the bog rather than increased decomposition) but found no significant variation with depth, contrary to the findings of Parry and

Charman (2013). Studies have reported that drainable porosity in blanket peatlands changes with depth and with proximity to the bog margin. For example, Holden et al. (2001) recorded values of porosity that were higher in the top 0.1 m (0.55) than between 0.1–0.25 m (0.35); and Lewis et al. (2012) reported higher values of porosity at the centre than at the margin of a blanket peatland (where decomposition is greater). These data suggest that there is a relationship between porosity, decomposition and possibly compression (i.e. bulk density), and not simply depth. However, as this relationship is currently not well specified, constant values were used (Morris et al. 2011a). In addition, the model did not simulate the continued loss of peat from erosion and the height of gully water was held constant throughout the year: the former may underestimate peat losses and the latter may overestimate the response to gully blocking. Whilst further work is needed to determine how these parameters might be incorporated into the model, the simulations nevertheless produced results that were consistent with a number of empirical studies. Furthermore, even though the model was used to explore the effect of a 2 m gully, the approach used here is also applicable to drainage ditches (Daniels et al. 2008; Baird 2014).

My results suggest that; (1) both gully blocking and revegetation are needed to arrest long term losses of peat at sites that are actively losing peat; (2) peatlands are unlikely to recover peat lost through increased oxidation even two centuries after gully blocking; (3) restoration managers should not assume that the water–table regimes of peatlands with blocked gullies can be compared to intact sites because they are likely to show greater variation, and be different, from intact sites for decades if not centuries because of increases in hydraulic conductivity caused by secondary decomposition (as well as because of site–to–site variations such as slope); (4) where practicable, gully dams should be set as close to the top of the gully as possible; and (5) in revegetated systems, gully blocking is likely to impose near–surface water tables in peat downslope of the blocked gully.

The model reported in this chapter provided new insights into how carbon storage and water tables in a blanket peatland may vary in response to gullying and gully blocking over annual to centennial timescales. The simulations show how secondary decomposition of previously well preserved peat, that forms newly created gully sides, causes peat upslope of the gully to decompose more rapidly than downslope peat. Although the investigations reported do not relate to a specific site, and simulate a gully that is parallel to a main gully, they demonstrate how a model such as **`DigiBog`** can be used to inform the management and restoration of gullied or drained blanket peatlands. This study highlights the importance of taking into account peat properties when undertaking assessments of the likely effect of restoration on water tables and carbon storage. To inform restoration management at specific locations, future modelling should explore; experiments with current and predicted climate variables; site topography; different gully types and orientations in 3D; and field observations of peat properties such as hydraulic conductivity and bulk density.

Chapter 7

Synthesis and conclusions

7.1 Introduction

Three converging themes formed the conceptual framework of this thesis. (1) Global land–use pressure has resulted in degraded semi–natural systems where the continued exploitation, and restoration of damaged ecosystem function is often contested. (2) Many ecosystems are complex systems that are the product of human and natural interactions. The ability of complex systems to respond adaptively to external forcing can make it difficult to understand and predict the impact of land use or climate change, and challenging to address issues of decline and degradation (e.g. Reynolds et al. 2009). (3) Decisions about future land use and repair of ecosystem function should be informed by local, practitioner and scientific knowledge, brought together in bottom–up participatory processes (Reed et al. 2007; Reynolds et al. 2009). There is a drive to repair degraded ecosystems because of the negative impact of land use on human well–being, and biodiversity, and concerns that poorly functioning ecosystems could exacerbate the impact of climate change (Martin and Watson 2016).

The conceptual framework of this thesis can be applied to many peatlands throughout the world: although peatlands are globally important for carbon storage and biodiversity, the direct or indirect impact of human activities has converted these long–standing terrestrial stores into sources of aquatic and atmospheric carbon (e.g. Moore et al. 2013; Turetsky et al. 2015). A current example of this degradation is in the draining and burning of tropical peat swamps for agriculture (Gaveau et al. 2014). Land use can also exacerbate the depletion of peatland carbon stocks caused by natural erosion processes (Tallis 1985). However, conflict at local scales is common where attempts to repair peatland ecosystem function appear to threaten livelihoods or traditions (e.g Noordwijk et al. 2014). Blanket peatlands in the UK are one example where concerns over the negative impact of land use on carbon stores, biodiversity and water quality, have triggered a switch in emphasis from peatlands as agricultural landscapes, to a focus on the conservation and restoration of ecosystem functions (e.g. carbon sequestration) to secure wider societal benefits such as climate change mitigation (Maltby 2010). However, this switch has seen conflict develop between traditional peatland users and proponents of blanket peatland conservation and restoration (Maltby 2010; Reed et al. 2013b).

I used this conceptual framework to address the overall aim of this thesis which was; "to develop new knowledge about blanket peatlands as a complex system, and to enhance our current understanding of the impact of social and ecological interactions on carbon storage". My research maps across the three components of the framework in the following way (Figure 7.1): (1) blanket peatlands in the South Pennines, UK, provided the location of the study area as an example of a degraded ecosystem where the aims of peatland restoration and future land uses are often contested; (2) I reasoned that because peatlands are complex systems (Belyea and Baird 2006) with many ecological, hydrological, and human interactions, that develop over long timescales, models were needed to enhance our understanding of the impact of these interactions on carbon storage. However, these models also need to be useful and accessible to stakeholders, if they are to play a role in determining how land–use objectives could be achieved; (3) Two models were used. First, peatland stakeholders co–developed a fuzzy cognitive map (a network of causal interactions), that incorporated knowledge from land user, practitioner and scientific sources, and was used in a workshop setting to propose how to achieve land–use objectives and to assess their impact on carbon storage. Secondly, I developed a new version of a process–based peatland development model to explore the impact on carbon storage, over centennial to millennial timescales, of two of the concepts from the fuzzy cognitive map.

7.2 Synthesis

7.2.1 Engaging stakeholders using networks

Addressing issues of contested land use is a difficult challenge, and is often described as a wicked problem (*sensu* Rittel and Webber 1973; Ritchey 2013) because the clash between conservation and livelihood objectives can seem intractable (Biggs et al. 2011; Redpath et al. 2013). If all relevant stakeholders groups should be involved in collaborative discussions about future land use (Kates et al. 2001; Young et al. 2005; Soliva et al. 2008; Maltby 2010), then suitable approaches need to be found that facilitate engagement. Studies have emphasised the need to understand the interactions between both social and ecological components of coupled systems (Kates et al. 2001; Reynolds et al. 2009; Liu et al. 2015a). I show how the structural properties of the network of interactions created from the aggregated mental models of peatland stakeholders can be developed (Chapter 3) and used (Chapter 4) within a participatory process to propose and discuss how land–use objectives could be achieved. The co–developed fuzzy cognitive map was a directed weighted network.

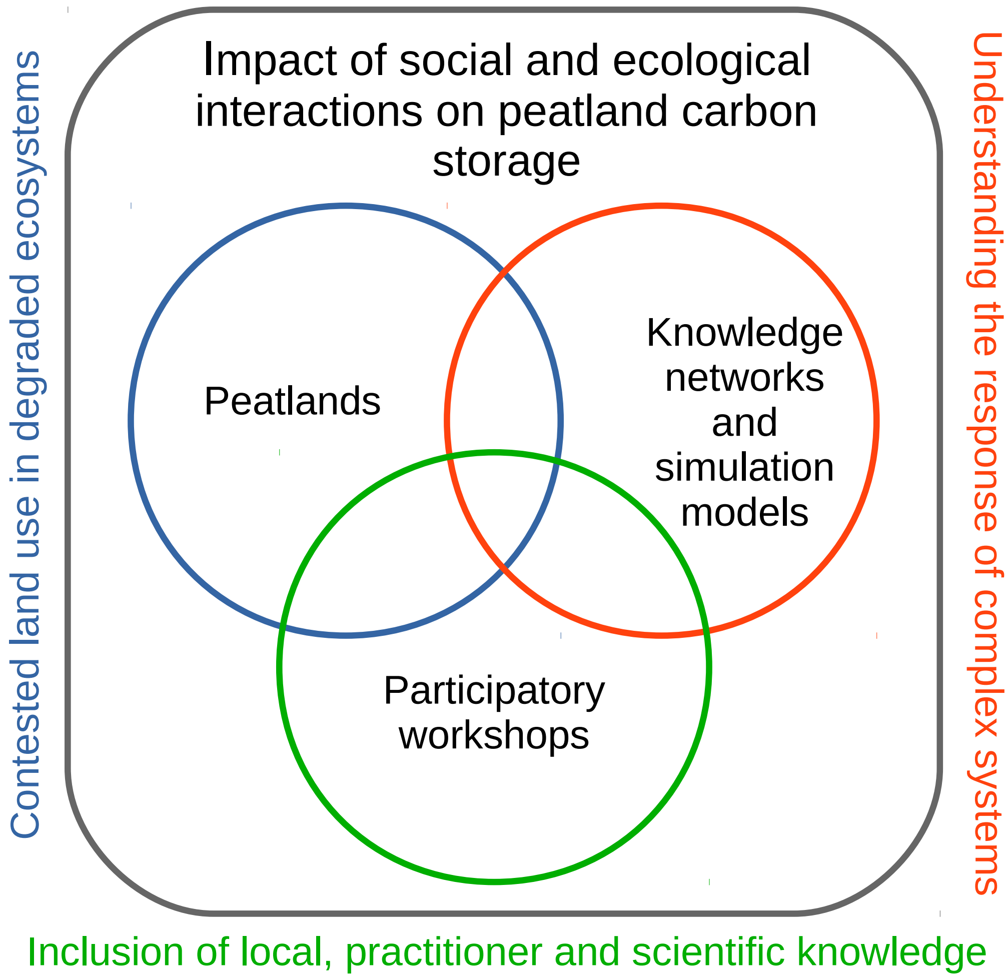


Figure 7.1. Conceptual framework used to address the aim of this thesis.

Networks are suited to stakeholder engagement (Pocock et al. 2016) because they are intuitive, can be developed from quantitative (Kawakami et al. 2016) and qualitative (Penn et al. 2013) sources, and the process used to acquire cognitive models can be relatively easily adapted for different stakeholder groups (Chapter 3).

Network development

I suggest that the approach used in Chapter 3 to obtain a number of mental models, from a wide variety of sources, not only increases the reliability of the final model (Kosko 1988), but also enables a first–order network to be built from stakeholder knowledge without early disagreement between stakeholders with competing land–use interests (c.f. Dougill et al. 2006). From this initial model, differences between stakeholder understanding of interactions can be identified, and some of the information about the network can be presented and discussed with stakeholders. Although the visually impenetrable structure of some networks (e.g. Figure 7.2b) have been referred to as "hairballs" (Krzywinski et al. 2012), such diagrams can help convey the complexity of the system in question, and can be simplified using clustering algorithms (Chapter 3), or presented in different ways depending on the objective of the discussion (e.g. Figure 7.2) (Pocock et al. 2016). Most importantly, regular early feedback can be given to participants by reviewing what has been found, which may promote continued engagement (Reed et al. 2014b). Findings about some of the structural properties of the network such as the number of concepts and connections, and the most connected concepts, as well as network diagrams, and how the data will be used in subsequent stages of the project, can be easily shared and discussed with participants.

Chapter 3 answered the first research question, "what are the key interacting social and ecological factors that are needed to represent blanket peatlands as a complex system?" By overlapping the mental models of blanket peatland stakeholders, a fuzzy cognitive map (FCM) of 126 social and ecological concepts, interconnected by 524 interactions, was produced (Figure 7.2b). This model builds on the work of Dougill et al. (2006), Prell et al. (2007) and Reed et al. (2013b) to co–develop a blanket peatland FCM – a network of directed weighted social and ecological interactions based on stakeholder perceptions: as far as I am aware this represents the first example of a blanket peatland FCM. The process provides an example, applicable to many social–ecological systems, of how important knowledge from conflicting viewpoints can be integrated and validated to define the structure of ecosystem interactions in a useful way. By focussing on the pairwise interaction of concepts, instead to the expected outcomes of the system, differences between the direction of interactions (i.e. caused an increase or a decrease) could be discussed and agreed. Many of these are the local interactions from which the properties of a complex system arise (Levin 1998; Belyea and Baird 2006).

Studies have included the group validation of networks developed in participatory processes: for example, Penn et al. (2013) asked a largely different group of stakeholders to verify a fuzzy cognitive map as a secondary check on the concepts and interactions to address possible biases included by the model builders. But I suggest that in cases of contested land use, group validation of how concepts interact could become the process by which stakeholders work towards consensus, or to identify areas of firm disagreement, and may help to increase model ownership. This process differs from Christen et al. (2015), who also recognised that FCMs provide a mechanism to identify conflicts in stakeholders' perceptions, because the differences identified here were discussed inside a group validation process and used to review conflicting perceptions. Such an approach also provides the opportunity to identify gaps in understanding, and uncertainties about interactions (which could be the subject of future research), or for participants to challenge the viewpoints of other stakeholders. In this way, discussions about the interactions between pairs of concepts are separated from discussions about land use (which may be more difficult to resolve), and participants have the opportunity to build relationships before debating how to achieve land–use objectives. As part of the validation process, South Pennine blanket peatland stakeholders also defined a subset of concepts that were perceived to be highly important for the maintenance or increase of blanket peat, which resulted in a network of 31 concepts and 120 interactions (Figure 7.2c). Interestingly, this produced a highly connected subset that included the concepts and interactions (such as burning intensity) that reflect the currently contested debate about the negative impact of managed burning on blanket peatland carbon storage and hydrological function (e.g. Douglas et al. 2015; Holden et al. 2015).

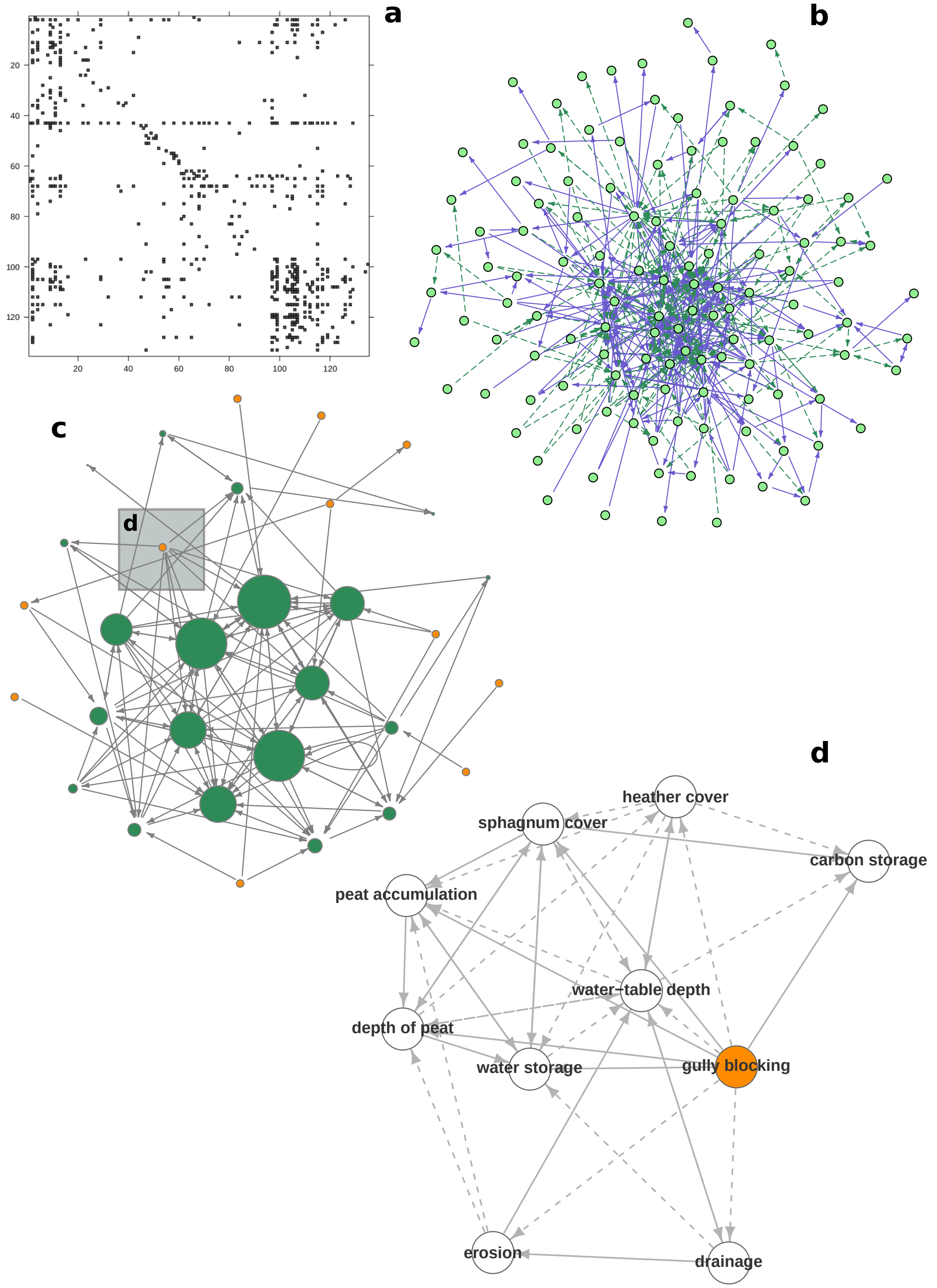


Figure 7.2. Blanket peatland network data. **a** Pre–validated adjacency matrix. Interactions run from rows to columns. **b** Validated co–developed network comprising 126 concepts and 524 interactions. Whilst visually difficult to interpret, the network has been drawn to show a dense core of interactions, and to highlight the complexity of the overall model (Chapter 3). **c** Highly important subset defined by stakeholders comprising 31 concepts and 120 interactions. Orange nodes represent driver concepts. Green (ordinary concepts) are sized according to the number of interactions (Chapter 4). The interactions of the concept in the shaded square are shown in more detail in panel d. **d** The interactions of gully blocking – shown in orange to represent a driver concept (Chapter 4). Solid lines represent interactions that cause an increase in the connected concept, and dashed lines represent interconnections that cause a decrease in the connected concept. This network was part of the highly important subset.

Network structure and participatory modelling

In Chapter 4 , I analysed the structure of the cognitive model to answer the question, "How can these factors be used to evaluate the impact of land–use objectives on blanket peatland carbon storage?" The analysis identified two classes of factor, hubs and drivers (e.g. Jeong et al. 2000; Liu et al. 2011, respecitively), that are a function of the distribution of connections within the network, and are consistent with the interaction networks of other natural and human systems. Because of their relatively high connectedness, hubs (e.g. *Sphagnum* cover) impart both robustness and vulnerability to the modelled peatland, whereas drivers (e.g. gully blocking) are the factors that can provide control of a network (Liu et al. 2011), and are distinct from hubs. Driver concepts will be present in all network structures, but the number will vary according the framework of interactions that have been identified (Liu et al. 2011). However, hubs may not be present in other co–developed networks because they are related to the number and configuration of interactions determined by stakeholders, but other structures, such as bow–ties (e.g. Kawakami et al. 2016), may be evident. The composition of groups of concepts, identified by clustering algorithms, may also reveal interesting properties of the network that can be discussed, and used by stakeholders. I applied the structural findings of the blanket peatland network to the highly important subset of factors and interactions defined by stakeholders. This analysis identified five hubs, that together connected > 60% of the network, and 11 drivers. By using these two classifications, I found that it was possible to simplify the problem of how to understand the impact of land–use objectives on carbon storage into the effect of control (drivers) on robustness (hubs) (e.g. Figure 7.2d).

To explore how this finding could be used in practice, a participatory workshop was coupled with modelling (Chapter 4) to answer the question, "How should blanket peatlands be managed to achieve the objectives of (a) maintaining or increasing carbon storage, (b) improving the quality of water supplied and, (c) supporting local livelihoods: what are the implications for current land uses?" I extended earlier participatory work (Penn et al. 2014) to demonstrate that blanket peatland network structures can used used by stakeholders to determine how land–use objectives might be achieved, and to assess the impact of the changes on carbon storage. I calculated the impact of land–use change on carbon storage and hubs using a simple model associated with fuzzy cognitive mapping (Chapter 4).

I found that the changes to drivers, proposed by stakeholders to deliver land-use objectives (a)–(c), were beneficial to the hubs that underpin peatland robustness. However, whereas the changes made to deliver an increase in carbon storage and water quality had a similar positive impact on carbon storage (also reflected in the overall increase in peat depth), those that were made to support local livelihoods had no effect on carbon storage (or peat depth) when compared to the current–state model. I showed that this was because; (1) some restoration techniques were unanimously perceived to be beneficial to livelihoods (e.g. revegetation), but those that may repair hydrological function and decrease water–table depth (e.g. gully blocking) were not; and (2) in the case of some drivers (e.g. burning intensity), the livelihood objective was perceived to be incompatible with carbon storage and water quality objectives. The combined result of (1) and (2) was that an increase in peat accumulation was offset by the interactions between natural and anthropogenic processes that remove carbon from the modelled peatland. These results suggest that although there was no 'win–win' scenario, land–use objectives could complement each other if farmers and game–keepers carried out peatland restoration, and burning intensity was reduced. These results suggest that in order to achieve these objectives those who favour peatland restoration should provide the opportunity for farmers and game–keepers to become integral to restoration processes and to contribute their knowledge and skills to restoration work and experiments.

7.2.2 Modelling complex systems

The network model used in this thesis was based on the relative interactions of pairs of concepts, and was limited to how concepts respond to changes in relation to each other. Other approaches to modelling dynamical networks could be used depending on how the interactions between concepts are specified during the construction of cognitive maps (e.g. Boccaletti et al. 2006; Vogt et al. 2015). Many networks have also been developed to incorporate empirical data about the interaction between concepts (nodes) (e.g. Jeong et al. 2000; Evans et al. 2013a; Pocock et al. 2012; Li et al. 2014; Kawakami et al. 2016), and studies of fuzzy cognitive maps have developed methods to take into account delays in the interaction between two concepts (Hagiwara 1992; Neocleoua et al. 2011), and absolute rather than relative values (e.g. Wise et al. 2012, modelled tonnes of fish landed from a fishery). However, these approaches could have made developing a model from disparate sources of stakeholder knowledge more difficult or even unlikely because of the time required in workshops, lack of a common language about interactions (Pocock et al. 2016), and the need to specify more complex relationships for pair–wise interactions.

The specification of interactions should be usable by all participant groups (either directly or indirectly) for model building and validation purposes, because although it is possible to specify more complex relationships between concepts, they may sacrifice transparency or ownership by stakeholders. One of the most important goals of participatory modelling in situations where land use is contested has to be to engage stakeholders. Whilst I recognise that the simple method used here, to model land–use objectives, has a number of limitations (Chapter 4), coupling network structure with discussions about land uses appears to be a promising approach to stakeholder engagement where land use is contested. Nevertheless, because of the limitations, a second approach to modelling blanket peatlands as complex systems was needed to address questions about how driver concepts affect carbon storage over centennial to millennial timescales.

Prell et al. (2007) suggested that models from different disciplines should be better integrated in order to explore the interactions in social–ecological systems. But getting the balance right between accessibility and usefulness for a broad range of land–use stakeholders, and modelling the future state of complex systems beyond that which has been previously observed is challenging, and likely to require different approaches. This is not to suggest that one approach is better than another, merely that models have different purposes (Evans et al. 2012). This thesis integrates the structural properties of a network of social and ecological interactions, which is often how complex systems are described (e.g. Reynolds et al. 2009), with a peatland development model that predicts future states by allowing the response of the system to emerge from small–scale local interactions (Levin 1998; Belyea and Baird 2006; Evans et al. 2012).

Blanket peat accumulation over centennial to millennial timescales

Peatlands play an important role in the exchange of greenhouse gases within the global climate system (Charman et al. 2013), but land use can also alter the internal processes that govern this exchange (e.g. Turetsky et al. 2015; Petrescu et al. 2015). Studies use palaeoecological analyses, direct measurement and models to understand these effects, especially in the light of predicted climate change (e.g. Ise et al. 2008; Gallego-Sala et al. 2010; Moore et al. 2013; Blundell and Holden 2015; Charman et al. 2015). Because statistical methods, that are based on current conditions, are unlikely to be helpful when predicting unfamiliar futures (Evans et al. 2012), process–based models of peatlands as complex systems are one of the key approaches that can be used to understand how future climates and land use may interact to affect carbon stocks in the long term. The important small–scale processes (Holden 2005b) that can be represented in peatland development models include the changes to peat structure that mediate the movement of water, and the interactions which, in turn, affect litter production and decomposition, and ultimately the fate of present and future carbon stocks.

Using the **`DigiBog`** 1D peat accumulation and 2D hydrological models (Baird et al. 2011; Morris et al. 2015a), I developed a new 2D model to represent blanket peat accumulation to answer the question, "What is the predicted impact of social and ecological factors on the centennial to millennial storage of carbon in blanket peatlands when conceptualised as a complex system?" This model represents the first attempt to simulate 2D blanket peatland development on slopes using hydrologically connected columns of peat that incorporates a feedback mechanism between water tables, decomposition, and hydraulic conductivity. To show how a process–based model can enrich the results of the network–based model, I investigated the impact of two drivers from the highly important set of concepts. The simulations produced two original studies that enhance our understanding about modelling the long–term development of blanket peatlands in response to climate variables, and gully blocking (Chapters 5 and 6 respectively).

In Chapter 5, I show that the temporal resolution of climate variables (i.e. rainfall and temperature) is a key consideration for modellers who aim to understand the impact of climate on blanket peatland carbon storage. Previously, peatland development models have used average annual climate variables (e.g. Frolking et al. 2010; Swindles et al. 2012), but this is likely to overestimate the amount of peat accumulated because some winter net–rainfall, that would be lost to streams via overland flow in real blanket peatlands, is redistributed to months where rainfall is typically much lower and temperature higher. This leads to a change in the balance between litter production and decomposition that favours increased peat accumulation. Simulations, run over 5,000 years, that used annual totals for rainfall and temperature produced average peat depths that were 1–1.5 m greater than weekly or monthly resolutions of the same climate data.

Because blanket peatlands are sensitive to the distribution of rainfall and temperature (Lindsay et al. 1988; Charman 2002), I recommend that peatland development models are parameterised with with weekly climate variables, so that winter rainfall that exceeds the storage capacity of the modelled peatland, is lost from the model domain, and higher mid–year temperatures have a greater effect on peat decomposition when water tables are likely to be deeper. Peatland development models are sensitive to the temporal resolution of rainfall and temperature inputs, and modellers should take this into account when aggregating climate variables. Although I did not model any specific future climate scenario, the droughty summer conditions imposed by the monthly resolution of rainfall and temperature suggest that peat accumulation is likely to continue at a reduced rate in intact blanket peatlands. However, the mean temperature of the warmest month did not exceed the apparent temperature threshold for blanket peatlands of 14.5°C (Gallego-Sala et al. 2010). The continued accumulation of peat was due to the resilient response of the modelled peatland to the dry conditions. Average annual water tables were shallower in simulations that used monthly resolutions of climate inputs because increased summer decomposition changed the structure of the peat and slowed down the flow of water out of the model.

Second, I demonstrated for the first time, the simulated centennial to millennial impact of gullies and gully blocking on blanket peatland carbon storage (Chapter 6). Overall, I found that the loss of peat caused by a gully over 100 years was not ameliorated 200 years after the gully was blocked. I also show how the effect of gullying and gully blocking varied according to up– or downslope proximity to the gully. When the gully was open, less peat was lost from downslope positions than those upslope of the gully. And after blocking, peat downslope of the gully accumulated at a greater rate than upslope of the gully due to changes in peat structure. I tested two types of gully dam, the first was fixed and the second simulated gully infilling by sediment and new vegetation growth. The infilling gully had the greatest effect on increased peat accumulation. The results suggest that; (1) gullies should be blocked to reverse or arrest loss of peat through oxidation, which is likely to continue when gullies are revegetated only; (2) gully dams should be set as close to the top of the gully as possible, although implementation in the field will depend on the width and depth of the gully; and (3) water–table regimes in restored peatlands are likely to show greater spatial variation than those in intact peatlands because gullying causes rapid secondary decomposition of gully–side peat, and upslope peat may show little response to gully blocking. These findings also imply that water table comparisons between intact and restored sites should be made with caution because of the likely differences in peat structure.

7.2.3 Feedback between the peatland development model and the co–developed network model

Previously I suggested that the peatland development model could enrich the co–developed network model by investigating the impact of driver concepts. I now consider how this might be achieved, using the results of Chapters 5 and 6, both directly to the co–developed network model, and indirectly via stakeholders' mental models.

In the network of highly important concepts used to model land–use objectives, total rainfall was the single concept included by stakeholders to represent climate variables (Chapter 3). In effect this represents an annual average value for rainfall in the same way as that used by previous studies of peatland development (e.g. Morris et al. 2015a). Although rainfall intensity and temperature were part of the complete network, they were classified by stakeholders during the validation process as very important, and important respectively. But three conclusions can be drawn from peatlands modelled using different temporal resolutions of rainfall and temperature (Chapter 5), and peatland climate studies that suggest a different representation and classification of rainfall and temperature could improve the network model: (1) The results of simulations highlighted the importance of including sub-annual variability in both rainfall and temperature in blanket peatland models; (2) peat accumulation is likely to increase in locations where a warmer annual temperature prolongs the growing season (Charman et al. 2015); and (3) peatlands appear to be less resilient to changes in temperature, than changes in rainfall, which can shift the peatland to a persistent wetter or drier state (Morris et al. 2015a).

Clearly, stakeholders did not perceive temperature and rainfall intensity to be highly important to the maintenance or increase of blanket peat carbon stocks, yet climate change was included as a

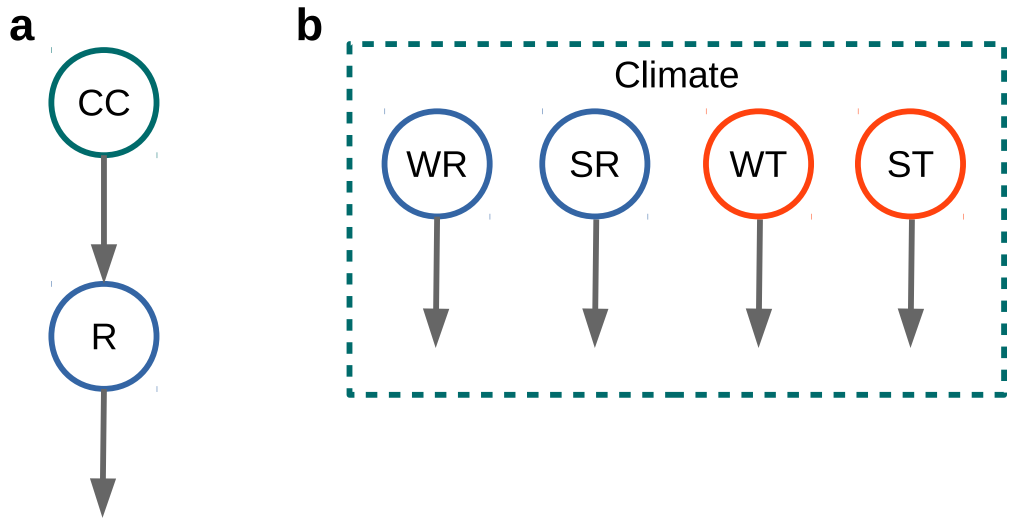


Figure 7.3. Proposed changes to climate variables in the co–developed stakeholder network (highly important subset) as a result of process–based modelling. a Current model. CC = climate change and R = total rainfall. **b** Proposed changes. WR = winter rainfall, SR = summer rainfall, WT = winter temperature, and ST = summer temperature. The climate change concept has been replaced with a notional climate meta concept (dashed rectangle).

separate concept in this subset (Chapter 3), and was classified as a driver concept in the analysis of network structure (Chapter 4). This might be due to the way stakeholders were asked to consider importance: I did not include any reference to timescales in my workshop question which may have affected how concepts were classified. Nonetheless, the conclusions (Section 7.2.3 items 1–3) suggest that a simple modification could be made to how climate variables are represented, and interact in the stakeholder network: four concepts could be added to represent winter and summer rainfall, and winter and summer temperature, which could be grouped under a notional climate meta concept, and the climate change concept removed (Figure 7.3). In this way, simple combinations of seasonal rainfall and temperature could be accommodated. Of course, any suggestions to make changes to the co–developed model would need to be discussed and agreed with stakeholders.

More generally, results from process–based modelling might not feedback directly into the network model as shown Figure 7.3 (C $\rightarrow$ B), but could influence the mental models of stakeholders (Figure 7.3 C $\rightarrow$ A). For example, gully blocking simulations (Chapter 6) could enrich future network models because together they improve our overall understanding of how one of the most used restoration techniques affects peat accumulation both spatially and temporally (Figure 7.4). Although substantive changes to individuals' mental models are not easily achieved (Biggs et al. 2011), Scott et al. (2013) reported persistent alignment of stakeholder mental models 12 months after workshop activities (the limit of the study period), and suggested this was because participants were introduced to ideas that contradicted their own, and acquired new knowledge during the engagement process. This potential for alignment and discovery of new knowledge, as a result of feedback between the two models, could help stakeholders work together where land use is

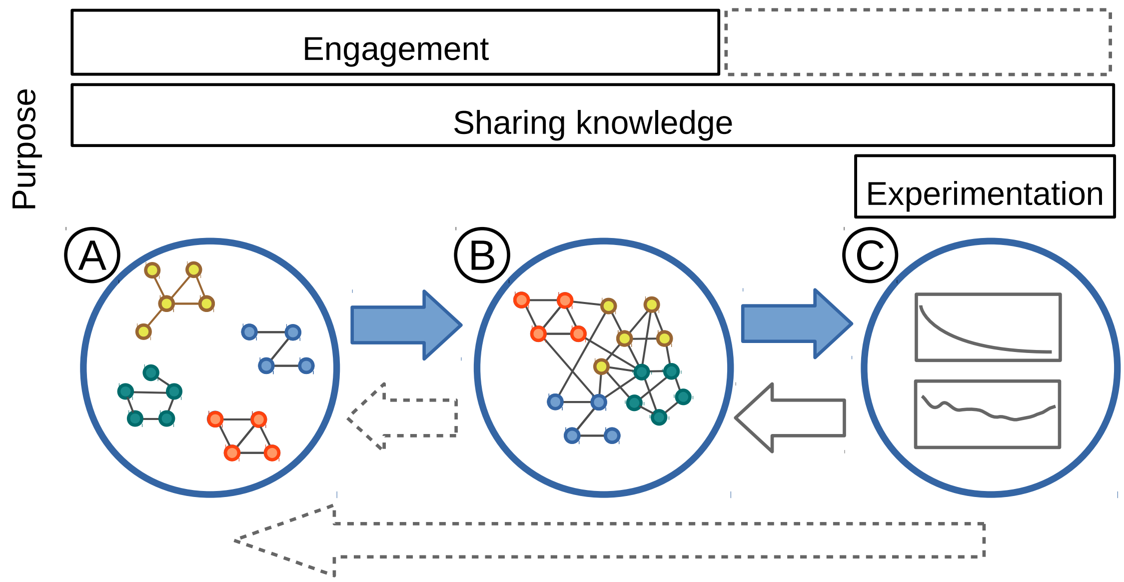


Figure 7.4. Engage–Share–Experiment. A = individual stakeholder mental models, B = aggregated and validated interaction network, C = outputs from simulation model. Blue arrows represent the stakeholder knowledge used to develop the network model and changes to driver concepts. Grey arrows represent the (potential) direct or indirect feedback of outputs to mental models. C $\rightarrow$ A is the indirect enrichment of the network model via stakeholders' mental models, and C $\rightarrow$ B is the direct feedback from the land use workshop and the simulation model to the network model. B $\rightarrow$ A is the feedback to mental models from the validation process. Dashed arrows represent potential routes for feedback. The dashed box represents possible engagement of local land users in fieldwork designed to help parametrise the simulation model.

contested. Similarly the network model can feed forward into the process–based model (as it has in the peatland development studies discussed here), and stakeholders can propose research questions that can be investigated using the process–based model. There also seems to be no reason why some stakeholders in farming and gamekeeping could not help with empirical evidence gathering, by following agreed protocols, that could help parameterise new model functions in **`DigiBog`** (Figure 7.4). Such an approach could also have additional benefits: integrating the involvement of land users in monitoring schemes has been shown to improve the speed at which land–use decisions are made at local scales (Danielsen et al. 2010).

7.3 Future research

There are a number of interesting alternatives that could be used to incorporate mental models in participatory modelling with blanket peatland stakeholders. For example, by using the cognitive map from Chapter 3, other approaches to participatory modelling of complex systems, that can incorporate networks of interactions, such as agent–based models could also be used to explore the future affect of land use on carbon storage. Both the network and peatland development models used are simplifications of real systems and are therefore imperfect: I have previously discussed the limitations of the models, and methods used in Chapters 3–6. I now discuss the potential routes for future research that have emerged during this thesis.

7.3.1 Land–use decisions

This thesis has has brought together knowledge of blanket peatland social and ecological interactions from a group of stakeholders that included land users, statutory bodies, conservationists, and scientists. During workshop activities and discussions, participants suggested how changes could be made to different driver concepts to achieve land–use objectives. Both activities and discussions provided insights into some of the key challenges that would need to be tackled by land users and conservationists if land–use objectives are to complement each other. Based on workshop discussions, it appears that there is an interest and willingness on behalf of many stakeholders (even if some are not yet fully engaged), to participate in a process that can help attain local and wider societal objectives for blanket peatlands.

Voinov and Bousquet (2010) proposed that the process of collaboration in participatory modelling can be more important than the outputs, and Reed (2008) emphasised the need to focus on the process of participation. Therefore stakeholder engagement should be seen as a process rather than series of separate activities or events (Reed 2008). But I am not aware of a systematic ongoing operational process, in the South Pennines, to incorporate the models and processes described in this thesis. Further research is needed to fully understand how the process described in Figure 7.4 can be extended to integrate with an ongoing participatory process to create a bridge between the results and opportunities described in this thesis, and implementation on the ground.

7.3.2 Peatland networks

There is significant scope for additional network–based research on peatlands. Here I discuss two potential approaches for the continued development of network science in peatlands. The first builds on the work started in this thesis where peatlands are represented as social and ecological interactions, and the second focuses on feedback mechanisms.

Interacting networks

Networks provide a simple and intuitive way to represent the interaction between two components of a social–ecological system (e.g. Prager and Pfeifer 2015), but the potential to use networks as a unifying approach across disciplines has not been widely explored (The QUINTESSENCE Consortium 2016). Although interacting networks have been previously studied (e.g. Buldyrev et al. 2010; Gao et al. 2011; Liu et al. 2016), The QUINTESSENCE Consortium (2016) have

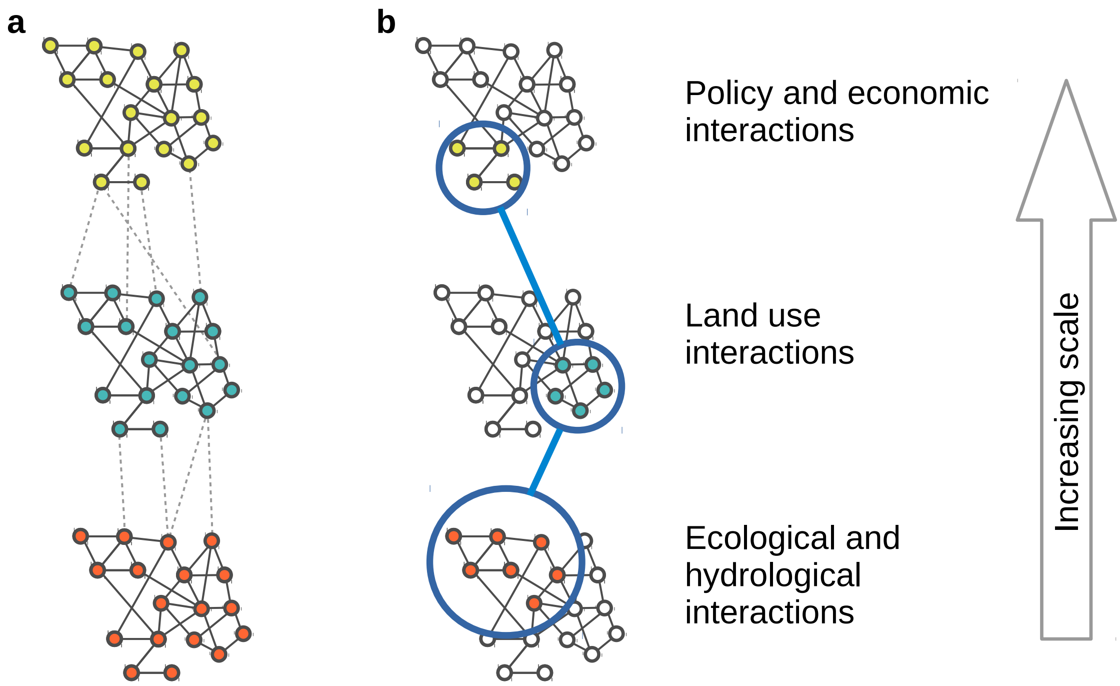


Figure 7.5. Hierarchical layers of network organisation. The current peatland network could be reconfigured and developed into three layers based on the approach of The QUINTESSENCE Consortium (2016). Currently concepts bridge across the three layers. **a** Interdependent networks coupled by dependency interactions For example, a reduction in erosion (ecohydrological layer) is dependent on gully blocking (land use layer) which in turn is dependent on funding for restoration (policy and economic layer). Dashed lines represent dependent interactions. **b** Interacting network subsets. Thick blue circles and links indicate hypothetical network subsets of interest interacting across scales. (Diagram adapted from The QUINTESSENCE Consortium 2016; and Liu et al. 2016).

recently proposed an approach to assess ecosystem service provision that layers ecological, social, and economic networks in a hierarchically organised structure to account for different scales of interactions. The challenges related to the future land use, carbon storage, and livelihoods in peatlands, discussed in Section 7.1, bridge these three layers and could be further investigated using this approach (Figure 7.5).

Peatland stakeholders included concepts from all three layers in their networks (Chapter 3), and therefore future research could focus on classifying concepts into ecohydrological, land use, and policy layers to investigate how interdependent interactions between these networks affect peatland carbon stocks (e.g. Liu et al. 2016) (Figure 7.5a). For example, changes in funding regimes in the policy layer could have cascading effects on land use that may affect peatland resilience. An understanding of how network layers interact could be used inform and communicate land use policy in a consistent manner across stakeholder groups (*sensu* Pocock et al. 2016).

The method I used to capture mental models enabled participants to specify concepts that interacted between different scales. When aggregated, the network included some sub-concepts of higher level concepts such as land use (e.g. types of land use), and vegetation (e.g. types of vegetation) for example. Although this was discussed during validation, not all of these higher level concepts were selected by participants to be removed. However, the approach shown in Figure 7.5 could address this issue if concepts were classified into layered networks. For example; managed burning and gully blocking are sub–concepts that are connected to the land management concept and could be included in the land use layer; whilst economic concepts related to the funding of peatland restoration and land use (e.g. agri–environment schemes and funding for restoration) could become part of the policy/economic layer, removing any ambiguities about the scale at which concepts interact.

The strength of interactions between concepts was specified by stakeholders according to a linguistic scale. This approach is helpful because it challenges participants to examine their beliefs and perceptions about pairwise interactions, but it is difficult to calibrate the different strengths between individual participants. I attempted to address this difficulty by asking groups of stakeholders to validate interaction strength and direction. However, if networks are to be developed across three disciplines, and between groups, there may not be an opportunity to validate interaction strength in the same way and, therefore, alternative approaches may be needed to investigate the topological and dynamical properties of interacting networks (Figure 7.5). The benefits of networks for engagement, visualisation, and communication (Pocock et al. 2016) could still be obtained with this approach because stakeholders can be involved diagramming, sharing mental models, and fieldwork to identify pairwise interactions, and to determine how network layers may be coupled.

Feedback networks

Feedback mechanisms are important characteristics of complex systems that mediate responses to allogenic forcing and drive autogenic behaviour. As such, feedbacks describe key functional relationships that should be considered when developing models of complex systems. Waddington et al. (2015) produced an evidence based set of interactions to represent hydrological feedbacks in northern peatlands, and proposed that other interaction networks such as ecological feedbacks could be developed, and the networks linked together. However, it is impractical to include all possible feedback mechanisms in models; but often it is difficult to know which mechanisms might be necessary to adequately represent the system in question (Grimm et al. 2005). There seems to be no reason why the knowledge of other peatland stakeholders should not be used to help develop these feedback networks in processes that could be similar to the one used in Chapter 3. Because feedbacks interact within and across scales, network analysis and visualisation of the structure of these interactions could help to identify the feedback mechanisms that should be tested and, possibly, incorporated in peatland development models.

7.3.3 Peatland development

The new version of **DigiBog** used in Chapters 5 and 6, has shown how a model of blanket peatlands, as a complex system, can be used to explore the impact of climate variables and land use on carbon storage. In Chapter 5, I discussed a number of future developments to **DigiBog** that includes reducing model run times, investigating the development of surface patterns that were apparent in one set of simulations, and new or modified functions for plant functional types, bulk density and drainable porosity. In this section I briefly discuss the potential of the 2D/3D model to inform land use decisions.

Firstly, those responsible for peatland restoration and land use are likely to want to consider how land management might affect peatland carbon stores when coupled with future climate scenarios. To determine the appropriate resolution of climate variables, I tested the impact on peat accumulation of three different temporal resolutions of the same temperature and rainfall data, and recommended that weekly timeseries should be used for blanket peatland models. The next step would be to use the 50 m 2D configuration of **DigiBog** from Chapter 5 (which represents an intact peatland), and test the affect of different climate scenarios (e.g. Li et al. 2015) for the South Pennines on peat accumulation. Studies have suggested that the northwards shift of a suitable climatic envelope for blanket peatlands may mean that peatlands in the South Pennines become more vulnerable to erosion (Gallego-Sala and Prentice 2012; Li et al. 2015). Although a future climate may be unsuitable for peat accumulation or initiation, the adaptive response of existing blanket peatlands to this shift has not yet been explored. And because degraded peatlands, which represent a significant extent of South Pennine peats, are likely to be less resilient to climate change than intact peatlands, the impact of restoration and climate change could be modelled. This 2D version of **DigiBog**, could also be adapted to understand how degraded peatlands throughout UK, and other locations around the globe are likely to respond to the coupled effects of climate change and attempts to restore ecological and hydrological processes.

Secondly, the simulations reported in Chapters 5 and 6 were carried out over a uniform 50 m slope or plateau. However, the underlying topography of blanket peatlands will be highly variable, and include deep basins and steep slopes as well as sections of plateau (e.g. Tipping 2008). It is possible to configure the model for irregular bases and horizontal extents, but the time taken to run these simulations precluded their use in this thesis. Nevertheless the underlying topography reported by studies such as Tipping (2008), or from field surveys of an area of interest, could be used to explore how the model represents peatland development over varying terrain using 50 m (2 m $\times$ 2 m columns) or 100 m (4 m $\times$ 4 m columns) transects. Irregular extents and longer transects could be simulated once the speed of model sub-routines have been improved (Chapter 5):

the inclusion of different topographies in simulations could reveal areas of the peatland that may have the greatest positive impact on carbon storage if restored.

7.4 Summary

The aim of this thesis was to develop new knowledge about blanket peatlands as complex systems, and to enhance our understanding of the impact of social and ecological interactions on carbon storage. This aim was achieved in the following ways:

1. Establishing an aggregate fuzzy cognitive map of the key blanket peatland social and ecological interactions, as perceived by a range of stakeholders from the South Pennines peatland community, and by using the structure of these interactions in a participatory process to evaluate the impact of land–use objectives on carbon storage.
2. Identifying the broad areas of agreement and disagreement in how the group of peatland stakeholders perceived that land–use objectives related to livelihoods, carbon, and water, could be achieved.
3. Developing a new version of the DigiBog peatland development model to simulate blanket peat accumulation in 2D on slopes and plateaus.
4. Using the new model to investigate the centennial to millennial impacts on blanket peat carbon storage of two concepts selected from the stakeholder network: (a) climate and (b) gully blocking.

This study linked participatory and process–based modelling of blanket peatlands as complex systems. The participatory approach builds on previous work by coupling mental models with the implications of network structure to provide a simple and accessible method for stakeholders to collaborate and to propose how to achieve land–use objectives for blanket peatlands, which is likely to be applicable to other social–ecological systems where land use is contested. The peatland development model provided new insights into the complex spatial and temporal responses of blanket peatlands to external forcing over the long term including key concepts from the stakeholder model of networked interactions, and has the potential to be configured for different underlying topographies and land uses. Both the network model and new peatland development model could be used separately to inform land–use decisions, but I suggest that the engagement of stakeholders will be enhanced, and simulation results more widely accepted if the models are coupled as in this thesis. The results from the models and insights from workshop discussions should also be integrated into a wider process of bottom–up stakeholder engagement, and land–use decision making.

References

Abel, N., H. Ross and P. Walker (1998). Mental models in rangeland research, communication and management. *The Rangeland Journal* 20 (1), pp. 77–91. DOI: 10.1071/RJ9980077.

Albert, R., H. Jeong and A. Barabási (2000). Error and attack tolerance of complex networks. *Nature* 406, pp. 378–382.

Albert, R. and A.-L. Barabási (2002). Statistical mechanics of complex networks. *Reviews of Modern Physics* 74 (1), pp. 47–97.

Albert, R., H. Jeong and A. L. Barabasi (1999). Internet: Diameter of the World-Wide Web. *Nature* 401, pp. 130–131. DOI: 10.1038/43601.

Albertson, K., J. Aylen, G. Cavan and J. McMorrow (2009). Forecasting the outbreak of moorland wildfires in the English Peak District. *Journal of Environmental Management* 90 (8), pp. 2642–2651. DOI: 10.1016/j.jenvman.2009.02.011.

Allen, K. A., M. P. K. Harris and R. H. Marrs (2013). Matrix modelling of prescribed burning in Calluna vulgaris-dominated moorland: short burning rotations minimize carbon loss at increased wildfire frequencies. *Journal of Applied Ecology* 50 (3), pp. 614–624. DOI: 10.1111/1365-2664.12075.

Allott, T., M. Evans, J. Lindsay, C. Agnew, J. Freer, A. Jones and M. Parnell (2009). *Water tables in Peak District blanket peatlands*. Tech. rep. No. 17. Moors for the Future.

An, L. (2012). Modeling human decisions in coupled human and natural systems: Review of agent-based models. *Ecological Modelling* 229, pp. 25–36. DOI: 10.1016/j.ecolmodel.2011.07.010.

Ancient peatlands grow again (2015). URL: http://www.moorsforthefuture.org.uk/news/ancient-peatlands-grow-again (visited on 10/06/2015).

Anderson, P. and D. Yalden (1981). Increased sheep numbers and the loss of heather moorland in the Peak District, England. *Biological Conservation* 20 (3), pp. 195–213. DOI: 10.1016/0006-3207(81)90029-X.

Anderson, P. and E. Radford (1993). Changes in vegetation following reduction in grazing pressure on the National Trust's Kinder estate, Peak District, Derbyshire, England. *Biological Conservation* 69, pp. 55–63.

Anderson, P. (2014). Bridging the gap between applied ecological science and practical implementation in peatland restoration. *Journal of Applied Ecology* 51 (5), pp. 1148–1152. DOI: 10.1111/1365-2664.12258.

Artz, R. R. E., S. J. Chapman, M. Saunders, C. D. Evans and R. B. Matthews (2013). Comment on "Soil CO_2, CH_4 and N_2O fluxes from an afforested lowland raised peat bog in Scotland: implications for drainage and restoratio" by Yamulki et al. (2013). *Biogeosciences* 10, pp. 7623–7630. DOI: 10.5194/bg-10-7623-2013.

Axelrod, R. (1976). *The Structure of decision: cognitive maps of political elites*. Princeton University Press.

Baird, A. J., T. Mason and D. P. Horn (1998). Validation of a Boussinesq model of beach ground water behaviour. *Marine Geology* 148, pp. 55–69. DOI: 10.1016/S0025-3227(98)00026-7.

Baird, A. J., P. J. Morris and L. R. Belyea (2011). The DigiBog peatland development model 1: rationale, conceptual model, and hydrological basis. *Ecohydrology* 5 (3), pp. 242–255. DOI: 10.1002/eco.230.

Baird, A. J. (2014). *Critical review of widths of drainage influence associated with artificial drains in blanket bog, with particular reference to the 2011 update of the Scottish Government Carbon Calculator Tool*. Tech. rep. Report for the Countryside Council of Wales.

Ballard, C., N. McIntyre, H. Wheater, J. Holden and Z. Wallage (2011). Hydrological modelling of drained blanket peatland. *Journal of Hydrology* 407, pp. 81–93. DOI: 10.1016/j.jhydrol.2011.07.005.

Banini, G. and R. Bearman (1998). Application of fuzzy cognitive maps to factors affecting slurry rheology. *International Journal of Mineral Processing* 52 (4), pp. 233–244. DOI: 10.1016/S0301-7516(97)00071-9.

Barabási, A.-L. and R. Albert (1999). Emergence of scaling in random networks. *Science* 286 (October), pp. 509–512. DOI: 10.1126/science.286.5439.509.

Barabási, A.-L. and Z. N. Oltvai (2004). Network biology: understanding the cell's functional organization. *Nature reviews. Genetics* 5, pp. 101–113. DOI: 10.1038/nrg1272.

Barabási, A.-L. (2009). Scale-Free Networks: A Decade and Beyond. *Science* 325, pp. 412–413.

Barabási, A.-L., N. Gulbahce and J. Loscalzo (2011). Network medicine: a network-based approach to human disease. *Nature reviews. Genetics* 12, pp. 56–68. DOI: 10.1038/nrg2918.

Barabási, A.-L. (2012). The network takeover. *Nature Physics* 8, pp. 14–16. DOI: 10.1038/nphys2188.

Barabási, A.-L. (2015). Network Robustness. In: *Network Science*. Creative Commons: CC BY-NC-SA 2.0. PDF V26, 05.09.2014. Chap. 8, pp. 1–16.

Barnosky, A. D., E. A. Hadly, J. Bascompte, E. L. Berlow, J. H. Brown, M. Fortelius, W. M. Getz, J. Harte, A. Hastings, P. A. Marquet, N. D. Martinez, A. Mooers, P. Roopnarine, G. Vermeij, J. W. Williams, R. Gillespie, J. Kitzes, C. Marshall, N. Matzke, D. P. Mindell, E. Revilla, A. B. Smith, D. Barnosky Anthony, A. Hadly Elizabeth, L. Berlow Eric and H. Brown James (2012). Approaching a state shift in Earth's biosphere. *Nature* 486, pp. 52–58. DOI: 10.1038/nature11018.

Bashari, H., C. Smith and O. Bosch (2008). Developing decision support tools for rangeland management by combining state and transition models and Bayesian belief networks. *Agricultural Systems* 99 (1), pp. 23–34. DOI: 10.1016/j.agsy.2008.09.003.

Beetz, S., H. Liebersbach, S. Glatzel, G. Jurasinski, U. Buczko and H. Höper (2013). Effects of land use intensity on the full greenhouse gas balance in an Atlantic peat bog. *Biogeosciences* 10, pp. 1067–1082. DOI: 10.5194/bg-10-1067-2013.

Bellamy, P. E., L. Stephen, I. S. Maclean and M. C. Grant (2012). Response of blanket bog vegetation to drain-blocking. *Applied Vegetation Science* 15 (1), pp. 129–135. DOI: 10.1111/j.1654-109X.2011.01151.x.

Belyea, L. R. (1996). Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland. *Oikos* 77 (3), pp. 529–539.

Belyea, L. R. and R. S. Clymo (2001). Feedback control of the rate of peat formation. *Proceedings. Biological sciences / The Royal Society B* 268, pp. 1315–1321. DOI: 10.1098/rspb.2001.1665.

Belyea, L. R., N. Malmer, D. Street and E. Building (2004). Carbon sequestration in peatland: patterns and mechanisms of response to climate change. *Global Change Biology* 10, pp. 1043–1052. DOI: 10.1111/j.1365-2486.2004.00783.x.

Belyea, L. R. and A. J. Baird (2006). Beyond the limits to peat bog growth: cross-scale feedback in peatland development. *Ecological Monographs* 76 (3), pp. 299–322.

Belyea, L. R. (2009). Nonlinear dynamics of peatlands and potential feedbacks on the climate system. In: A. J. Baird, L. R. Belyea, X. Comas, A. S. Reeve and L. D. Slater (Eds.), *Carbon Cycling in Northern Peatlands*. Geophysica. Washington, D.C.: AGU, pp. 5–18. DOI: 10.1029/2008GM000875.

Bestelmeyer, B. T., J. E. Herrick, J. R. Brown, D. a. Trujillo and K. M. Havstad (2004). Land management in the American southwest: a state-and-transition approach to ecosystem complexity. *Environmental management* 34 (1), pp. 38–51. DOI: 10.1007/s00267-004-0047-4.

Beyer, C. and H. Höper (2015). Greenhouse gas exchange of rewetted bog peat extraction sites and a *Sphagnum* cultivation site in northwest Germany. *Biogeosciences* 12, pp. 2101–2117. DOI: 10.5194/bg-12-2101-2015.

Biggs, D., N. Abel, A. T. Knight, A. Leitch, A. Langston and N. C. Ban (2011). The implementation crisis in conservation planning: Could "mental models" help? *Conservation Letters* 4 (3), pp. 169–183.

Biggs, R., S. R. Carpenter and W. A. Brock (2009). Turning back from the brink: Detecting an impending regime shift in time to avert it. *Proceedings of the National Academy of Sciences* 106 (3), pp. 826–831. DOI: 10.1073/pnas.0811729106.

Biggs, R., M. Schlüter, D. Biggs, E. L. Bohensky, S. BurnSilver, G. Cundill, V. Dakos, T. M. Daw, L. S. Evans, K. Kotschy, A. M. Leitch, C. Meek, A. Quinlan, C. Raudsepp-Hearne, M. D. Robards, M. L. Schoon, L. Schultz and P. C. West (2012). Toward principles for enhancing the resilience of ecosystem services. *Annual Review of Environment and Resources* 37, pp. 421–448. DOI: 10.1146/annurev-environ-051211-123836.

Billett, M., D. Charman, J. Clark, C. Evans, M. Evans, N. Ostle, F. Worrall, A. Burden, K. Dinsmore, T. Jones, N. McNamara, L. Parry, J. Rowson and R. Rose (2010). Carbon balance of UK peatlands: current state of knowledge and future research challenges. *Climate Research* 45, pp. 13–29. DOI: 10.3354/cr00903.

Blundell, A. and J. Holden (2015). Using palaeoecology to support blanket peatland management. *Ecological Indicators* 49, pp. 110–120. DOI: 10.1016/j.ecolind.2014.10.006.

Boccaletti, S., V. Latora, Y. Moreno, M. Chavez and D. U. Hwang (2006). Complex networks: Structure and dynamics. *Physics Reports* 424, pp. 175–308. DOI: 10.1016/j.physrep.2005.10.009.

Bodin, O. and B. Crona (2011). Barriers and opportunities in transforming to sustainable governance: the role of key individuals. In: O. Bodin and C. Prell (Eds.), *Social networks and natural resource management. Uncoververing the social fabric of environmental governance*. Cambridge University Press. Chap. 4, pp. 75–94.

Bommel, P., F. Dieguez, D. Bartaburu, Duarte, E. Montes, M. P. Machin, J. Corral, C. J. Pereia de Lucena and M. Grosskopf (2014). A further step towards participatory modelling. Fostering stakeholder involvement in designing models by using executable UML. *Journal of Artificial Societies and Social Simulation* 17 (1), p. 6.

Bonn, A., M. Rebane and C. Reid (2009). Ecosystem services. A new rationale for conservation of upland environments. In: A. Bonn, T. Allott, K. Hubacek and J. Stewart (Eds.), *Drivers of environmental change in the uplands*. Abingdon, UK: Routledge. Chap. 25, pp. 448–475.

Bonn, A., M. S. Reed, C. D. Evans, H. Joosten, C. Bain, J. Farmer, I. Emmer, J. Couwenberg, A. Moxey, R. Artz, F. Tanneberger, M. von Unger, M.-A. Smyth and D. Birnie (2014). Investing in nature: Developing ecosystem service markets for peatland restoration. *Ecosystem Services* 9, pp. 54–65. DOI: 10.1016/j.ecoser.2014.06.011.

Borgmark, A. and K. Schoning (2006). A comparative study of peat proxies from two eastern central Swedish bogs and their relation to meteorological data. *Journal of Quaternary Science* 21 (2), pp. 109–114. DOI: 10.1002/jqs.959.

Box, G. E. P. (1976). Science and Statistics. *Journal of the American Statistical Association* 71 (356), pp. 791–799.

Bragg, O. M. and J. H. Tallis (2001). The sensitivity of peat-covered upland landscapes. *Catena* 42, pp. 345–360. DOI: 10.1016/S0341-8162(00)00146-6.

Brin, S. and L. Page (1998). The anatomy of a large-scale hypertextual Web search engine. *Computer Networks and ISDN Systems* 30, pp. 107–117. DOI: 10.1016/S0169-7552(98)00110-X.

Brook, B. W., E. C. Ellis, M. P. Perring, A. W. Mackay and L. Blomqvist (2013). Does the terrestrial biosphere have planetary tipping points? *Trends in Ecology & Evolution* 28 (7), pp. 396–401. DOI: 10.1016/j.tree.2013.01.016.

Brown, L. E., K. Johnston, S. M. Palmer, K. L. Aspray and J. Holden (2013). River Ecosystem Response to Prescribed Vegetation Burning on Blanket peatland. *PLoS ONE* 8 (11). DOI: 10.1371/journal.pone.0081023.

Buldyrev, S. V., R. Parshani, G. Paul, H. E. Stanley and S. Havlin (2010). Catastrophic cascade of failures in interdependent networks. *Nature* 464, pp. 1025–1028. DOI: 10.1038/nature08932.

Bullock, C. H. and M. Collier (2011). When the public good conflicts with an apparent preference for unsustainable behaviour. *Ecological Economics* 70 (5), pp. 971–977. DOI: 10.1016/j.ecolecon.2010.12.013.

Bullock, C. H., M. J. Collier and F. Convery (2012). Peatlands, their economic value and priorities for their future management – The example of Ireland. *Land Use Policy* 29 (4), pp. 921–928. DOI: 10.1016/j.landusepol.2012.01.010.

Busch, J., K. Ferretti-Gallon, J. Engelmann, M. Wright, K. G. Austin, F. Stolle, S. Turubanova, P. V. Potapov, B. Margono, M. C. Hansen and A. Baccini (2015). Reductions in emissions from deforestation from Indonesia's moratorium on new oil palm, timber, and logging concessions. *Proceedings of the National Academy of Sciences* 112 (5), pp. 1328–1333. DOI: 10.1073/pnas.1412514112.

Butler, J. R. A., J. C. Young, I. A. G. McMyn, B. Leyshon, I. M. Graham, I. Walker, J. M. Baxter, J. Dodd and C. Warburton (2015). Evaluating adaptive co-management as conservation conflict resolution: Learning from seals and salmon. *Journal of Environmental Management* 160, pp. 212–225. DOI: 10.1016/j.jenvman.2015.06.019.

Capistrano, D. and C. Samper (2005). *Reflections and Lessons Learned*. Tech. rep. http://www.millenniumassessment.org/en/Multiscale.html. Chap. 12, pp. 279–289.

Carley, K. M. and M. Palmquist (1992). Extracting, representing, and analyzing mental models. *Social Forces* 70 (3), pp. 601–636. DOI: 10.2307/2579746.

Carroll, M. J., P. Dennis, J. W. Pearce-Higgins and C. D. Thomas (2011). Maintaining northern peatland ecosystems in a changing climate: Effects of soil moisture, drainage and drain blocking on craneflies. *Global Change Biology* 17, pp. 2991–3001. DOI: 10.1111/j.1365-2486.2011.02416.x.

Carvalho, J. P., L. Wise, A. Murta and M. Mesquita (2008). ‘Issues on Dynamic Cognitive Map modelling of purse-seine fishing skippers behavior’. In: *Fuzzy Systems, 2008. FUZZ-IEEE 2008. (IEEE World Congress on Computational Intelligence). IEEE International Conference on*, pp. 1503–1510. DOI: 10.1109/FUZZY.2008.4630571.

Carvalho, J. P. (2013). On the semantics and the use of fuzzy cognitive maps and dynamic cognitive maps in social sciences. *Fuzzy Sets and Systems* 214, pp. 6–19. DOI: 10.1016/j.fss.2011.12.009.

Ceballos, G., P. R. Ehrlich, A. D. Barnosky, A. García, R. M. Pringle and T. M. Palmer (2015). Accelerated modern human–induced species losses: Entering the sixth mass extinction. *Science Advances* 1 (5). DOI: 10.1126/sciadv.1400253. eprint: http://advances.sciencemag.org/content/1/5/e1400253.full.pdf.

Chan, T., H. Ross, S. Hoverman and B. Powell (2010). Participatory development of a Bayesian network model for catchment-based water resource management. *Water Resources Research* 46, W07544. DOI: 10.1029/2009WR008848.

Chapin, F. S., F. M. Alan, R. A. Mitchell and K. J. M. Dickinson (2012). Design principles for social-ecological transformation toward sustainability : lessons from New Zealand sense of place. *Ecosphere* 3 (5), p. 22.

Chapman, D. S., A. Bonn, W. E. Kunin and S. J. Cornell (2009a). Random Forest characterization of upland vegetation and management burning from aerial imagery. *Journal of Biogeography* 37, pp. 37–46. DOI: 10.1111/j.1365-2699.2009.02186.x.

Chapman, D. S., M. Termansen, C. H. Quinn, N. Jin, A. Bonn, S. J. Cornell, E. D. G. Fraser, K. Hubacek, W. E. Kunin and M. S. Reed (2009b). Modelling the coupled dynamics of moorland management and upland vegetation. *Journal of Applied Ecology* 46 (2), pp. 278–288. DOI: 10.1111/j.1365-2664.2009.01618.x.

Chapman, S., A. Buttler, A.-J. Francez, F. Laggoun-Défarge, H. Vasander, M. Schloter, J. Combe, P. Grosvernier, H. Harms, D. Epron, D. Gilbert and E. Mitchell (2003). Exploitation of northern peatlands and biodiversity maintenance: a conflict between economy and ecology. *Frontiers in Ecology and the Environment* 1 (10), pp. 525–532. DOI: 10.1890/1540-9295(2003)001[0525:EONPAB]2.0.CO;2.

Charman, D. J. (2002). *Peatlands and environmental change*. UK: John Wiley & Sons Ltd.

Charman, D. J. et al. (2013). Climate-related changes in peatland carbon accumulation during the last millennium. *Biogeosciences* 10, pp. 929–944. DOI: 10.5194/bg-10-929-2013.

Charman, D. J., M. J. Amesbury, W. Hinchliffe, P. D. Hughes, G. Mallon, W. H. Blake, T. J. Daley, A. V. Gallego-Sala and D. Mauquoy (2015). Drivers of Holocene peatland carbon accumulation across a climate gradient in northeastern North America. *Quaternary Science Reviews* 121, pp. 110–119. DOI: 10.1016/j.quascirev.2015.05.012.

Chokkalinga, U., C. Sabogal, E. Almeida, A. P. Carandang, T. Gumartini, W. de Jong, S. Brienza Jr, A. M. Lopez, Murniati, A. A. Nawir, L. R. Wibowo, T. Toma, E. Wollenberg and Z. Zaizhi (2005). Local participation, livelihood needs and institutional arrangements: three keys to sustainable rehabilitation of degraded tropical forestlands. In: M. S, D. Vallauri and N. Dudley (Eds.), *Forest restoration in landscapes: beyond planting trees*. Chap. 58, pp. 405–414.

Christen, B., C. Kjeldsen, T. Dalgaard and J. Martin-Ortega (2015). Can fuzzy cognitive mapping help in agricultural policy design and communication? *Land Use Policy* 45, pp. 64–75. DOI: 10.1016/j.landusepol.2015.01.001.

Clark, J. M., a. V. Gallego-Sala, T. E. H. Allott, S. J. Chapman, T. Farewell, C. Freeman, J. I. House, H. G. Orr, I. C. Prentice and P. Smith (2010). Assessing the vulnerability of blanket peat to climate change using an ensemble of statistical bioclimatic envelope models. *Climate Research* 45, pp. 131–150. DOI: 10.3354/cr00929.

Clauset, A., M. Newman and C. Moore (2004). Finding community structure in very large networks. *Physical Review E* 70 (066111). DOI: 10.1103/PhysRevE.70.066111.

Clauset, A., C. R. Shalizi and M. E. J. Newman (2009). Power-law distributions in empirical data. *SIAM Review* 51 (4), pp. 661–703. DOI: 10.1137/070710111.

Clay, G. D., F. Worrall, E. Clark and E. D. Fraser (2009). Hydrological responses to managed burning and grazing in an upland blanket bog. *Journal of Hydrology* 376, pp. 486–495. DOI: 10.1016/j.jhydrol.2009.07.055.

Clay, G. D., S. Dixon, M. G. Evans, J. G. Rowson and F. Worrall (2012). Carbon dioxide fluxes and DOC concentrations of eroding blanket peat gullies. *Earth Surface Processes and Landforms* 37 (5), pp. 562–571. DOI: 10.1002/esp.3193.

Clay, G. D., F. Worrall and N. J. Aebischer (2015). Carbon stocks and carbon fluxes from a 10-year prescribed burning chronosequence on a UK blanket peat. *Soil Use and Management* 31, pp. 39–51. DOI: 10.1111/sum.12164.

Clymo, R. S., T. R. Society, P. Transactions, R. Society and B. Sciences (1984). The Limits to Peat Bog Growth. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* 303 (1117), pp. 605–654.

Clymo, R. S., D. M. E. Pearce and R. Conrad (1995). Methane and Carbon Dioxide Production in, Transport through, and Efflux from a Peatland [and Discussion]. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* 351, pp. 249–259. DOI: 10.1098/rsta.1995.0032.

Cohen, R., K. Erez, D. Ben-Avraham and S. Havlin (2000). Resilience of the Internet to random breakdowns. *Physical Review Letters* 85 (21), pp. 4626–4628. DOI: 10.1103/PhysRevLett.85.4626.

Cohen, R., K. Erez, D. Ben-Avraham and S. Havlin (2001). Breakdown of the internet under intentional attack. *Physical Review Letters* 86 (16), pp. 3682–3685. DOI: 10.1103/PhysRevLett.86.3682.

Committee on Climate Change (2015). *Progress in preparing for climate change 2015. Report to Parliament*. Tech. rep., p. 240.

Condliffe, I. (2009). Policy change in the uplands. In: A. Bonn, T. Allott, K. Hubacek and J. Stewart (Eds.), *Drivers of Environmental Change in the Uplands*. Routledge. Chap. 4, pp. 60–89.

Conway, V. M. (1954). Stratigraphy and Pollen Analysis of Southern Pennine Blanket Peats. *Journal of Ecology* 42 (1), pp. 117–147.

Corral-Quintana, S., D. Legna-de la Nuez, C. Legna Verna, J. H. Hernández and D. Romero-Manrique de Lara (2016). How to improve strategic decision-making in complex systems when only qualitative information is available. *Land Use Policy* 50, pp. 83–101. DOI: 10.1016/j.landusepol.2015.09.004.

Couix, N. and H. Gonzalo-Turpin (2015). Towards a land management approach to ecological restoration to encourage stakeholder participation. *Land Use Policy* 46, pp. 155–162. DOI: 10.1016/j.landusepol.2015.01.025.

Coulthard, T. J. and C. J. Skinner (2016). The sensitivity of landscape evolution models to spatial and temporal rainfall resolution. *Earth Surface Dynamics Discussions*. DOI: 10.5194/esurf-2016-2.

Csardi, G. and T. Nepusz (2006). The igraph software package for complex network research. *InterJournal* Complex Systems, p. 1695.

Cuddington, K., M.-J. Fortin, L. R. Gerber, A. Hastings, A. Liebhold, M. O'Connor and C. Ray (2013). Process-based models are required to manage ecological systems in a changing world. *Ecosphere* 4 (2). DOI: 10.1890/ES12-00178.1.

Dakos, V. and J. Bascompte (2014). Critical slowing down as early warning for the onset of collapse in mutualistic communities. *PNAS* 111 (49), pp. 17546–17551. DOI: 10.1073/pnas.1406326111.

Daniels, S., C. Agnew, T. Allott and M. Evans (2008). Water table variability and runoff generation in an eroded peatland, South Pennines, UK. *Journal of Hydrology* 361, pp. 214–226. DOI: 10.1016/j.jhydrol.2008.07.042.

Danielsen, F., N. D. Burgess, P. M. Jensen and K. Pirhofer-Walzl (2010). Environmental monitoring: the scale and speed of implementation varies according to the degree of peoples involvement. *Journal of Applied Ecology* 47 (6), pp. 1166–1168. DOI: 10.1111/j.1365-2664.2010.01874.x.

Davies, K. K., K. T. Fisher, M. E. Dickson, S. F. Thrush and R. L. Heron (2015). Improving ecosystem service frameworks to address wicked problems. *Ecology and Society* 20 (2). 37. http://dx.doi.org/10.5751/ES-07581-200237.

Defra (2007). *The Heather and Grass Burning Code*. Tech. rep. London: Department for Environment, Food and Rural Affairs.

Defra (2011). *Uplands Policy Review*. Tech. rep. London: Department for Environment Food and Rural Affiars.

Dengel, S., D. Zona, T. Sachs, M. Aurela, M. Jammet, F. J. W. Parmentier, W. Oechel and T. Vesala (2013). Testing the applicability of neural networks as a gap-filling method using CH_4 flux data from high latitude wetlands. *Biogeosciences* 10 (12), pp. 8185–8200. DOI: 10.5194/bg-10-8185-2013.

Dexter, N., D. S. L. Ramsey, C. MacGregor and D. Lindenmayer (2012). Predicting Ecosystem Wide Impacts of Wallaby Management Using a Fuzzy Cognitive Map. *Ecosystems* 15 (8), pp. 1363–1379. DOI: 10.1007/s10021-012-9590-7.

Dioumaeva, I., S. Trumbore, E. A. G. Schuur, M. L. Goulden, M. Litvak and A. I. Hirsch (2003). Decomposition of peat from upland boreal forest: Temperature dependence and sources of respired carbon. *Journal of Geophysical Research: Atmospheres* 107 (D3). 8222. DOI: 10.1029/2001JD000848.

Dixon, S. D., S. M. Qassim, J. G. Rowson, F. Worrall, M. G. Evans, I. M. Boothroyd and A. Bonn (2014). Restoration effects on water table depths and CO2 fluxes from climatically marginal blanket bog. *Biogeochemistry* 118, pp. 159–176. DOI: DOI10.1007/s10533-013-9915-4.

Dixon, S. D., F. Worrall, J. G. Rowson and M. G. Evans (2015). Calluna vulgaris canopy height and blanket peat CO2 flux: Implications for management. *Ecological Engineering* 75, pp. 497–505. DOI: 10.1016/j.ecoleng.2014.11.047.

Dougill, A. J., E. D. G. Fraser, J. Holden, K. Hubacek, C. Prell, M. S. Reed, S. Stagl and L. C. Stringer (2006). Learning from Doing Participatory Rural Research: Lessons from the Peak District National Park. *Journal of Agricultural Economics* 57 (2), pp. 259–275. DOI: 10.1111/j.1477-9552.2006.00051.x.

Douglas, D. J., G. M. Buchanan, P. Thompson, A. Amar, D. a. Fielding, S. M. Redpath and J. D. Wilson (2015). Vegetation burning for game management in the UK uplands is increasing and overlaps spatially with soil carbon and protected areas. *Biological Conservation* 191, pp. 243–250. DOI: 10.1016/j.biocon.2015.06.014.

Doyle, J. and D. Ford (1998). Mental models concepts for research. *Systems Dynamics Review* 14 (1), pp. 3–29.

Drew, S., S. Waldron, D. Gilvear, I. Grieve, A. Armstrong, O. Bragg, F. Brewis, M. Cooper, T. Dargie, C. Duncan, L. Harris, L. Wilson, C. McIver, R. Padfield and N. Shah (2013). The price of knowledge in the knowledge economy: Should development of peatland in the UK support a research levy? *Land Use Policy* 32, pp. 50–60. DOI: 10.1016/j.landusepol.2012.10.007.

Dunne, J. a., R. J. Williams and N. D. Martinez (2002). Food-web structure and network theory: The role of connectance and size. *Proceedings of the National Academy of Sciences of the United States of America* 99 (20), pp. 12917–12922. DOI: 10.1073/pnas.192407699.

Durance, P. and M. Godet (2010). Scenario building: Uses and abuses. *Technological Forecasting and Social Change* 77 (9), pp. 1488–1492. DOI: 10.1016/j.techfore.2010.06.007.

Egoh, B. N., M. L. Paracchini, G. Zulian, J. P. Schägner and G. Bidoglio (2014). Exploring restoration options for habitats, species and ecosystem services in the European Union. *Journal of Applied Ecology* 51 (4), pp. 899–908. DOI: 10.1111/1365-2664.12251.

Elias, A. A. (2008). Towards a shared systems model of stakeholders in environmental conflict. *International Transactions in Operational Research* 15, pp. 239–253.

Elsawah, S., J. H. A. Guillaume, T. Filatova, J. Rook and A. J. Jakeman (2015). A methodology for eliciting, representing, and analysing stakeholder knowledge for decision making on complex socio-ecological systems: From cognitive maps to agent-based models. *Journal of Environmental Management* 151, pp. 500–516. DOI: 10.1016/j.jenvman.2014.11.028.

Elston, D. A., L. Spezia, D. Baines and S. M. Redpath (2014). Working with stakeholders to reduce conflict - modelling the impact of varying hen harrier Circus cyaneus densities on red grouse Lagopus lagopus populations. *Journal of Applied Ecology* 51, pp. 1236–1245. DOI: 10.1111/1365-2664.12315.

Eppinga, M. B., M. Rietkerk, M. J. Wassen, P. C. Ruiter, B. Eppinga Maarten, J. Wassen Martin, C. Ruiter Peter and P. C. De Ruiter (2009). Linking habitat modification to catastrophic shifts and vegetation patterns in bogs. *Plant Ecology* 200, pp. 53–68. DOI: 10.1007/s11258-007-9309-6.

Etienne, M., D. R. D. Toit and S. Pollard (2011). ARDI : A Co-construction Method for Participatory Modeling in Natural. *Ecology and Society* 16 (1). 44. http://www. ecologyandsociety.org/vol16/iss1/art44/, Online.

Evans, C. D., A. Bonn, J. Holden, M. S. Reed, M. G. Evans, F. Worrall, J. Couwenberg and M. Parnell (2014). Relationships between anthropogenic pressures and ecosystem functions in UK blanket bogs: Linking process understanding to ecosystem service valuation. *Ecosystem Services* 9, pp. 5–19. DOI: 10.1016/j.ecoser.2014.06.013.

Evans, D. M., M. J. O. Pocock and J. Memmott (2013a). The robustness of a network of ecological networks to habitat loss. *Ecology Letters* 16 (7), pp. 844–852. DOI: 10.1111/ele.12117.

Evans, M. R. (2012). Modelling ecological systems in a changing world. *Philosophical Transactions of the Royal Society B: Biological Sciences* 367 (1586), pp. 181–190. DOI: 10.1098/rstb.2011.0172.

Evans, M. R., K. J. Norris and T. G. Benton (2012). Predictive ecology: systems approaches. *Philosophical Transactions of the Royal Society B: Biological Sciences* 367 (1586), pp. 163–169. DOI: 10.1098/rstb.2011.0191.

Evans, M. R., V. Grimm, K. Johst, T. Knuuttila, R. de Langhe, C. M. Lessells, M. Merz, M. a. O'Malley, S. H. Orzack, M. Weisberg, D. J. Wilkinson, O. Wolkenhauer and T. G. Benton (2013b). Do simple models lead to generality in ecology? *Trends in Ecology & Evolution* 28 (10), pp. 578–583. DOI: 10.1016/j.tree.2013.05.022.

Evans, M., T. Allott, J. Holden, C. Flitcroft, A. Bonn, C. Brookes, S. Crowe, G. Hobson, S. Hodson, B. Irvine, T. James, L. Liddaman, S. Lindop, E. Maxfield, S. Mchale, S. Milner, S. Trotter and C. Worman (2005). *Understanding Gully Blocking in Deep Peat*. Tech. rep. 4. Moors for the Future.

Evans, M., J. Warburton and J. Yang (2006). Eroding blanket peat catchments: Global and local implications of upland organic sediment budgets. *Geomorphology* 79, pp. 45–57. DOI: 10.1016/j.geomorph.2005.09.015.

Evans, M. and J. Lindsay (2010). Impact of gully erosion on carbon sequestration in blanket peatlands. *Climate Research* 45, pp. 31–41. DOI: 10.3354/cr00887.

Fairweather, J. (2010). Farmer models of socio-ecologic systems: Application of causal mapping across multiple locations. *Ecological Modelling* 221 (3), pp. 555–562. DOI: 10.1016/j.ecolmodel.2009.10.026.

Farrell, C. A. and G. J. Doyle (2003). Rehabilitation of industrial cutaway Atlantic blanket bog in County Mayo, North-West Ireland. *Wetlands Ecology and Management* 11, pp. 21–35. DOI: 10.1023/A:1022097203946.

Fenner, N. and C. Freeman (2011). Drought-induced carbon loss in peatlands. *Nature Geoscience* 4, pp. 895–900. DOI: 10.1038/ngeo1323.

Ferguson, P., J. Lee and J. Bell (1978). Effects of sulphur pollutants on the growth of Sphagnum species. *Environmental Pollution* 16 (2), pp. 151–162. DOI: 10.1016/0013-9327(78)90129-5.

Foley Jonathan, A., R. Defries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. a. Howard, C. J. Kucharik, C. Monfreda, J. a. Patz, I. C. Prentice, N. Ramankutty and P. K. Snyder (2005). Global consequences of land use. *Science* 309, pp. 570–574. DOI: 10.1126/science.1111772.

Folke, C., Å. Jansson, J. Rockström, P. Olsson, S. R. Carpenter, F. S. Chapin, A.-S. Crépin, G. Daily, K. Danell, J. Ebbesson, T. Elmqvist, V. Galaz, F. Moberg, M. Nilsson, H. Österblom, E. Ostrom, Å. Persson, G. Peterson, S. Polasky, W. Steffen, B. Walker and F. Westley (2011). Reconnecting to the Biosphere. *Ambio* 40 (7), pp. 719–738. DOI: 10.1007/s13280-011-0184-y.

Foulds, S. a., J. Warburton and A. Foulds Simon (2007). Significance of wind-driven rain (wind-splash) in the erosion of blanket peat. *Geomorphology* 88 (2), pp. 183–192. DOI: 10.1016/j.geomorph.2006.07.001.

Freeman, C., C. D. Evans, D. T. Monteith, B. Reynolds and N. Fenner (2001). Export of organic carbon from peat soils. *Nature* 412, pp. 785–786. DOI: 10.1038/35090628.

Freeman, L. C. (1979). Centrality in social networks conceptual clarification. *Social Networks* 1 (3), pp. 215–239. DOI: 10.1016/0378-8733(78)90021-7.

Frolking, S., N. T. Roulet, T. R. Moore, P. J. H. Richard, M. Lavoie and S. D. Muller (2001). Modeling Northern Peatland Decomposition and Peat Accumulation. *Ecosystems* 4 (5), pp. 479–498. DOI: 10.1007/s10021-001-0105-1.

Frolking, S. and N. T. Roulet (2007). Holocene radiative forcing impact of northern peatland carbon accumulation and methane emissions. *Global Change Biology* 13 (5), pp. 1079–1088. DOI: 10.1111/j.1365-2486.2007.01339.x.

Frolking, S., N. T. Roulet, E. Tuittila, J. L. Bubier, A. Quillet and J. Talbot (2010). A new model of Holocene peatland net primary production, decomposition, water balance, and peat accumulation. *Earth system dynamics* 1, pp. 1–21. DOI: 10.5194/esd-1-1-2010.

Gallego-Sala, A. V., J. Clark, J. House, H. Orr, I. Prentice, P. Smith, T. Farewell and S. Chapman (2010). Bioclimatic envelope model of climate change impacts on blanket peatland distribution in Great Britain. *Climate Research* 45, pp. 151–162. DOI: 10.3354/cr00911.

Gallego-Sala, A. V. and C. Prentice (2012). Blanket peat biome endangered by climate change. *Nature Climate Change* 3, pp. 152–155. DOI: 10.1038/nclimate1672.

Gallego-Sala, A. V., D. J. Charman, S. P. Harrison, G. Li and I. C. Prentice (2016). Climate-driven expansion of blanket bogs in Britain during the Holocene. *Climate of the Past Discussions* 12 (5), pp. 129–136. DOI: 10.5194/cpd-11-4811-2015.

Gao, J., S. V. Buldyrev, H. E. Stanley and S. Havlin (2011). Networks formed from interdependent networks. *Nature Physics* 8, pp. 40–48. DOI: 10.1038/nphys2180.

Gao, J., Y.-Y. Liu, R. M. D'Souza and A.-L. Barabási (2014). Target control of complex networks. *Nature Communications* 5, p. 5415. DOI: 10.1038/ncomms6415.

Gao, J., B. Barzel and A.-L. Barabási (2016). Universal resilience patterns in complex networks. *Nature* 530, pp. 307–312. DOI: 10.1038/nature16948.

Gao, J., J. Holden and M. Kirkby (2015). A distributed TOPMODEL for modelling impacts of land-cover change on river flow in upland peatland catchments. *Hydrological Processes* 29 (13), pp. 2867–2879. DOI: 10.1002/hyp.10408.

Garnett, M., P. Ineson and A. Stevenson (2000). Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK. *The Holocene* 10 (6), pp. 729–736. DOI: 10.1191/09596830094971.

Gaveau, D. L. a., M. a. Salim, K. Hergoualc'h, B. Locatelli, S. Sloan, M. Wooster, M. E. Marlier, E. Molidena, H. Yaen, R. DeFries, L. Verchot, D. Murdiyarso, R. Nasi, P. Holmgren and D. Sheil (2014). Major atmospheric emissions from peat fires in Southeast Asia during non-drought years: evidence from the 2013 Sumatran fires. *Scientific reports* 4, 6112. DOI: 10.1038/srep06112.

Gershenson, C. and M. a. Niazi (2013). Multidisciplinary applications of complex networks modeling, simulation, visualization, and analysis. *Complex Adaptive Systems Modeling* 1 (1), pp. 1–4. DOI: 10.1186/2194-3206-1-17.

Ghoshal, G. and A.-L. Barabási (2011). Ranking stability and super-stable nodes in complex networks. *Nature communications* 2:394, p. 7. DOI: 10.1038/ncomms1396.

Gillespie, C. S. (2015). Fitting Heavy Tailed Distributions: The poweRlaw Package. *Journal of Statistical Software* 64 (2). http://www.jstatsoft.org/v64/i02/, pp. 1–16.

Girvan, M. and M. E. J. Newman (2002). Community structure in social and biological networks. *PNAS* 99 (12), pp. 7821–7826. DOI: 10.1073/pnas.122653799.

González, E., S. W. Henstra, L. Rochefort, G. E. Bradfield and M. Poulin (2013). Is rewetting enough to recover Sphagnum and associated peat-accumulating species in traditionally exploited bogs? *Wetlands Ecology and Management* 22 (1), pp. 49–62. DOI: 10.1007/s11273-013-9322-6.

Gorham, E. (1991). Northern Peatlands: Role in the Carbon Cycle and Probable Responses to Climatic Warming. *Ecological Applications* 1 (2), pp. 182–195.

Gramberger, M., K. Zellmer, K. Kok and M. J. Metzger (2014). Stakeholder integrated research (STIR): a new approach tested in climate change adaptation research. *Climatic Change* 128, pp. 201–214. DOI: 10.1007/s10584-014-1225-x.

Gray, S., A. Chan, D. Clark and R. Jordan (2012). Modeling the integration of stakeholder knowledge in social–ecological decision-making: Benefits and limitations to knowledge diversity. *Ecological Modelling* 229, pp. 88–96. DOI: 10.1016/j.ecolmodel.2011.09.011.

Gray, S. A., S. Gray, J. L. D. Kok, A. E. R. Helfgott, B. O. Dwyer, R. Jordan and A. Nyaki (2015). Using fuzzy cognitive mapping as a participatory approach to analyze change , preferred states , and perceived resilience of social-ecological systems. *Ecology and Society* 20 (2). 11. DOI: 10.5751/ES-07396-200211.

Grayson, R., J. Holden, R. Jones, J. Carle and a.R. Lloyd (2012). Improving particulate carbon loss estimates in eroding peatlands through the use of terrestrial laser scanning. *Geomorphology* 179, pp. 240–248. DOI: 10.1016/j.geomorph.2012.08.015.

Green, S. M., A. J. Baird, C. P. Boardman and V. Gauci (2014). A mesocosm study of the effect of restoration on methane (CH4) emissions from blanket peat. *Wetlands Ecology and Management* 22 (5), pp. 523–537. DOI: 10.1007/s11273-014-9349-3.

Griggs, D., M. Stafford-Smith, O. Gaffney, J. Rockström, M. C. Ohman, P. Shyamsundar, W. Steffen, G. Glaser, N. Kanie and I. Noble (2013). Policy: Sustainable development goals for people and planet. *Nature* 495, pp. 305–307. DOI: 10.1038/495305a.

Grimble, R. and K. Wellard (1997). Stakeholder methodologies in natural resource management: A review of principles, contexts, experiences and opportunities. *Agricultural Systems* 55 (2), pp. 173–193. DOI: 10.1016/S0308-521X(97)00006-1.

Grimm, V., E. Revilla, U. Berger, F. Jeltsch, W. M. Mooij, S. F. Railsback, H.-H. Thulke, J. Weiner, T. Wiegand and D. L. DeAngelis (2005). Pattern-oriented modeling of agent-based complex systems: lessons from ecology. *Science* 310, pp. 987–991. DOI: 10.1126/science.1116681.

Groumpos, P. P. (2014). Large Scale Systems and Fuzzy Cognitive Maps : A critical overview of challenges and research opportunities. *Annual Reviews in Control* 38 (1), pp. 93–102. DOI: 10.1016/j.arcontrol. 2014.03.009.

Gulbahce, N. and S. Lehmann (2008). The art of community detection. *BioEssays* 30 (10), pp. 934–8. DOI: 10.1002/bies.20820.

Gunderson, L. H. and C. S. Holling (2002). *Panarchy: Understadning transformations in human and natural systems*. WA, USA: Island Press.

Hagiwara, M. (1992). 'Extended fuzzy cognitive maps'. In: *Fuzzy Systems, 1992., IEEE International Conference on*, pp. 795–801. DOI: 10.1109/FUZZY.1992.258761.

Harris, M. P. K., K. a. Allen, H. a. McAllister, G. Eyre, M. G. Le Duc and R. H. Marrs (2011). Factors affecting moorland plant communities and component species in relation to prescribed burning. *Journal of Applied Ecology* 48 (6), pp. 1411–1421. DOI: 10.1111/j.1365-2664.2011.02052.x.

Hastings, A. and D. B. Wysham (2010). Regime shifts in ecological systems can occur with no warning. *Ecology Letters* 13 (4), pp. 464–472. DOI: 10.1111/j.1461-0248.2010.01439.x.

Heinemeyer, A., S. Croft, M. Garnett, E. Gloor, J. Holden, R. Lomas, M and P. Ineson (2010). The MILLENNIA peat cohort model: predicting past, present and future soil carbon budgets and fluxes under changing climates in peatlands. *Climate Research* 45, pp. 207–226. DOI: 10.3354/cr00928.

Helbing, D. (2013). Globally networked risks and how to respond. *Nature* 497, pp. 51–59. DOI: 10.1038/nature12047.

Helfter, C., C. Campbell, K. J. Dinsmore, J. Drewer, M. Coyle, M. Anderson, U. Skiba, E. Nemitz, M. F. Billett and M. A. Sutton (2015). Drivers of long-term variability in CO_2 net ecosystem exchange in a temperate peatland. *Biogeosciences* 12, pp. 1799–1811. DOI: 10.5194/bg-12-1799-2015.

Heppenstall, A., N. Malleson and A. Crooks (2016). "Space, the Final Frontier": How Good are Agent-Based Models at Simulating Individuals and Space in Cities? *Systems* 4 (1). 9. DOI: 10.3390/systems4010009.

Hilbert, D. W., N. Roulet and T. I. M. Moore (2000). Modelling and Analysis of Peatlands as Dynamical Systems. *Journal of Ecology* 88 (2), pp. 230–242.

Hobbs, B. F., S. A. Ludsin, R. L. Knight, P. A. Ryan, J. Biberhofer and J. J. H. Ciborowski (2002). Fuzzy cognitive mapping as a tool to define management objectives for complex ecosystems. *Ecological Applications* 12 (5), pp. 1548–1565.

Hoberg, G. and J. Phillips (2011). Playing Defence: Early Responses to Conflict Expansion in the Oil Sands Policy Subsystem. *Canadian Journal of Political Science* 44 (3), pp. 507–27. DOI: 10.1017/S0008423911000473.

Hodgson, A. M. (1992). Hexagons for systems thinking. *European Journal of Operational Research* 59, pp. 220–230. DOI: 10.1016/0377-2217(92)90019-6.

Hogg, E. H. (1993). Decay Potential of Hummock and Hollow Sphagnum peats at Different Depths in a Swedish Raised Bog. *Oikos* 66 (2), pp. 269–278.

Holden, J., T. P. Burt and N. J. Cox (2001). Macroporosity and infiltration in blanket peat: The implicaions of tension disc infiltrometer measurements. *Hydrological Processes* 15 (2), pp. 289–303. DOI: 10.1002/hyp.93.

Holden, J. and T. P. Burt (2003). Studies on blanket peat : the significance hydrological of the acrotelm-catotelm model. *Journal of Ecology* 91 (1), pp. 86–102.

Holden, J. (2005a). Controls of soil pipe frequency in upland blanket peat. *Journal of Geophysical Research* 110. F01002. DOI: 10.1098/rsta.2005.1671.

Holden, J. (2005b). Peatland hydrology and carbon release: why small-scale process matters. *Philosophical transactions. Series A, Mathematical, physical, and engineering sciences* 363, pp. 2891–2913. DOI: 10.1098/rsta.2005.1671.

Holden, J. (2005c). Piping and woody plants in peatlands: Cause or effect? *Water Resources Research* 41, W06009, p. 10. DOI: 10.1029/2004WR003909.

Holden, J., M. G. Evans, T. P. Burt and M. Horton (2006). Impact of land drainage on peatland hydrology. *Journal of environmental quality* 35 (5), pp. 1764–78. DOI: 10.2134/jeq2005.0477.

Holden, J., L. Shotbolt, A. Bonn, T. Burt, P. Chapman, A. Dougill, E. Fraser, K. Hubacek, B. Irvine, M. Kirkby, M. Reed, C. Prell, S. Stagl, L. Stringer, A. Turner and F. Worrall (2007). Environmental change in moorland landscapes. *Earth-Science Reviews* 82, pp. 75–100. DOI: 10.1016/j.earscirev.2007.01.003.

Holden, J., J. Walker, M. G. Evans, F. Worrall and A. Bonn (2008). *A Compendium of Peat Restoration and Management Projects. Defra report SP0556*. Tech. rep. DEFRA.

Holden, J., Z. Wallage, S. Lane and A. McDonald (2011). Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology* 402, pp. 103–114. DOI: 10.1016/j.jhydrol.2011.03.010.

Holden, J., P. J. Chapman, S. M. Palmer, P. Kay and R. Grayson (2012). The impacts of prescribed moorland burning on water colour and dissolved organic carbon: A critical synthesis. *Journal of Environmental Management* 101, pp. 92–103. DOI: 10.1016/j.jenvman.2012.02.002.

Holden, J., L. Brown, S. Palmer, K. Johnston and C. Wearing (2015). Impact of prescribed and repeated vegetation burning on blanket peat hydrology. *Water Resources Research* 51 (1-13). DOI: 10.1002/2014WR016782.

Holling, C. S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecological Systems* 4, pp. 1–23. DOI: 10.1146/annurev.es.04.110173.000245.

Holme, P., B. J. Kim, C. Yoon and S. K. Han (2002). Attack vulnerability of complex communication networks. *Physical Review E* 65, pp. 65–69. DOI: 10.1103/PhysRevE.65.056109.

Hossard, L., M. Jeuffroy, E. Pelzer, X. Pinochet and V. Souchere (2013). A participatory approach to design spatial scenarios of cropping systems and assess their effects on phoma stem canker management at a regional scale. *Environmental Modelling & Software* 48, pp. 17–26. DOI: 10.1016/j.envsoft.2013.05.014.

Hubacek, K., K. Dehnen-Schmutz, M. Qasim and M. Termansen (2009). Description of the upand economy. In: A. Bonn, T. Allott, K. Hubacek and J. Stewart (Eds.), *Drivers of Environmental Change in the Uplands*. Routledge. Chap. 16, pp. 294–308.

IPCC (2014). *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fith Assessment Report of the Intergovernmental Panel on Climate Change*. Tech. rep. IPCC, Geneva, p. 151.

Ireland, A. W. and R. K. Booth (2012). Upland deforestation triggered an ecosystem state-shift in a kettle peatland. *Journal of Ecology* 100 (3), pp. 586–596. DOI: 10.1111/j.1365-2745.2012.01961.x.

Isbell, F., D. Tilman, S. Polasky, S. Binder and P. Hawthorne (2013). Low biodiversity state persists two decades after cessation of nutrient enrichment. *Ecology Letters* 16 (4), pp. 454–460. DOI: 10.1111/ele.12066.

Ise, T., A. L. Dunn, S. C. Wofsy and P. R. Moorcroft (2008). High sensitivity of peat decomposition to climate change through water-table feedback. *Nature Geoscience* 1, pp. 763–766. DOI: 10.1038/ngeo331.

Jackson, S. T. and R. J. Hobbs (2009). Ecological restoration in the light of ecological history. *Science* 325, pp. 567–569. DOI: 10.1098/rsta.2005.1671.

Jeong, H., B. Tombor, R. Albert, Z. N. Oltvai and A.-L. Barabási (2000). The large-scale organization of metabolic networks. *Nature* 407, pp. 651–654.

Jetter, A. and W. Schweinfort (2011). Building scenarios with Fuzzy Cognitive Maps: An exploratory study of solar energy. *Futures* 43 (1), pp. 52–66. DOI: 10.1016/j.futures.2010.05.002.

Jetter, A. J. and K. Kok (2014). Fuzzy Cognitive Maps for futures studies—A methodological assessment of concepts and methods. *Futures* 61, pp. 45–57. DOI: 10.1016/j.futures.2014.05.002.

Jia, T., Y.-Y. Liu, E. Csóka, M. Pósfai, J.-J. Slotine and A.-L. Barabási (2013). Emergence of bimodality in controlling complex networks. *Nature communications* 4. DOI: 10.1038/ncomms3002.

JNCC (2015). *Natura 2000 - Standard Data Form. South Pennine Moors*. URL: http://jncc.defra.gov.uk/pdf/SPA/UK9007022.pdf (visited on 10/01/2016).

JNCC and Defra (2012). *JNCC and Defra (on behalf of the Four Countries' Biodiversity Group). UK Post-2010 Biodiversity Framework*. Tech. rep. JNCC, Peterborough, UK.

Jones, N. A., H. Ross, T. Lynam, P. Perez and A. Leitch (2011). Mental Model an Interdisciplinary Synthesis of Theory and Methods. *Ecology and Society* 16 (1). 46. http://www. ecologyandsociety.org/vol16/iss1/art46/.

Kates, R. W., W. C. Clark, R. Corell, J. M. Hall, C. C. Jaeger, I. Lowe, J. J. Mccarthy, H. J. Schellnhuber, B. Bolin, N. M. Dickson, S. Faucheux, G. C. Gallopin, A. Grübler, B. Huntley, J. Jäger, N. S, R. E. Kasperson, A. Mabogunje, P. Matson, H. Mooney, B. Moore, T. O. Riordan, U. Svedin and B. M. III (2001). Sustainability Science. *Science*. Faculty Research Working Paper Series 292, pp. 641–642. DOI: 10.1016/S0140-6736(08)61659-1.

Kates, R. (2011). From the Unity of Nature to Sustainability Science: Ideas and Practice. CID Working Paper No. 218. Cambridge, MA: Harvard University.

Kawakami, E., V. K. Singh, K. Matsubara, T. Ishii, Y. Matsuoka, T. Hase, P. Kulkarni, K. Siddiqui, J. Kodilkar, N. Danve, I. Subramanian, M. Katoh, Y. Shimizu-Yoshida, S. Ghosh, A. Jere and H. Kitano (2016). Network analyses based on comprehensive molecular interaction maps reveal robust control structures in yeast stress response pathways. *npj Systems Biology and Applications* 2, 15018. DOI: 10.1038/npjsba.2015.18.

Kéfi, S., V. Dakos, M. Scheffer, E. H. Van Nes and M. Rietkerk (2012). Early warning signals also precede non-catastrophic transitions. *Oikos* 122 (5), pp. 641–648. DOI: 10.1111/j.1600-0706.2012.20838.x.

Kelly, F. P., a. K. Maulloo and D. K. H. Tan (1998). Rate control for communication networks: shadow prices, proportional fairness and stability. *Journal of the Operational Research Society* 49 (3), pp. 237–252. DOI: 10.1057/palgrave.jors.2600523.

Kettridge, N. and J. M. Waddington (2014). Towards quantifying the negative feedback regulation of peatland evaporation to drought. *Hydrological Processes* 28 (11), pp. 3728–3740. DOI: 10.1002/hyp.9898.

Kleinen, T., V. Brovkin and R. J. Schuldt (2012). A dynamic model of wetland extent and peat accumulation: results for the Holocene. *Biogeosciences* 9, pp. 235–248. DOI: 10.5194/bg-9-235-2012.

Klerkx, L. and A. Proctor (2013). Beyond fragmentation and disconnect: Networks for knowledge exchange in the English land management advisory system. *Land Use Policy* 30 (1), pp. 13–24. DOI: 10.1016/j. landusepol.2012.02.003.

Knapp, C. N., M. Fernandez-Gimenez, E. Kachergis and A. Rudeen (2011). Using Participatory Workshops to Integrate State-and-Transition Models Created With Local Knowledge and Ecological Data. *Rangeland Ecology & Management* 64 (2), pp. 158–170. DOI: 10.2111/REM-D-10-00047.1.

Knight, C. J., D. J. Lloyd and A. S. Penn (2014). Linear and sigmoidal fuzzy cognitive maps: An analysis of fixed points. *Applied Soft Computing* 15, pp. 193–202. DOI: 10.1016/j.asoc.2013.10.030.

Kok, K. (2009). The potential of Fuzzy Cognitive Maps for semi-quantitative scenario development, with an example from Brazil. *Global Environmental Change* 19 (1), pp. 122–133. DOI: 10.1016/j.gloenvcha. 2008.08.003.

Konecny, K., U. Ballhorn, P. Navratil, J. Jubanski, S. E. Page, K. Tansey, A. Hooijer, R. Vernimmen and F. Siegert (2016). Variable carbon losses from recurrent fires in drained tropical peatlands. *Global Change Biology* 22 (4), pp. 1469–1480. DOI: 10.1111/gcb.13186.

Kontogianni, A., E. Papageorgiou, L. Salomatina, M. Skourtos and B. Zanou (2012). Risks for the Black Sea marine environment as perceived by Ukrainian stakeholders: A fuzzy cognitive mapping application. *Ocean & Coastal Management* 62, pp. 34–42. DOI: 10.1016/j.ocecoaman.2012.03.006.

Kosko, B. (1986). Fuzzy cognitive maps. *International Journal of Man-Machine Studies* 24, pp. 65–75.

Kosko, B. (1988). Hidden patterns in combined and adaptive knowledge networks. *International Journal of Approximate Reasoning* 2 (4), pp. 377–393. DOI: 10.1016/0888-613X(88)90111-9.

Kovács, I. A. and A.-L. Barabási (2015). Destruction perfected. *Nature* 524, pp. 38–39.

Krueger, T., T. Page, K. Hubacek, L. Smith and K. Hiscock (2012). The role of expert opinion in environmental modelling. *Environmental Modelling & Software* 36, pp. 4–18. DOI: 10.1016/j.envsoft.2012.01.011.

Krzywinski, M., I. Birol, S. J. Jones and M. A. Marra (2012). Hive plots–rational approach to visualizing networks. *Briefings in Bioinformatics* 13 (5), pp. 627–644. DOI: 10.1093/bib/bbr069.

Laiho, R. (2006). Decomposition in peatlands: Reconciling seemingly contrasting results on the impacts of lowered water levels. *Soil Biology and Biochemistry* 38 (8), pp. 2011–2024. DOI: 10.1016/j.soilbio.2006.02.017.

Lambin, E. F. and P. Meyfroidt (2011). Global land use change, economic globalization, and the looming land scarcity. *PNAS* 108 (9), pp. 3465–3472. DOI: 10.1073/pnas.1100480108.

Lamers, L. P. M., R. Bobbink and J. G. M. Roelofs (2000). Natural nitrogen filter fails in polluted raised bogs. *Global Change Biology* 6 (5), pp. 583–586.

Lane, D. C. (2008). The Emergence and Use of Diagramming in System Dynamics : A Critical Account. *Systems Research and Behavioral Science* 25, pp. 3–23. DOI: 10.1002/sres.826.

Lang, D. J., A. Wiek, M. Bergmann, M. Stauffacher, P. Martens, P. Moll, M. Swilling and C. J. Thomas (2012). Transdisciplinary research in sustainability science: Practice, principles, and challenges. *Sustainability Science* 7 (Supplement 1), pp. 25–43. DOI: 10.1007/s11625-011-0149-x.

Lapen, D. R., J. S. Price and R. Gilbert (2005). Modelling two-dimensional steady-state groundwater flow and flow sensitivity to boundary conditions in blanket peat complexes. *Hydrological Processes* 19 (2), pp. 371–386. DOI: 10.1002/hyp.1507.

Law, E., B. Bryan, E. Meijaard, T. Mallawaarachchi, M. Struebig and K. Wilson (2015). Ecosystem services from a degraded peatland of Central Kalimantan: implications for policy, planning and management. *Ecological Applications* 25 (1), pp. 70–87.

Lee, H., J. G. Alday, R. J. Rose, J. O'Reilly and R. H. Marrs (2013). Long-term effects of rotational prescribed burning and low-intensity sheep grazing on blanket-bog plant communities. *Journal of Applied Ecology* 50 (3), pp. 625–635. DOI: 10.1111/1365-2664.12078.

Levin, S. a. (1998). Biosphere Complex Adaptive Systems. *Ecosystems* 1 (5), pp. 431–436. DOI: 10.1007/s100219900037.

Lewis, C., J. Albertson, X. Xu and G. Kiely (2012). Spatial variability of hydraulic conductivity and bulk density along a blanket peatland hillslope. *Hydrological Processes* 26 (10), pp. 1527–1537. DOI: 10.1002/hyp.8252.

Li, D., B. Fu, Y. Wang, G. Lu, Y. Berezin and H. E. Stanley (2014). Percolation transition in dynamical traffic network with evolving critical bottlenecks. *PNAS* 112 (3), pp. 669–672. DOI: 10.1073/pnas.1419185112.

Li, P., J. Holden and B. Irvine (2015). Prediction of blanket peat erosion across Great Britain under environmental change. *Climatic Change* 134, pp. 177–191. DOI: 10.1007/s10584-015-1532-x.

Limpens, J., F. Berendse, C. Blodau, J. G. Canadell, C. Freeman, J. Holden, N. Roulet, H. Rydin and G. Schaepman-Strub (2008). Peatlands and the carbon cycle: from local processes to global implications – a synthesis. *Biogeosciences* 5 (5), pp. 1475–1491. DOI: 10.5194/bg-5-1475-2008.

Lin, C.-T. (1974). Structural controllability. *IEEE Transactions on Automatic Control* 19 (3). DOI: 10.1109/TAC.1974.1100557.

Lindsay, R. A. L., D. J. Charman, F. Everingham, R. M. O. Reilly, R. M. O'Reilly, M. A. Palmer, T. A. Rowel and D. A. Stroud (1988). *The Flow Country - The Peatlands of Caithness and Sutherland*. Tech. rep. JNCC, Peterborough.

Liu, J., T. Dietz, S. R. Carpenter, M. Alberti, C. Folke, E. Moran, A. N. Pell, P. Deadman, T. Kratz, J. Lubchenco, E. Ostrom, Z. Ouyang, W. Provencher, C. L. Redman, S. H. Schneider and W. W. Taylor (2007). Complexity of coupled human and natural systems. *Science* 317, pp. 1513–1516. DOI: 10.1126/science.1144004.

Liu, J., H. Mooney, V. Hull, S. J. Davis, J. Gaskell, T. Hertel, J. Lubchenco, K. C. Seto, P. Gleick, C. Kremen and S. Li (2015a). Systems integration for global sustainability. *Science* 347 (6225), pp. 1–9. DOI: 10.1126/science.1258832.

Liu, M., D. Li, P. Qin, C. Liu, H. Wang and F. Wang (2015b). Epidemics in Interconnected Small-World Networks. *Plos One* 10 (3), p. 9. DOI: 10.1371/journal.pone.0120701.

Liu, X., H. E. Stanley and J. Gao (2016). Breakdown of interdependent directed networks. *PNAS* 113 (5), pp. 1138–1143. DOI: 10.1073/pnas.1523412113.

Liu, Y.-Y., J.-J. Slotine and A.-L. Barabási (2011). Controllability of complex networks. *Nature* 473, pp. 167–173. DOI: 10.1038/nature10011.

Lombardi, A. and M. Hörnquist (2007). Controllability analysis of networks. *Physical Review E - Statistical, Nonlinear, and Soft Matter Physics* 75 (5), pp. 1–5. DOI: 10.1103/PhysRevE.75.056110.

Lorscheid, I., B. O. Heine and M. Meyer (2012). Opening the 'Black Box' of Simulations: Increased Transparency and Effective Communication Through the Systematic Design of Experiments. *Computational and Mathematical Organization Theory* 18 (1), pp. 22–62. DOI: 10.1007/s10588-011-9097-3.

Ludwig, D., B. Walker and C. S. Holling (1997). Sustainability, Stability, and Resilience. *Ecology and Society* 1 (1). 7. http://www.consecol.org/vol1/iss1/art7/.

Ludwig, D. (2001). The era of management is over. *Ecosystems* 4 (8), pp. 758–764. DOI: 10.1007/s10021-001-0044-x.

Lynam, T. and K. Brown (2011). Mental models in human–environment interactions: theory, policy implications, and methodological explorations. *Ecology and Society* 17 (3). 24. http://dx.doi.org/10.5751/ES-04257-170324. DOI: 10.5751/ES-04257-170324.

MacDougall, a. S., K. S. McCann, G. Gellner and R. Turkington (2013). Diversity loss with persistent human disturbance increases vulnerability to ecosystem collapse. *Nature* 494, pp. 86–90. DOI: 10.1038/nature11869.

Madden, F. and B. McQuinn (2014). Conservation's blind spot: The case for conflict transformation in wildlife conservation. *Biological Conservation* 178, pp. 97–106. DOI: 10.1016/j.biocon.2014.07.015.

Maddock, A. (2011). *UK Biodiversity action plan; priority habitat descriptions.* Tech. rep. Peterborough: JNCC.

Maltby, E. (2010). Effects of climate change on the societal benefits of UK upland peat ecosystems: Applying the ecosystem approach. *Climate Research* 45, pp. 249–259. DOI: 10.3354/cr00893.

Martin, T. G. and J. E. M. Watson (2016). Intact ecosystems provide best defence against climate change. *Nature Climate Change* 6, pp. 122–124. DOI: 10.1038/nclimate2918.

Mathieu, J. E., T. S. Heffner, G. F. Goodwin, K. Hobson, K. Ivory, M. Trip and N. Windefelder (2000). The influence of shared mental models on team process and performance. *Journal of Applied Psychology* 85 (2), pp. 273–283. DOI: 10.I037t/0021-9010.85.2.273.

Maxwell, P. S., K. A. Pitt, A. D. Olds, D. Rissik and R. M. Connolly (2015). Identifying habitats at risk : simple models can reveal complex ecosystem dynamics. *Ecological Society of America* 25 (2), pp. 573–587.

May, R. M. (1977). Thresholds and breakpoints in ecosystems with a multiplicity of stable states. *Nature* 269, pp. 471–477.

Mcpherson, J. M., J. Sammy, D. J. Sheppard, J. J. Mason and T. A. Brichieri-colombi (2016). Integrating traditional knowledge when it appears to conflict with conservation : lessons from the discovery and protection of sitatunga in. *Ecology and Society* 21 (1). 24. http://dx.doi.org/10.5751/ES-08089-210124.

McShane, T. O., P. D. Hirsch, T. C. Trung, A. N. Songorwa, A. Kinzig, B. Monteferri, D. Mutekanga, H. V. Thang, J. L. Dammert, M. Pulgar-Vidal, M. Welch-Devine, J. Peter Brosius, P. Coppolillo and S. O'Connor (2011). Hard choices: Making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144 (3), pp. 966–972. DOI: 10.1016/j.biocon.2010.04.038.

Mei, S. (2014). Individual Decision Making Can Drive Epidemics: A Fuzzy Cognitive Map Study. *IEE Transactions on Fuzzy Systems* 22 (2), pp. 264–273. DOI: 10.1109/TFUZZ.2013.2251638.

Mendonça, M., B. Angelico, L. Arruda and F. Neves (2013). A dynamic fuzzy cognitive map applied to chemical process supervision. *Engineering Applications of Artificial Intelligence* 26 (4), pp. 1199–1210. DOI: 10.1016/j.engappai.2012.11.007.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: Synthesis*. Tech. rep. Island Press, Washington, D.C.

Millington, J. D., D. Demeritt and R. Romero-Calcerrada (2011). Participatory evaluation of agent-based land-use models. *Journal of Land Use Science* 6 (2-3), pp. 195–210. DOI: 10.1080/1747423X.2011.558595.

Montoya, J. M., S. L. Pimm and R. V. Solé (2006). Ecological networks and their fragility. *Nature* 442, pp. 259–264. DOI: 10.1038/nature04927.

Moon, K. and V. M. Adams (2016). Using quantitative influence diagrams to map natural resource managers' mental models of invasive species management. *Land Use Policy* 50, pp. 341–351. DOI: 10.1016/j.landusepol.2015.10.013.

Moore, P. D. (1973). The Influence of Prehistoric Cultures upon the Initiation and Spread of Blanket Bog in Upland Wales. *Nature* 241, pp. 350–353. DOI: 10.1038/241350a0.

Moore, P. D. (1975). Origin of blanket mires. *Nature* 256 (5515), pp. 267–269. DOI: 10.1038/256267a0.

Moore, P. D. (1987). Ecological and hydrological aspects of peat formation. *Geological Society, London, Special Publications* 32, pp. 7–15. DOI: 10.1144/GSL.SP.1987.032.01.02.

Moore, S., C. D. Evans, S. E. Page, M. H. Garnett, T. G. Jones, C. Freeman, A. Hooijer, A. J. Wiltshire, S. H. Limin and V. Gauci (2013). Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. *Nature* 493, pp. 660–664. DOI: 10.1038/nature11818.

Morone, F. and H. a. Makse (2015). Influence maximization in complex networks through optimal percolation. *Nature* 524, pp. 65–68. DOI: 10.1038/nature14604.

Morris, P. J., A. J. Baird and L. R. Belyea (2011a). The DigiBog peatland development model 2: ecohydrological simulations in 2D. *Ecohydrology* 5 (3), pp. 256–268. DOI: 10.1002/eco.229.

Morris, P. J., L. R. Belyea and A. J. Baird (2011b). Ecohydrological feedbacks in peatland development: a theoretical modelling study. *Journal of Ecology* 99 (5), pp. 1190–1201. DOI: 10.1111/j.1365-2745.2011.01842.x.

Morris, P. J., A. J. Baird, D. M. Young and G. T. Swindles (2015a). Untangling climate signals from autogenic changes in long-term peatland development. *Geophysical Research Letters* 42. DOI: 10.1002/2015GL066824.

Morris, P. J., A. J. Baird and L. R. Belyea (2015b). Bridging the gap between models and measurements of peat hydraulic conductivity. *Water Resources Research* 51, pp. 5353–5364. DOI: 10.1002/2015WR017264.

Nakamura, K., S. Iwai and T. Sawaragi (1982). Decision Support Using Causation Knowledge Base. *IEEE Transactions on Systems, Man and Cybernetics* 12 (6), pp. 765–777.

Natural England (2009). *Vital Uplands*. URL: http://webarchive.nationalarchives.gov.uk/20111121191513/http:/www.naturalengland.org.uk/about_us/news/2009/121109.aspx (visited on 10/06/2015).

Natural England (2014). *Site Improvement Plan South Pennine Moors*. URL: http://publications.naturalengland.org.uk/publication/5412834661892096 (visited on 18/01/2016).

Neocleoua, C., M. Papaioannou and C. Schizas (2011). 'Important issues to be considered in developing fuzzy cognitive maps'. In: *Fuzzy Systems (FUZZ), 2011 IEEE International Conference on*, pp. 662–665. DOI: 10.1109/FUZZY.2011.6007694.

Newig, J. and O. Fritsch (2009). Environmental governance: Participatory, multi-level - and effective? *Environmental Policy and Governance* 19, pp. 197–214. DOI: 10.1002/eet.509.

Newman, M. E. J. and M. Girvan (2003). 'Mixing Patterns and Community Structure in Networks'. In: R. Pastor-Satorras, M. Rubi and A. Diaz-Guilera (Eds.), *Statistical Mechanics of Complex Networks*. Vol. 625. Lecture Notes in Physics, Berlin Springer Verlag, pp. 66–87. DOI: 10.1007/978-3-540-44943-0_5. eprint: cond-mat/0210146.

Newman, M. E. J. (2006). Finding community structure in networks using the eigenvectors of matrices. *Phys. Rev. E* 74 (3), p. 036104. DOI: 10.1103/PhysRevE.74.036104.

Newman, M. E. J. (2010). *Networks: An Introduction*. Oxford University Press, UK.

Newman, M. E. J. and C. R. Ferrario (2013). Interacting Epidemics and Coinfection on Contact Networks. *PLoS ONE* 8 (8), pp. 1–8. DOI: 10.1371/journal.pone.0071321. arXiv: 1305.4648.

Noordwijk, M. van, R. Matthews, F. Agus, J. Farmer, L. Verchot, K. Hergoualc'h, S. Persch, H. L. Tata, B. Lusiana, A. Widayati and S. Dewi (2014). Mud, muddle and models in the knowledge value-chain to action on tropical peatland conservation. *Mitigation and Adaptation Strategies for Global Change* 19 (6), pp. 887–905. DOI: 10.1007/s11027-014-9576-1.

North, M. J. (2014). A theoretical formalism for analyzing agent-based models. *Complex Adaptive Systems Modeling* 2 (1), pp. 1–34. DOI: 10.1186/2194-3206-2-3.

Olito, C. and J. W. Fox (2015). Species traits and abundances predict metrics of plant-pollinator network structure, but not pairwise interactions. *Oikos* 124 (4), pp. 428–436. DOI: 10.1111/oik.01439.

Özesmi, U. and S. L. Özesmi (2004). Ecological models based on people's knowledge: a multi-step fuzzy cognitive mapping approach. *Ecological Modelling* 176, pp. 43–64. DOI: 10.1016/j.ecolmodel.2003.10.027.

Paavola, J., A. Gouldson and T. Kluvánková-Oravská (2009). Interplay of actors, scales, frameworks and regimes in the governance of biodiversity. *Environmental Policy and Governance* 19 (3), pp. 148–158. DOI: 10.1002/eet.505.

Palmer, S., I. Gordon, A. Hester and R. Pakeman (2004). Introducing spatial grazing impacts into the prediction of moorland vegetation dynamics. *Landscape Ecology* 19 (8), pp. 817–827. DOI: 10.1007/s10980-004-0094-5.

Palmer, W. C. and A. V. Havens (1958). A graphical technique for determining evapotranspiration by the thornthwaite method. *Monthly Weather Review* 86, pp. 123–128. DOI: 10.1175/1520-0493(1958)086<0123:AGTFDE>2.0.CO;2.

Pan, Y., A. Birdsey, Richard, J. Fang, R. Houghton, P. E. Kauppi, W. A. Kurz, O. L. Phillips, A. Shvidenko, S. L. Lewis, J. G. Canadell, P. Ciais, R. B. Jackson, S. W. Pacala, A. D. McGuire, S. Piao, A. Rautiainen, S. Sitch and D. Hayes (2011). A large and persistent carbon sink in the world's forests. *Science* 333, pp. 988–993. DOI: 10.1126/science.1201609.

Papageorgiou, E. I., A. T. Markinos and T. A. Gemtos (2011). Fuzzy cognitive map based approach for predicting yield in cotton crop production as a basis for decision support system in precision agriculture application. *Applied Soft Computing Journal* 11 (4), pp. 3643–3657. DOI: 10.1016/j.asoc.2011.01.036.

Papageorgiou, E. I. and J. L. Salmeron (2012). Learning Fuzzy Grey Cognitive Maps using Nonlinear Hebbian-based approach. *International Journal of Approximate Reasoning* 53 (1), pp. 54–65. DOI: 10.1016/j.ijar.2011.09.006.

Papageorgiou, E. I. and J. Salmeron (2013). A Review of Fuzzy Cognitive Maps research during the last decade. *IEE Transactions on Fuzzy Systems* 21 (1), pp. 66–79.

Papale, D. and R. Valentini (2003). A new assessment of European forests carbon exchanges by eddy fluxes and artificial neural network spatialization. *Global Change Biology* 9, pp. 525–535.

Parish, F., A. Sirin, D. Charman, H. Joosten, T. Minayeva, M. Silvus and L. C. Stringer (2008). *Assessment on peatlands, biodiversity and climate change*. Tech. rep. Global Environment Centre, Kuala Lumpur & Wetlands International, Wageningen, p. 215.

Parker, D. C., S. M. Manson, M. a. Janssen, M. J. Hoffmann and P. Deadman (2003). Multi-agent systems for the simulation of land-use and land-cover change: A review. *Annals of the Association of American Geographers* 93 (2), pp. 314–337. DOI: 10.1111/1467-8306.9302004.

Parry, L. E. and D. J. Charman (2013). Modelling soil organic carbon distribution in blanket peatlands at a landscape scale. *Geoderma* 211-212, pp. 75–84. DOI: 10.1016/j.geoderma.2013.07.006.

Parry, L. E., J. Holden and P. J. Chapman (2014). Restoration of blanket peatlands. *Journal of environmental management* 133, pp. 193–205. DOI: 10.1016/j.jenvman.2013.11.033.

Patterson, G. and R. Anderson (2000). *Forests and peatland habitats: guideline note*. Tech. rep. Forestry Commission, Edinburgh, pp. 1–16.

Peatland restoration - what's in it for me? (2015). URL: https://uplandsalliance.wordpress.com/2015/05/26/peatland-restoration-whats-in-it-for-me/ (visited on 10/06/2015).

Penn, A. S., C. J. K. Knight, D. J. B. Lloyd, D. Avitabile, K. Kok, F. Schiller, A. Woodward, A. Druckman and L. Basson (2013). Participatory Development and Analysis of a Fuzzy Cognitive Map of the Establishment of a Bio-based Economy in the Humber Region. *PloS one* 8 (11). e78319. DOI: 10.1371/journal.pone.0078319.

Penn, A. S., C. J. K. Knight, G. Chalkias and D. J. B. Lloyd (2014). Extending Participatory Fuzzy Cognitive Mapping with a Control Nodes Methodology: a case study of the development bio-based economy in the Humber region, UK. In: S. Gray, R. Jordan and M. Pallisimio (Eds.), *Including Stakeholders in Environmental Modeling: Considerations, Methods and Applications*. Forthcoming.

Percival, S. M. (2005). Birds and windfarms: What are the real issues? *British Birds* 98 (4), pp. 194–204.

Petrescu, A. M. R. et al. (2015). The uncertain climate footprint of wetlands under human pressure. *PNAS* 112 (15), pp. 4594–4599. DOI: 10.1073/pnas.1416267112.

Pilkington, M. G. (2015). *Background, location, design and restoration*. Tech. rep. Moors for the Future Partnership.

Pilkington, M. G., J. Walker, R. Maskill, T. Allott and M. G. Evans (2015). *Restoration of Blanket bogs; flood risk reduction and other ecosystem benefits. Final report of the Making Space for Water project*. Tech. rep. Moors for the Future Partnership.

Pocock, M. J. O., D. M. Evans and J. Memmott (2012). The Robustness and Restoration of a Network of Ecological Networks. *Science* 335, pp. 973–977. DOI: 10.1126/science.1214915.

Pocock, M. J., D. M. Evans, C. Fontaine, M. Harvey, R. Julliard, Ó. McLaughlin, J. Silvertown, A. Tamaddoni-Nezhad, P. C. White and D. A. Bohan (2016). The Visualisation of Ecological Networks, and Their Use as a Tool for Engagement, Advocacy and Management. In: G. Woodward and D. A. Bohan (Eds.), *Ecosystem Services: From Biodiversity to Society, Part 2*. Vol. 54. Elsevier. Chap. 2, pp. 41–85. DOI: 10.1016/bs.aecr.2015.10.006.

Polk, M. (2014). Achieving the promise of transdisciplinarity: a critical exploration of the relationship between transdisciplinary research and societal problem solving. *Sustainability Science* 9 (4), pp. 439–451. DOI: 10.1007/s11625-014-0247-7.

Prager, S. D. and C. Pfeifer (2015). Network approaches for understanding rainwater management from a social–ecological systems perspective. *Ecology and Society* 20 (4). 13. http://dx.doi.org/10.5751/ES-07950-200413.

Prell, C., K. Hubacek, M. Reed, C. Quinn, N. Jin, J. Holden, T. Burt, M. Kirby and J. Sendzimir (2007). If you have a hammer everything looks like a nail: traditional versus participatory model building. *Interdisciplinary Science Reviews* 32 (3), pp. 263–282. DOI: 10.1179/030801807X211720.

Prell, C., K. Hubacek, C. Quinn and M. Reed (2008). 'Who's in the network?' When stakeholders influence data analysis. *Systemic Practice and Action Research* 21 (6), pp. 443–458. DOI: 10.1007/s11213-008-9105-9.

Prell, C., K. Hubacek and M. Reed (2009). Stakeholder Analysis and Social Network Analysis in Natural Resource Management. *Society & Natural Resources* 22 (6), pp. 501–518. DOI: 10.1080/08941920802199202.

Price, J. S., A. L. Heathwaite and A. J. Baird (2003). Hydrological processes in abandoned and restored peatlands : An overview of management approaches. *Wetlands Ecology and Management* 11, pp. 65–83.

Pu, C.-L., W.-J. Pei and A. Michaelson (2012). Robustness analysis of network controllability. *Physica A: Statistical Mechanics and its Applications* 391 (18), pp. 4420–4425. DOI: 10.1016/j.physa.2012.04.019.

R Core Team (2015). *R: A Language and Environment for Statistical Computing*. https://www.R-project.org. R Foundation for Statistical Computing. Vienna, Austria.

Ramchunder, S. J., L. E. Brown and J. Holden (2013). Rotational vegetation burning effects on peatland stream ecosystems. *Journal of Applied Ecology* 50 (3), pp. 636–648. DOI: 10.1111/1365-2664.12082.

Ramirez, J. A., A. J. Baird, T. J. Coulthard and J. M. Waddington (2015). Ebullition of methane from peatlands: Does peat act as a signal shredder ? *Geophysical Research Letters* 42. DOI: 10.1002/2015GL063469.

Ramsey, D. and C. Veltman (2005). Predicting the effects of perturbations on ecological communities: what can qualitative models offer? *Journal of Animal Ecology* 74 (5), pp. 905–916. DOI: 10.1111/j.1365-2656.2005.00986.x.

Raymond, C. M., I. Fazey, M. S. Reed, L. C. Stringer, G. M. Robinson and A. C. Evely (2010). Integrating local and scientific knowledge for environmental management. *Journal of environmental management* 91 (8), pp. 1766–1777. DOI: 10.1016/j.jenvman.2010.03.023.

Reckien, D. (2014). Weather extremes and street life in India—Implications of Fuzzy Cognitive Mapping as a new tool for semi-quantitative impact assessment and ranking of adaptation measures. *Global Environmental Change* 26, pp. 1–13. DOI: 10.1016/j.gloenvcha.2014.03.005.

Redpath, S. M., J. Young, A. Evely, W. M. Adams, W. J. Sutherland, A. Whitehouse, A. Amar, R. a. Lambert, J. D. C. Linnell, A. Watt and R. J. Gutiérrez (2013). Understanding and managing conservation conflicts. *Trends in Ecology and Evolution* 28 (2), pp. 100–109. DOI: 10.1016/j.tree.2012.08.021.

Redpath, S. M., S. Bhatia and J. Young (2015). Tilting at wildlife: reconsidering human–wildlife conflict. *Oryx* 49 (2), pp. 222–225. DOI: 10.1017/S0030605314000799.

Reed, M. S., A. J. Dougill and M. J. Taylor (2007). Integrating local and scientific knowledge for adaptation to land degradation: Kalahari rangeland management options. *Land Degradation and Development* 18, pp. 249–268. DOI: 10.1002/ldr.777.

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation* 141 (10), pp. 2417–2431. DOI: 10.1016/j.biocon.2008.07.014.

Reed, M. S., A. Bonn, W. Slee, N. Beharry-Borg, J. Birch, I. Brown, T. Burt, D. Chapman, P. Chapman, G. Clay, S. Cornell, E. Fraser, J. Glass, J. Holden, J. Hodgson, K. Hubacek, B. Irvine, N. Jin, M. Kirkby, W. Kunin, O. Moore, D. Moseley, C. Prell, M. Price, C. Quinn, S. Redpath, C. Reid, S. Stagl, L. Stringer, M. Termansen, S. Thorp, W. Towers and F. Worrall (2009a). The future of the uplands. *Land Use Policy* 26 (Supplement 1), S204–S216. DOI: 10.1016/j.landusepol.2009.09.013.

Reed, M. S., K. Arblaster, C. Bullock, R. Burton, A. Davies, J. Holden, K. Hubacek, R. May, J. Mitchley, J. Morris, D. Nainggolan, C. Potter, C. Quinn, V. Swales and S. Thorp (2009b). Using scenarios to explore UK upland futures. *Futures* 41 (9), pp. 619–630. DOI: 10.1016/j.futures.2009.04.007.

Reed, M. S., A. Graves, N. Dandy, H. Posthumus, K. Hubacek, J. Morris, C. Prell, C. Quinn and L. C. Stringer (2009c). Who's in and why? A typology of stakeholder analysis methods for natural resource management. *Journal of Environmental Management* 90 (5), pp. 1933–1949. DOI: 10.1016/j.jenvman.2009.01.001.

Reed, M. S., A. Bonn, C. Evans, H. Joosten, B. Bain, J. Farmer, I. Emmer, J. Couwenberg, A. Moxey, R. Artz, F. Tanneberger, M. Von Unger, M. Smyth, R. Birnie, I. Inman, S. Smith, T. Quick, C. Cowap, S. Prior and R. A. Lindsay (2013a). *Peatland Code research project. Final report*. Tech. rep. Available online at: http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=18642. Defra, London.

Reed, M. S., K. Hubacek, A. Bonn, T. P. Burt, J. Holden, L. C. Stringer, N. Beharry-Borg, S. Buckmaster, D. Chapman, P. J. Chapman, G. D. Clay, S. J. Cornell, A. J. Dougill, A. C. Evely, E. D. G. Fraser, N. Jin, B. J. Irvine, M. J. Kirkby, W. E. Kunin, C. Prell, C. H. Quinn, B. Slee, S. Stagl, M. Termansen, S. Thorp and F. Worrall (2013b). Anticipating and Managing Future Trade-offs and Complementarities between Ecosystem Services. *Ecology and Society* 18 (1). 5. http://dx.doi.org/10.5751/ES-04924-180105.

Reed, M. S., J. Kenter, A. Bonn, D. Nainggolan, K. Broad, C. Quinn, T. Burt, L. Stringer, I. Fazey, F. Ravera, E. Fraser and K. Hubacek (2013c). Participatory scenario development for environmental management. *Jounrnal of Environmental Management* 128, pp. 345–362.

Reed, M. S., A. Moxey, K. Prager, N. Hanley, J. Skates, A. Bonn, C. D. Evans, K. Glenk and K. Thomson (2014a). Improving the link between payments and the provision of ecosystem services in agri-environment schemes. *Ecosystem Services* 9, pp. 44–53. DOI: 10.1016/j.ecoser.2014.06.008.

Reed, M. S., L. Stringer, I. Fazey, A. Evely and J. Kruijsen (2014b). Five principles for the practice of knowledge exchange in environmental management. *Journal of Environmental Management* 146, pp. 337–345. DOI: 10.1016/j.jenvman.2014.07.021.

Reenberg, A. and N. A. Fenger (2011). Globalising land use transitions: the soybean acceleration. *Danish Journal of Geography* 111 (1), pp. 85–92.

Reichardt, J. and S. Bornholdt (2006). Statistical mechanics of community detection. 74 (1), 016110, p. 016110. DOI: 10.1103/PhysRevE.74.016110. eprint: cond-mat/0603718.

Reynolds, J. F., J. F. Reynolds, D. M. S. Smith, E. F. Lambin, B. L. T. Ii, M. Mortimore, S. P. J. Batterbury, T. E. Downing, H. Dowlatabadi, R. J. Fernández, J. E. Herrick, E. Huber-sannwald and H. Jiang (2009). Global Desertification : Building a Science for Dryland Development. *Science* 316 (5826), pp. 847–851. DOI: 10.1126/science.1131634.

Rickard, J. T., S. Member, J. Aisbett, R. R. Yager and L. Fellow (2015). A new fuzzy cogntive map structure based on the weighted power mean. *IEEE Transactions on Fuzzy Systems* 23 (6), pp. 2188–2201.

Ritchey, T. (2013). Wicked Problems. Modelling social messes with morphological analysis. *Acta Morphologica Generalis* 2 (1), pp. 1–8.

Rittel, H. W. J. and M. M. Webber (1973). Dilemmas in a General Theory of Planning. *Policy Sciences* 4, pp. 155–169.

Rockstrom, J., W. Steffen, K. Noone, A. Persson, F. S. Chapin, E. F. Lambin, T. M. Lenton, M. Scheffer, C. Folke, H. J. Schellnhuber, B. Nykvist, C. A. de Wit, T. Hughes, S. van der Leeuw, H. Rodhe, S. Sorlin, P. K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R. W. Corell, V. J. Fabry, J. Hansen,

B. Walker, D. Liverman, K. Richardson, P. Crutzen and J. A. Foley (2009). A safe operating space for humanity. *Nature* 461, pp. 472–475. DOI: 10.1038/461472a.

Rodwell, J. S. (1991). *British Plant Communities. Volume 2. Mires and heath*. Vol. 2. Cambridge University Press.

Rooney, R. C., S. E. Bayley and D. W. Schindler (2012). Oil sands mining and reclamation cause massive loss of peatland and stored carbon. *PNAS* 109 (13), pp. 4933–4937. DOI: 10.1073/pnas.1117693108.

RoTAP (2012). *Review of Transboundary Air Pollution: Acidification, Eutrophication, Ground Level Ozone and Heavy Metals in the UK. Summary for Policy Makers. Contract Report to the Department for Environment, Food and Rural Affairs.* Tech. rep. Centre for Ecology & Hydrology, Oxford.

Rouwette, E. A. J. A. and J. A. M. Vennix (2006). Systems dynamics and organisational interventions. *Systems Research and Behavioral Science* 23, pp. 451–466. DOI: 10.1002/sres.772.

Rudd, M. A. (2011). How research-prioritization exercises affect conservation policy. *Conservation biology : the journal of the Society for Conservation Biology* 25 (5), pp. 860–866. DOI: 10.1111/j.1523-1739.2011.01712.x.

Saarikoski, H., J. Mustajoki and M. Marttunen (2013a). Participatory multi-criteria assessment as 'opening up' vs. 'closing down' of policy discourses: A case of old-growth forest conflict in Finnish Upper Lapland. *Land Use Policy* 32, pp. 329–336. DOI: 10.1016/j.landusepol.2012.11.003.

Saarikoski, H., K. Raitio and J. Barry (2013b). Understanding 'successful' conflict resolution: Policy regime changes and new interactive arenas in the Great Bear Rainforest. *Land Use Policy* 32, pp. 271–280. DOI: 10.1016/j.landusepol.2012.10.019.

Scheffer, M., S. Carpenter, J. a. Foley, C. Folke and B. Walker (2001). Catastrophic shifts in ecosystems. *Nature* 413, pp. 591–596. DOI: 10.1038/35098000.

Scheffer, M., S. R. Carpenter, T. M. Lenton, J. Bascompte, W. Brock, V. Dakos, J. van de Koppel, I. A. van de Leemput, S. a. Levin, E. H. van Nes, M. Pascual and J. Vandermeer (2012). Anticipating critical transitions. *Science* 338, pp. 344–348. DOI: 10.1126/science.1225244.

Schimelpfenig, D. W., D. J. Cooper and R. A. Chimner (2013). Effectiveness of Ditch Blockage for Restoring Hydrologic and Soil Processes in Mountain Peatlands. *Restoration Ecology* 22 (2), pp. 257–265. DOI: 10.1111/rec.12053.

Scott, R. J., R. Y. Cavana and D. Cameron (2013). Evaluating immediate and long-term impacts of qualitative group model building workshops on participants' mental models. *System Dynamics Review* 29 (4), pp. 216–236. DOI: 10.1002/sdr.1505.

Secretariat of the CBD (2010). *Conference of the Parties 10 Decision X/2*. Tech. rep. October 2010. Tenth Meeting. Nagoya, Japan. Available from: https://www.cbd.int/doc/decisions/cop-10/cop-10-dec-02-en.pdf.

Sedlacko, M., A. Martinuzzi, I. Røpke, N. Videira and P. Antunes (2014). Participatory systems mapping for sustainable consumption: Discussion of a method promoting systemic insights. *Ecological Economics* 106, pp. 33–43. DOI: 10.1016/j.ecolecon.2014.07.002.

Shoreman-Ouimet, E. and H. Kopnina (2015). Reconciling ecological and social justice to promote biodiversity conservation. *Biological Conservation* 184, pp. 320–326. DOI: 10.1016/j.biocon.2015.01.030.

Smith, J., D. R. Nayak and P. Smith (2014). Wind farms on undegraded peatlands are unlikely to reduce future carbon emissions. *Energy Policy* 66, pp. 585–591. DOI: 10.1016/j.enpol.2013.10.066.

Soler, L. S., K. Kok, G. Camara and A. Veldkamp (2012). Using fuzzy cognitive maps to describe current system dynamics and develop land cover scenarios: a case study in the Brazilian Amazon. *Journal of Land Use Science* 7 (2), pp. 149–175. DOI: 10.1080/1747423X.2010.542495.

Soliva, R., K. Rønningen, I. Bella, P. Bezak, T. Cooper, B. E. Flø, P. Marty and C. Potter (2008). Envisioning upland futures: Stakeholder responses to scenarios for Europe's mountain landscapes. *Journal of Rural Studies* 24 (1), pp. 56–71. DOI: 10.1016/j.jrurstud.2007.04.001.

Stone-Jovicich, S. S., T. Lynam, A. Leitch and N. A. Jones (2011). Using Consensus Analysis to Assess Mental Models about Water Use and Management in the Crocodile River Catchment, South Africa. *Ecology And Society* 16 (1). 45. http://www.ecologyandsociety.org/vol16/iss1/art45/.

Strack, M. and Y. C. a. Zuback (2013). Annual carbon balance of a peatland 10 yr following restoration. *Biogeosciences* 10 (5), pp. 2885–2896. DOI: 10.5194/bgd-9-17203-2012.

Strogatz, S. H. (2001). Exploring complex networks. *Nature* 410, pp. 268–276.

Stylios, C. and P. Groumpos (1999). A Soft Computing Approach for Modelling the Supervisor of Manufacturing Systems. *Intelligent and Robotic Systems* 26, pp. 389–403.

Suweis, S., F. Simini, J. R. Banavar and A. Maritan (2013). Emergence of structural and dynamical properties of ecological mutualistic networks. *Nature* 500, pp. 449–452. DOI: 10.1038/nature12438.

Swindles, G. T., P. J. Morris, A. J. Baird, M. Blaauw and G. Plunkett (2012). Ecohydrological feedbacks confound peat-based climate reconstructions. *Geophysical Research Letters* 39 (11). L11401. DOI: 10.1029/2012GL051500.

Swindles, G. T., P. J. Morris, D. Mullan, E. J. Watson, E. Turner, T. P. Roland, M. J. Amesbury, U. Kokfelt, K. Schoning, S. Pratte, A. Gallego-Sala, D. J. Charman, N. Sanderson, M. Garneau, J. Carrivick, C. Woulds, J. Holden, L. Parry and J. M. Galloway (2015). The long-term fate of permafrost peatlands under rapid climate warming. *Nature Scientific Reports* 5:17951. DOI: 10.1038/srep17951.

Tahvanainen, T. (2011). Abrupt ombrotrophication of a boreal aapa mire triggered by hydrological disturbance in the catchment. *Journal of Ecology* 99 (2), pp. 404–415. DOI: 10.1111/j.1365-2745.2010.01778.x.

Tallis, J. H. (1985). Mass Movement and Erosion of a Southern Pennine Blanket Peat. *Journal of Ecology* 73 (1), pp. 283–315.

Tallis, J. H. (1987). Fire and Flood at Holme Moss : Erosion Processes in an Upland Blanket Mire. *Journal of Ecology* 75 (4), pp. 1099–1129.

Tassa, T. (2012). Finding all maximally-matchable edges in a bipartite graph. *Theoretical Computer Science* 423, pp. 50–58. DOI: 10.1016/j.tcs.2011.12.071.

The QUINTESSENCE Consortium (2016). Networking Our Way to Better Ecosystem Service Provision. *Trends in Ecology & Evolution* 31 (2), pp. 105–115. DOI: 10.1016/j.tree.2015.12.003.

Thompson, D. B. A., A. J. MacDonald, J. H. Marsden and C. A. Galbraith (1995). Upland heather moorland in Great Britain: A review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation* 71 (2), pp. 163–178.

Tipping, R. (1995). Holocene evolution of a lowland Scottish landscape: Kirkpatrick Fleming. Part I, peat and pollen-stratigraphic evidence for raised moss development and climatic change. *The Holocene* 5 (1), pp. 69–81. DOI: 10.1177/095968369500500108.

Tipping, R. (2008). Blanket peat in the Scottish Highlands: timing, cause, spread and the myth of environmental determinism. *Biodiversity and Conservation* 17 (9), pp. 2097–2113. DOI: 10.1007/s10531-007-9220-4.

Tollefson, J. (2015). Battle for the Amazon. *Nature* 520, pp. 20–23.

Tolvanen, A., A. Juutinen and R. Svento (2013). Preferences of Local People for the Use of Peatlands : the Case of the Richest Peatland Region in Finland. *Ecology and Society* 18 (2). 19. http://dx.doi.org/10.5751/ES-05496-180219.

Topping, C. J., T. T. Høye and C. R. Olesen (2010). Opening the black box-Development, testing and documentation of a mechanistically rich agent-based model. *Ecological Modelling* 221 (2), pp. 245–255. DOI: 10.1016/j.ecolmodel.2009.09.014.

Traag, V. A. and J. Bruggeman (2009). Community detection in networks with positive and negative links. 80 (3), 036115, p. 036115. DOI: 10.1103/PhysRevE.80.036115. arXiv: 0811.2329.

Turetsky, M. R., W. Donahue and B. Benscoter (2011). Experimental drying intensifies burning and carbon losses in a northern peatland. *Nature Communications* 2:514. DOI: 10.1038/ncomms1523.

Turetsky, M. R., B. Benscoter, S. Page, G. Rein, G. R. V. D. Werf and A. Watts (2015). Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience* 8, pp. 11–14. DOI: 10.1038/ngeo2325.

Turner, B. L., E. F. Lambin and A. Reenberg (2007). The emergence of land change science for global change and sustainability. *Proceedings of the National Academy of Sciences* 104 (52), pp. 20666–20671.

Turner, T. E., A. J. Baird, J. Holden, P. J. Chapman, M. F. Billett and K. J. Dinsmore (in preparation). Water movement though near surface blanket peats is dominated by a complicated pattern of near-surface flows.

Upham, P. and J. García Pérez (2015). A cognitive mapping approach to understanding public objection to energy infrastructure: The case of wind power in Galicia, Spain. *Renewable Energy* 83, pp. 587–596. DOI: 10.1016/j.renene.2015.05.009.

Venter, O., H. P. Possingham, L. Hovani, S. Dewi, B. Griscom, G. Paoli, P. Wells and K. A. Wilson (2013). Using systematic conservation planning to minimize REDD+ conflict with agriculture and logging in the tropics. *Conservation Letters* 6 (2), pp. 116–124. DOI: 10.1111/j.1755-263X.2012.00287.x.

Vogt, R., H. Wang, B. Gregor and A. Bettinardi (2015). Potential changes to travel behaviors & patterns: a fuzzy cognitive map modeling approach. *Transportation* 42 (6), pp. 967–984. DOI: 10.1007/s11116-015-9657-3.

Voinov, A. and F. Bousquet (2010). Modelling with stakeholders. *Environmental Modelling & Software* 25, pp. 1268–1281.

Waddington, J., P. J. Morris, N. Kettridge, G. Granath, D. Thompson and P. Moore (2015). Hyrdological feedbacks in northern peatlands. *Ecohydrology* 8, pp. 113–127.

Walker, B., S. Carpenter, J. Anderies, N. Abel, G. Cumming, M. Janssen, L. Lebel, J. Norberg, G. D. Peterson and R. Pritchard (2002). Resilience management in social-ecological systems: a working hypothesis for a participatory approach. *Conservation Ecology* 6 (1). 14. http://www.consecol.org/vol6/iss1/art14.

Wallage, Z. E., J. Holden and A. T. McDonald (2006). Drain blocking: an effective treatment for reducing dissolved organic carbon loss and water discolouration in a drained peatland. *The Science of the total environment* 367, pp. 811–821. DOI: 10.1016/j.scitotenv.2006.02.010.

Wallage, Z. E. and J. Holden (2011). Near-surface macropore flow and saturated hydraulic conductivity in drained and restored blanket peatlands. *Soil Use and Management* 27 (2), pp. 247–254. DOI: 10.1111/j.1475-2743.2011.00336.x.

Wang, H., C. J. Richardson and M. Ho (2015). Dual controls on carbon loss during drought in peatlands. *Nature Climate Change* 5, pp. 584–588.

Ward, S. E., R. D. Bardgett, N. P. McNamara, J. K. Adamson and N. J. Ostle (2007). Long-Term Consequences of Grazing and Burning on Northern Peatland Carbon Dynamics. *Ecosystems* 10 (7), pp. 1069–1083. DOI: 10.1007/s10021-007-9080-5.

Watts, C. D., P. S. Naden, J. Machell and J. Banks (2001). Long term variation in water colour from Yorkshire catchments. *Science of the Total Environment* 278 (1-3), pp. 57–72. DOI: 10.1016/S0048-9697(00)00888-3.

Wenstøp, F. (1980). Quantitative analysis with linguistic values. *Fuzzy Sets and Systems* 4, pp. 99–115.

Wheater, H. and E. Evans (2009). Land use, water management and future flood risk. *Land Use Policy* 26S (Supplement 1), S251–S264. DOI: 10.1016/j.landusepol.2009.08.019.

Whitfield, S. and M. Reed (2012). Participatory environmental assessment in drylands: Introducing a new approach. *Journal of Arid Environments* 77, pp. 1–10. DOI: 10.1016/j.jaridenv.2011.09.015.

Wiek, A., B. Ness, P. Schweizer-Ries, F. S. Brand and F. Farioli (2012). From complex systems analysis to transformational change: A comparative appraisal of sustainability science projects. *Sustainability Science* 7 (Supplement 1), pp. 5–24. DOI: 10.1007/s11625-011-0148-y.

Wilson, L., J. Wilson, J. Holden, I. Johnstone, A. Armstrong and M. Morris (2010). Recovery of water tables in Welsh blanket bog after drain blocking: Discharge rates, time scales and the influence of local conditions. *Journal of Hydrology* 391 (3-4), pp. 377–386. DOI: 10.1016/j.jhydrol.2010.07.042.

Wilson, L., J. M. Wilson and I. Johnstone (2011a). The effect of blanket bog drainage on habitat condition and on sheep grazing, evidence from a Welsh upland bog. *Biological Conservation* 144 (1), pp. 193–201. DOI: 10.1016/j.biocon.2010.08.015.

Wilson, L., J. Wilson, J. Holden, I. Johnstone, A. Armstrong and M. Morris (2011b). Ditch blocking, water chemistry and organic carbon flux: Evidence that blanket bog restoration reduces erosion and fluvial carbon loss. *Science of the Total Environment* 409 (11), pp. 2010–2018. DOI: 10.1016/j.scitotenv.2011.02.036.

Wise, L., A. G. Murta, J. P. Carvalho and M. Mesquita (2012). Qualitative modelling of fishermen's behaviour in a pelagic fishery. *Ecological Modelling* 228, pp. 112–122. DOI: 10.1016/j.ecolmodel.2011.12.008.

Witze, A. (2015). Minnesota bog study turns up the heat on peat. *Nature* 524, p. 397.

Wood, K. A., R. A. Stillman and J. D. Goss-Custard (2015). Co-creation of individual-based models by practitioners and modellers to inform environmental decision-making. *Journal of Applied Ecology* 52 (4), pp. 810–815. DOI: 10.1111/1365-2664.12419.

Worrall, F., M. J. Bell and A. Bhogal (2010). Assessing the probability of carbon and greenhouse gas benefit from the management of peat soils. *Science of The Total Environment* 408 (13), pp. 2657–2666. DOI: 10.1016/j.scitotenv.2010.01.033.

Worrall, F., J. Holden, P. Chapman, P. Smith, R. Artz, C. Evans and R. Grayson (2011). *A review of current evidence on carbon fluxes and greenhouse gas emissions from UK peatland.* Tech. rep. 442. JNCC, Peterborough.

Yallop, A. R., J. I. Thacker, G. Thomas, M. Stephens, B. Clutterbuck, T. Brewer and C. a. D. Sannier (2006). The extent and intensity of management burning in the English uplands. *Journal of Applied Ecology* 43 (6), pp. 1138–1148. DOI: 10.1111/j.1365-2664.2006.01222.x.

Yamulki, S., R. Anderson, a. Peace and J. I. L. Morison (2013). Soil CO_2 CH_4 and N_2O fluxes from an afforested lowland raised peatbog in Scotland: implications for drainage and restoration. *Biogeosciences* 10, pp. 1051–1065. DOI: 10.5194/bg-10-1051-2013.

Young, J. C., A. Watt, P. Nowicki, D. Alard, J. Clitherow, K. Henle, R. Johnson, E. Laczko, D. McCracken, S. Matouch, J. Niemela and C. Richards (2005). Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodiversity and Conservation* 14 (7), pp. 1641–1661. DOI: 10.1007/s10531-004-0536-z.

Young, J. C., M. Marzano, R. M. White, D. I. McCracken, S. M. Redpath, D. N. Carss, C. P. Quine and A. D. Watt (2010). The emergence of biodiversity conflicts from biodiversity impacts: characteristics and management strategies. *Biodiversity and Conservation* 19 (14), pp. 3973–3990. DOI: 10.1007/s10531-010-9941-7.

Young, J. C., K. Searle, A. Butler, P. Simmons, A. D. Watt and A. Jordan (2016a). The role of trust in the resolution of conservation conflicts. *Biological Conservation* 195, pp. 196–202. DOI: 10.1016/j.biocon.2015.12.030.

Young, J. C., D. Thompson, P. Moore, A. MacGugan, A. Watt and S. M. Redpath (2016b). A conflict management tool for conservation agencies. *Journal of Applied Ecology* Early view. DOI: 10.1111/1365-2664.12612.

Yu, Z., J. Loisel, D. P. Brosseau, D. W. Beilman and S. J. Hunt (2010). Global peatland dynamics since the Last Glacial Maximum. *Geophysical Research Letters* 37 (13). L13402. DOI: 10.1029/2010GL043584.